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Beneficial Use Evaluation:

Flue Gas Desulfurization Gypsum as an Agricultural Amendment

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Prepared By:

United States Environmental Protection Agency
Office of Land and Emergency Management
Office of Resource Conservation and Recovery

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Agricultural Research Service

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USDA Agricultural Research Service
U.S. DEPARTMENT OF AGRICULTURE

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Executive Summary

The United States (U.S.) Environmental Protection Agency (hereafter "EPA" or "the Agency") Sustainable Materials Management (SMM) Program supports the productive and sustainable use of resources throughout all stages of their lifecycles, from resource acquisition through disposal. The SMM Program seeks to avoid or minimize adverse impacts to the environment while accounting for economic efficiency and social considerations. The beneficial use of nonhazardous industrial materials presents a significant opportunity to advance the goals of the SMM Program by providing economic benefits, preserving natural resources, and avoiding negative environmental impacts associated with acquisition and processing of virgin raw materials. Therefore, EPA supports the beneficial use of these industrial materials when done in a manner that is protective of human health and the environment.

State, tribal and territorial regulatory bodies often make the determination whether or not to allow a given beneficial use within their jurisdiction. Although requests for such determinations have increased over time, it has been reported that insufficient information about the potential impacts to human health and the environment from these uses has been a major barrier to making decisions about proposed beneficial uses. To help address this barrier, EPA developed two documents: *Methodology for Evaluating the Beneficial Use of Industrial Non-Hazardous Secondary Materials* and *Beneficial Use Compendium: A Collection of Resources and Tools to Support Beneficial Use Evaluation* ("the Methodology" and "the Compendium," respectively). These documents provide an analytical framework that can be used to evaluate the potential for adverse environmental impacts from a wide range of industrial materials and their proposed beneficial uses, as well as a list of existing resources and tools that can assist with these evaluations.

The primary purpose of this document is to demonstrate how the analytical framework from the Methodology and Compendium can be applied to a real-world beneficial use scenario, specifically the use of flue gas desulfurization (FGD) gypsum as an agricultural amendment. FGD gypsum is a type of coal combustion residual (CCR) generated from the pollution control technologies designed to reduce sulfur gas emissions from electric utilities. FGD gypsum can substitute for mined gypsum, which is a mineral that occurs naturally in sedimentary rock formations, because both materials are composed primarily of calcium sulfate. FGD gypsum has been shown to offer a range of benefits when applied to fields, such as a providing key nutrients to crops and limiting phosphorus runoff to nearby water bodies. Yet there is also potential for higher levels of some trace contaminants in FGD gypsum as a result of the industrial process that generates this material, which warranted further evaluation to ensure that application of this industrial material will not harm human health or the environment.

As detailed in this document, EPA sequentially applied each step of the analytical framework, culminating in a national-scale probabilistic model of potential environmental fate and transport. No concerns were identified for the vast majority of modeled application scenarios. Some limited potential for risk was identified from release of selenium to surface water when FGD gypsum is applied on across every available field at the highest rates and frequencies. Yet even in this extreme and unlikely scenario, identified risks can be mitigated through minor limits on application practices. Based on these results, the beneficial use of FGD gypsum can provide meaningful benefits to agricultural fields while remaining protective of human health and the environment.

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Acronym List

ACAA	American Coal Ash Association
AD	Anderson-Darling
AERMOD	American Meteorological Society/EPA Regulatory Model
ASTSWMO	Association of State and Territorial Solid Waste Management Officials
ATSDR	Agency for Toxic Substances and Disease Registry
BCE	Before the common era
BFI	Base flow index
CCR	Coal combustion residuals
DAF	Dilution-attenuation factor
DBP	Disinfection byproducts
EIA	Energy Information Agency
EPA	Environmental Protection Agency
EPACMTP	EPA's Composite Model for Leachate Migration with Transformation Products
EPRI	Electric Power Research Institute
FGD	Flue gas desulfurization
HELP	Hydrologic Evaluation of Landfill Performance model
HQ	Hazard quotient
HUC	Hydrologic unit code
IWEM	Industrial Waste Evaluation Model
KS	Kolmogorov-Smirnov
LAU	Land application unit
LEAF	Leaching Environmental Assessment Framework
L/S	Liquid-to-solid ratio
MIDEQ	Michigan Department of Environmental Quality
MINTEQA2	Metal Speciation Equilibrium Model for Surface and Groundwater
ND	Non-detect
NHD	National Hydrography Dataset
OLEM	Office of Land and Emergency Management
ORNL	Oak Ridge National Laboratory
QA/QC	Quality assurance/quality control
SSM	Sustainable Materials Management
SPLC	Synthetic Precipitation Leaching Procedure
TCLP	Toxicity Characteristic Leaching Procedure
USDA	United States Department of Agriculture
USGS	United States Geological Survey
WSR	Wilcoxon signed-rank

1. Introduction

The United States Environmental Protection Agency ("EPA" or "the Agency") Sustainable Materials Management (SMM) Program aims to minimize the negative environmental impacts of materials through the sustainable use/reuse of resources throughout the product lifecycle, from resource acquisition through ultimate disposal. When done in a responsible manner, the beneficial use of secondary materials can advance these goals. Beneficial use involves the substitution of these secondary materials, either as generated or following additional processing, for some or all of the virgin, raw materials in a natural or commercial product (an "analogous product") in a way that provides a functional benefit, meets product specifications, and does not pose concern to human health or the environment.

Coal combustion residuals (CCRs) are the byproducts of coal combustion that are captured from plant effluent and flue gases prior to discharge to the environment. Once generated, CCRs may be either disposed of or beneficially used. Flue gas desulfurization (FGD) gypsum is one type of gypsum that is generated by the pollution control technologies intended to reduce sulfur emissions from plant stacks. One use that has been proposed for FGD gypsum is as an agricultural amendment for fields, which would replace the naturally occuring gypsum that would otherwise have to be mined.

1.1. Background

A survey of state beneficial use programs conducted by the Association of State and Territorial Solid Waste Management Officials in 2006 found that, although the number of requests for determinations is increasing, "insufficient information to determine human or ecological impacts of use rather than disposal" has been a major barrier for states when reviewing proposed beneficial uses (ASTSWMO, 2007). To help address this barrier, the EPA Office of Land and Emergency Management (OLEM) developed two documents to provide a framework that can be used to ensure that evaluations are conducted in a manner that is clear, consistent and comprehensive:

- Methodology for Evaluating Beneficial Uses of Industrial Non-Hazardous Secondary Materials (U.S. EPA, 2016a)
- Beneficial Use Compendium: A Collection of Resources and Tools to Support Beneficial Use Evaluations (U.S. EPA, 2016b)

EPA applied both documents to the evaluation of FGD gypsum in agricultural applications. EPA partnered with the U.S. Department of Agriculture (USDA) - Agricultural Research Service to ensure that all the data and assumptions relied upon in this evaluation accurately reflect current agricultural practices. The remainder of this document details the step-wise evaluation. The scope of the evaluation was limited to FGD gypsum generated in the United States through forced oxidation scrubbers following particulate collection. This evaluation does not address products that contain additional additives that may alter either the composition of or releases from FGD gypsum.

1.2. Document Organization

This beneficial use evaluation is divided into seven main sections and five appendices. The main text provides a step-wise summary of the analyses performed, model results and conclusions. At the end of each discrete analysis, a summary of the release pathways and constituents retained for further evaluation is provided. Appendices provide more detailed discussion of the data and models that underpin the analyses discussed in the main text. The remainder of this section provides a brief summary of the contents in each section and appendix.

- **Section 2 (Planning and Scoping):** This section aims to identify the questions that will be answered by the evaluation and the types of information required to answer them. The information and conceptual model defined in this section formed the basis for all subsequent data collection efforts.
- **Section 3 (Existing Evaluations):** This step consists of a literature review to identify any existing evaluations that are of sufficient quality to rely upon in the beneficial use evaluation. The purpose of this step is to avoid duplication of effort by building on previous works.
- **Section 4 (Comparison with Analogous Product):** This step consists of a comparison between the beneficial use and an analogous product made with virgin materials. The objective is to determine whether the potential for adverse impacts from the beneficial use is comparable to or lower than from an analogous product.
- **Section 5 (Screening Analysis):** This step characterizes the potential for adverse impacts from the beneficial use through a comparison with screening benchmarks. The objective is to identify individual constituents or entire exposure pathways that can be eliminated from further consideration with a high degree of confidence prior to more intensive modeling.
- **Section 6 (Risk Modeling):** This step consists of a refined, quantitative and qualitative characterization of the potential for adverse impacts from the beneficial use. The objective is to reduce remaining uncertainties enough to permit well-substantiated conclusions about the proposed use
- **Section 7 (Uncertainty and Sensitivity Analysis):** This step of consists of a review of major uncertainties associated with the model and identification of any sensitive model inputs that might drive identified risks. The goal is to discuss the key findings from the main analysis while considering the potential effects of uncertainties to reach a final set of conclusions about the proposed beneficial use.
- **Section 8 (Final Characterization):** This is the final phase for beneficial use evaluations conducted using this methodology. The objective is to integrate key findings, assumptions, limitations and uncertainties identified throughout the evaluation into final conclusions about the potential impacts to human health and the environment associated with the beneficial use.

- **Appendix A (Constituent Data):** This appendix provides a summary of the collection and management of raw data drawn from the available literature and considered in the beneficial use evaluation of FGD gypsum in agricultural applications.
- **Appendix B (Benchmarks):** This appendix describes the approach used to identify benchmarks used in this beneficial use evaluation to estimate the potential for adverse impacts to human and ecological receptors.
- **Appendix C (Use Characterization):** This appendix provides a summary of the collection and management of data used to define how FGD gypsum may be used in agricultural applications.
- **Appendix D (Screening Analysis):** This appendix provides a summary of the model inputs used to conduct the air pathway screening.
- **Appendix E (Probabilistic Modeling):** This appendix provides a summary of the data management and modeling used to model receptor exposures on a national scale.

2. Planning and Scoping

This section defines the scope of this beneficial use evaluation and details the conceptual model for the different environmental releases and associated exposures that may occur. This information helps to define the questions that the beneficial use evaluation will aim to answer and ensures that the objectives of the evaluation are well-defined, realistic, and form a sound basis for subsequent beneficial use determinations. The following subsections detail the information that forms the basis for all subsequent data collection and analytical efforts.

2.1. Background

Calcium sulfate dihydrate [$CaSO_4•2H_2O$], more commonly known as gypsum, occurs naturally in sedimentary rock formations across the globe. The utility of gypsum has been recognized for centuries, with the oldest known use as a building material dating back as early as 6,000 BCE in Anatolia and 3,700 BCE in Egypt (U.S. DOI, 2001). Naturally occurring gypsum remains a commodity of great economic value, with large quantities extracted each year from mines and quarries (hereafter referred to as "mined gypsum"). In 2014, the United States produced an estimated 17.1 million tons of mined gypsum from 17 states (U.S. DOI, 2015). Once extracted, mined gypsum may be further ground into a fine powder and heated at high temperatures to drive off the majority of chemically bound water. The resulting powder is used in a number of commercial products, such as cement and wallboard.

In recent decades, industries have explored the potential use of byproduct gypsum, which is generated as the byproduct of various industrial processes, as a substitute for mined gypsum. Because these synthetic gypsums are also composed primarily of calcium sulfate, it is sometimes possible to substitute them for mined gypsum. This is evidenced by the fact that synthetic gypsums currently account for approximately half of the of gypsum use in the United States (U.S. DOI, 2015). Yet, even though the composition of these byproduct gypsums is predominately calcium sulfate, there is the potential for higher levels of some trace contaminants introduced by the industrial processes. These contaminants may be released into the environment when synthetic gypsum is used in place of mined gypsum. Therefore, further evaluation is warranted to determine whether the use of synthetic gypsum is an appropriate beneficial use.

2.2. Flue Gas Desulfurization Gypsum

The largest source of synthetic gypsum in the United States is FGD gypsum, a CCR generated at coal-fired electric utilities by the pollution control technologies intended to reduce sulfur emissions from plant stacks. These utilities may employ any number of different pollution control devices to remove sulfur (often referred to as "flue gas desulfurization units" or "scrubbers"). These devices differ in how they remove sulfur gases, but all generate some form of FGD waste that can range from a dry powder to a wet sludge. FGD gypsum is a specific subset of the wet sludges.

Figure 2-1 illustrates a generalized layout of a coal-fired plant and the collection points for fly ash and FGD gypsum.

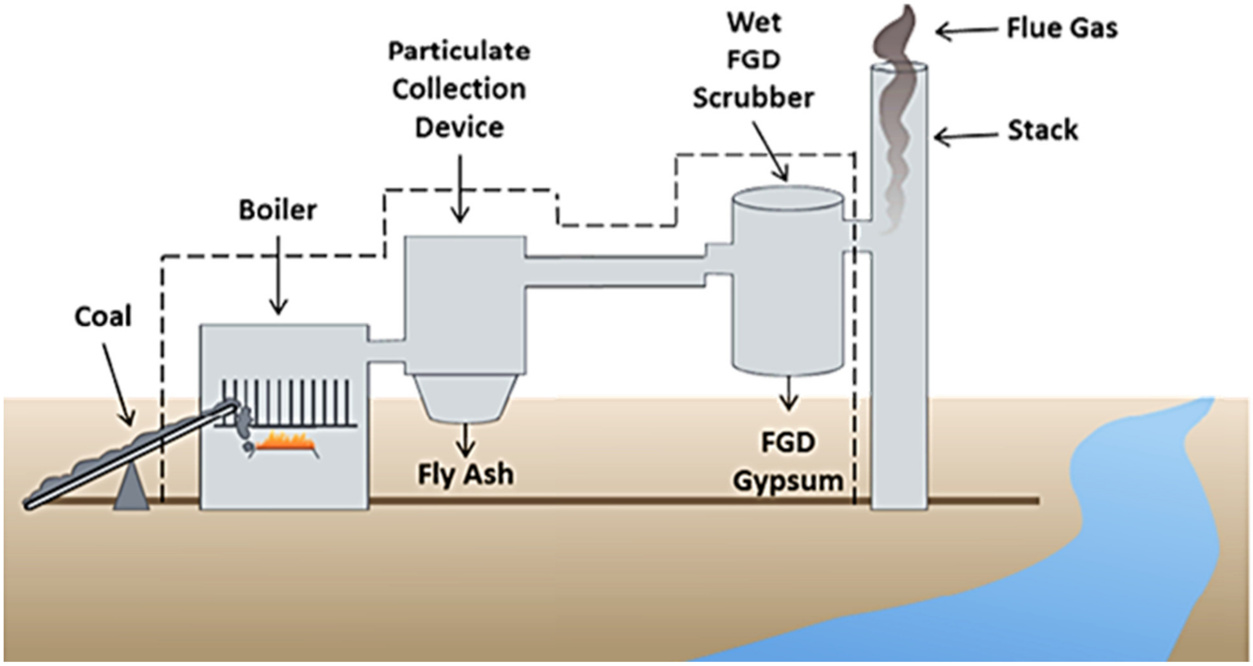

Figure 2-1: Diagram of generic coal combustion processes.

The generation of FGD gypsum begins with the removal of as much fly ash from the flue gas as practicable, which is accomplished with collection devices, such as electrostatic precipitators, bag houses or cyclones. The performance of different removal systems varies, but current technologies have achieved greater than 99% reduction of the initial particulate mass. The remaining flue gas is sprayed with a wet limestone-based reagent, which reacts with and entrains the sulfur dioxide, reducing the amount that can escape into the atmosphere. In the presence of excess oxygen, the chemical reaction between limestone and sulfur dioxide produces gypsum. To ensure that the majority of the resulting sludge is gypsum, a power plant may pump air into the chamber during the reaction in a process called "forced oxidation" (EPRI, 2008a). This process is illustrated in **Figure 2-2**.

The American Coal Ash Association (ACAA) tracks the quantities of different CCRs generated and beneficially used through voluntary annual surveys.[1] According to these surveys, approximately 23 million tons of FGD gypsum were generated in 2019. Of these, 13 million tons were beneficially used (ACAA, 2021). EPA previously conducted an evaluation of the largest single use of FGD gypsum, as a raw material for wallboard, and found it to be an appropriate use (U.S. EPA, 2014a).

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1) In 2020, the ACAA survey response rate was equivalent to 55% of the total U.S. coal-fired electric generation capacity. This estimated response rate is based on a ratio of the generating capacity of the individual plants reporting and the total coal-fired generation capacity reported by the Energy Information Administration (EIA) in 2020 (available online at: www.eia.gov/coal/data.cfm). Reported beneficial use rates were extrapolated for the entire industry sector using the 2020 survey data, historical ACAA survey data, EIA data, and other miscellaneous data sources.

This document details the beneficial use evaluation conducted for the different uses of FGD gypsum in agriculture. In 2018, approximately 1 million tons of FGD gypsum were used in various agricultural applications. This represents the fourth largest use of FGD gypsum listed in the survey, but the single largest use of any CCR in agriculture.

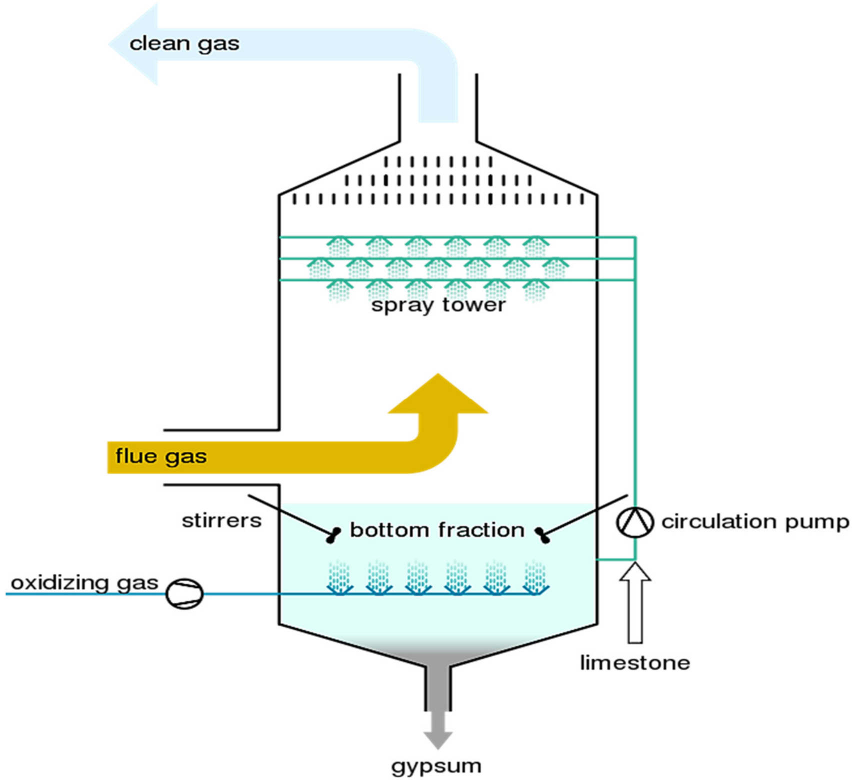

Figure 2-2: Diagram of a generic flue gas desulfurization scrubber.

After generation, FGD gypsum may be washed to reduce impurities, such as soluble salts, and to meet market specifications for products such as wallboard (Henkels and Gaynor, 1996). Yet washing also creates a new waste stream that must be managed appropriately, so there is incentive to avoid the practice if it is not required. EPA did not identify any existing legal requirements or industry standards that specify the use of washed FGD gypsum in agriculture. Thus, this beneficial use evaluation considers use of both washed and unwashed FGD gypsum. Washing is not known to be a common practice for mined gypsum and so it is not considered in this evaluation.

2.3. Agricultural Applications of Gypsum

The following subsections describe the objectives of agricultural applications of gypsum considered in this evaluation. These applications were selected based on a review of the available literature and current recommendations from state extension services. It is also important to note that inclusion of the subsequently described uses in this beneficial use evaluation does not necessarily reflect the widespread adoption of these applications at the time of the evaluation. Although the described uses have demonstrated the potential to achieve desired benefits, a particular use may still be uncommon due to high upfront costs or other barriers to application.

2.3.1. Nutrient Amendment

Calcium and sulfur are essential nutrients for plant growth and development. Calcium is incorporated in plant cell walls, where it acts as a cement between adjacent cells and is involved in cell elongation of root tip growth. Sulfur is incorporated by plants as a component of amino acids (e.g., methionine, cysteine) that are essential to the structure of proteins and involved in many enzyme activities and other functions (Jones, 1982). Insufficient amounts of these elements can inhibit plant development and decrease crop yields. Different crops will have varying sensitivities to the level of calcium or sulfur in the soil. Some crops, such as peanuts, tend to be more sensitive to calcium levels (Bledsoe et al., 1949; Cox et al., 1976; Hallock and Garren, 1968; Howe et al., 2012), while other crops, such as alfalfa and soybean, tend to be more sensitive to sulfur levels (Chen et al., 2005). Soil deficiencies are considered distinct from other physiological conditions (e.g., disease) or environmental conditions (e.g., heat stress) that might prevent plants from drawing nutrients from the soil. Calcium and sulfur are two of the primary components of gypsum. Therefore, this material can provide a concentrated source of both elements. While there are other fertilizers available on the market, the higher solubility of gypsum compared to other sources makes it an attractive source of nutrients (OSU-E, 2011).

2.3.2. Soluble Phosphorus

Some soils contain phosphorus in excess of that needed by crops as a result of current and historic application of livestock wastes and, to a lesser degree, chemical fertilizer. High excess phosphorus in surface soils can result in releases to nearby water bodies, directly through overland runoff or indirectly through subsurface tile drainage. Because algae in freshwater bodies are commonly limited by phosphorus concentrations, loading from fields may result in the eutrophication of downstream waters.

Gypsum has been shown to effectively reduce soluble phosphorus in soils with high phosphorus (Anderson et al., 1995; Stout et al., 1999; Dao, 1999; Norton, 2008; Torbert et al., 2005; Watts and Torbert, 2009; Endale et al., 2014; Torbert and Watts, 2014; Watts and Torbert, 2016; King et al., 2016). It has been suggested that this results from the formation of insoluble complexes between calcium and phosphate (e.g., hydroxyapatite, fluorapatite) (Lindsay, 1979; Brauer et al., 2005), but the exact mechanism is not yet fully understood. Under a broad range of manure loading, pH and redox conditions, gypsum has been demonstrated to reduce water-soluble phosphorus. Reductions of water-soluble phosphorus at or above 50% have been reported. While the greatest reductions in dissolved phosphorus occurred immediately after application, gypsum has also been reported to reduce phosphorus concentrations in succeeding runoff events regardless of timing, suggesting that the effect is persistent over a growing season. Improvement of water quality by reducing phosphorus runoff with gypsum is an established practice for which the National Resources Conservation Service of the USDA has developed standard conservation practices (USDA, 2015a).

2.3.3. Aluminum Toxicity

When soil pH drops below 5.2, aluminum ions (Al^{3+}) readily dissolve into water present within the pore spaces of the soil. At lower pH, the dissolved Al^{3+} concentrations may reach levels that are phytotoxic to the growing roots of crops. Some plant species have adapted to high dissolved Al^{3+}, but many commercial crops and most vegetables are sensitive to high Al^{3+} concentrations.

Gypsum has been shown to reduce the adverse effects of subsoil Al^{3+} (Reeve and Sumner, 1972; Pavan et al., 1984; Oates and Caldwell, 1985; Shainberg et al., 1989; Sumner et al., 1986, 1990). It can reduce or eliminate the negative effects of Al^{3+} in the subsoil on plant rooting because the dissolved sulfate (SO_4) penetrates to a soil depth where it reacts with Al^{3+} to form a complex ion that readily leaches from the soil. At the same time, Ca^{2+} replaces any exchangeable Al^{3+} still bound on the surfaces of clay and organic matter. For greatest effect, application would also involve mixing liming products with the subsoil layer to adjust the pH and ensure any remaining excess Al^{3+} has been removed from solution.

2.3.4. Sodic Soils

Most clay particles have a negative electrical charge on the external surface, which will repel other clay particles. However, positive cations present near the clay surface can attract (or “flocculate”) clay particles toward one another. The resulting aggregation of particles helps to form the structure of soil pores that allows infiltration (Horn et al., 1995). Divalent cations (e.g., Ca^{2+}, Mg^{2+}) have the greatest ability to flocculate clays. While monovalent ions (e.g., K^+, Na^+) also attract negatively charged clay particles, they allow a greater number of water molecules to surround each clay particle. The greater contact with water results in increased suspension and dispersal of smaller clay particles. Movement of these smaller suspended particles can clog the soil pore spaces and prevent water from infiltrating through the soil column. Reduced infiltration limits the availability of water to crop roots (USDA, 1954). Sodic soils are a particular type of dispersed soil that occurs when the soil contains high levels of exchangeable sodium (Na^+) relative to the levels of divalent cations.

Soil amendments like gypsum that contain divalent cations (e.g., Ca^{2+}) can ameliorate sodic soils by displacing Na^+ from the surface of clay particles (Suarez, 2001). Studies have shown that, when gypsum is incorporated into topsoil of sodic soils, the added Ca^{2+} promotes aggregation of clays if non-saline irrigation water is applied to leach Na^+ out of the soil (CSU-E, 2007; Suarez, 2001). This increases the percolation of rain and irrigation water, as well as the retention of water, which supports plant growth. Remediation of sodic soils with gypsum is a well-developed practice, and the USDA has included it in Standard Conservation Practice (USDA, 2010).

2.3.5. Infiltration

The concentration of ions in rainwater is many-fold lower than in soil pore water. In certain soils that contain higher levels of clay, rainfall can deplete soluble Ca^{2+} from the topsoil. If the depletion of Ca^{2+} from the soil results in an excess of monovalent cations, then the topsoil may disperse in a similar way as sodic soils. However, poor infiltration is distinct from sodic soils in that the soil has depleted calcium, rather than excessive sodium levels, and is more likely to occur primarily near the soil surface. The reduced infiltration limits the availability of water to crop roots (US Salinity Laboratory, 1954).

Soil amendments like gypsum can provide a fresh supply of Ca^{2+} to the soil, which supports clay aggregation and improves water infiltration (Norton, 1995; Norton and Dontsova, 1998; Zhang et al., 1998). Improved infiltration can be economically important in some soil series, and gypsum has been marketed in the United States based on this benefit. Utilization of gypsum to improve water infiltration is a well-developed practice that the USDA has included in the Standard Conservation Practice (USDA, 2015a). Similar benefits for non-sodic, heavy-textured soils has not yet been demonstrated in the literature, although some research on this application has been published in recent years (EPRI, 2006; OSU-E, 2011).

2.4. Conceptual Model

The *Supplemental Report to Congress on Remaining Wastes from Fossil Fuel Combustion Technical Background Document: Beneficial Use of Fossil Fuel Combustion Wastes* (U.S. EPA, 1998) identified the following types of releases to the surrounding environment that may occur from CCR products: 1) generation of dust, 2) emanation to air, 3) leaching to ground and surface water, and 4) decay of naturally occurring radionuclides. Because this evaluation addresses the beneficial use of CCRs, these findings are considered applicable to the current evaluation of FGD gypsum used in agriculture. Therefore, each of these release routes was included in the conceptual model for further consideration.

EPA developed a conceptual model with available information to help organize and visualize the different media that may come in contact with the FGD gypsum, the types of releases that may occur, and the receptors that may be exposed. This conceptual model formed the basis for all subsequent data collection and modeling efforts. Every use of gypsum considered in this evaluation involved applications directly on the ground surface or mixed into with surface soils. As a result, regardless of the specific use, the gypsum will be exposed to the same media, and the routes through which chemical constituents can be released into the environment will be fundamentally the same. Therefore, a single conceptual model was used to represent all of the uses. The exposure pathways that may be present once the FGD gypsum has been applied to the land are depicted in **Figure 2-3** and discussed in the following subsections. It is assumed that FGD gypsum will be treated as a valuable commodity prior to application and managed in a way that will minimize releases during transport and storage. Therefore, these stages of the product lifecycle were not included in the conceptual model.

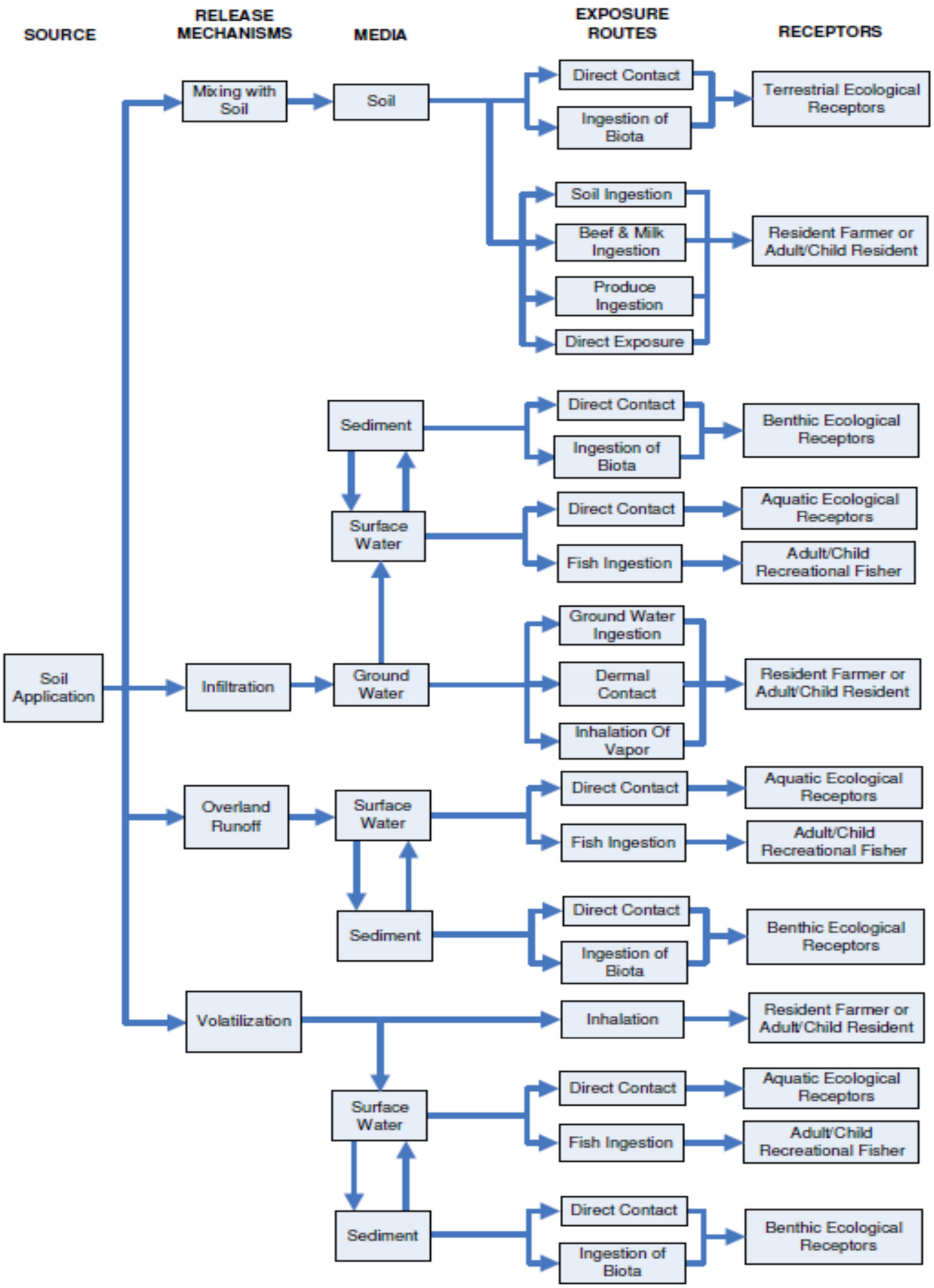

Figure 2-3: Conceptual model for FGD used in agricultural applications.

2.4.1. Mixing with Soil

Any chemical constituents present within the gypsum will be mixed with surface soils during application. Human and ecological receptors may be exposed directly to these constituents through the incidental ingestion of gypsum present in the fields or tracked into nearby homes. The constituents may also accumulate in crops grown on the gypsum-amended soil and in livestock that are fed these crops or that graze around the fields. Human receptors may be exposed to these constituents through the consumption of the resulting produce, meat and dairy.

2.4.2. Infiltration

Soluble constituents may leach into the water from precipitation and irrigation water that comes in contact with the gypsum. The fraction of the resulting leachate that is not evaporated or taken up by plants will percolate through the soil and mix with underlying ground water. Transport of the dissolved constituents will be driven by the advective flow of ground water until the water is either extracted from a well for consumption or discharged into a water body. The constituents that discharge to water bodies will mix with the surface water and either remain dissolved or settle out to the underlying sediments. Human receptors can be exposed through ingestion of and dermal contact with water drawn from private wells, as well as inhalation of any constituents that volatilize from this water. Ecological receptors may be exposed through direct contact and ingestion of the constituents in surface water and sediment. These constituents may accumulate in the tissue of fish and other aquatic receptors. Human receptors may then be exposed through the elevated concentration in biota caught from the water body.

2.4.3. Overland Runoff

Soluble constituents may leach into the water from precipitation and irrigation water that comes in contact with the gypsum. Some fraction of the runoff will flow overland and carry dissolved constituents and suspended particulates into downgradient water bodies. The constituents entering the water body will mix with the surface water and either remain dissolved or settle out into the underlying sediment. Ecological receptors may be exposed through direct contact and ingestion of the constituents in surface water and sediment. These constituents may accumulate in the tissue of fish and other aquatic receptors. Human receptors may then be exposed through the elevated concentration in biota caught from the water body.

While some of the suspended particulates may settle out onto downgradient soils, the resulting soil concentrations will always be lower than at the site of application. Because it is possible for sensitive human and ecological receptors to be present near the farm, these lower downgradient exposures were not retained for further evaluation. This evaluation also did not consider exposures from surface water used as a source of potable water. Surface water is assumed to be routed through a municipal water treatment facility prior to consumption, reducing the levels of any chemical constituents, suspended solids, pathogens, and other contaminants to below applicable standards prior to distribution. In addition, this evaluation did not consider exposures to constituents that may occur during swimming or other recreational activities near a water body. For human receptors, it is assumed that these exposures are infrequent and small in comparison to similar exposures from bathing with ground water.

2.4.4. Volatilization

Constituents with high vapor pressure may volatilize from gypsum under standard ambient conditions. Once entrained in the air, prevailing wind currents will drive constituent transport until the vapors are sequestered by fine particulates or water droplets and deposited into nearby water bodies. Ecological receptors can be exposed to constituents through direct contact with and ingestion of surface water and sediment. Human receptors may be exposed to these elevated concentrations through the consumption of fish and other biota. Human receptors may also inhale the vapors suspended in the air.

Some of the volatilized mass may also settle out onto nearby soil through a combination of dry and wet deposition. However, the resulting soil concentrations will inevitably be lower than at the initial site of application. Therefore, EPA did not explicitly evaluate exposures from deposition to downgradient soils because higher soil exposures are possible around the site of application.

3. Existing Evaluations

EPA conducted a search of the publicly available literature to identify any evaluations that had previously drawn conclusions relevant to the potential for adverse impacts associated with the use of FGD gypsum in agriculture. The purpose of this review was to avoid duplicating previous analyses that are sufficient to demonstrate whether individual constituents or entire exposure pathways pose concern to human health or the environment. The remainder of this section summarizes the identification and review of these existing evaluations.

3.1. Identification of Existing Evaluations

EPA first reviewed all of the available literature and assembled those sources that appeared to contain information on the constituent concentrations present in or released from FGD and mined gypsum. A number of relevant literature sources, in particular grey literature, had already been obtained through previous EPA or USDA investigations. Thus, EPA began with a review of the references cited in these studies. After exhausting the references in those and subsequently collected sources, EPA queried Environmental Sciences and Pollution Management, EBSCO HOST, PubMed, Science Direct, Web of Science, and JSTOR for the key words “gypsum,” “flue gas desulfurization gypsum,” “FGD gypsum,” “mined gypsum,” “natural gypsum,” and “synthetic gypsum.” Although some literature sources used other terms, such as “coal gypsum” or “FGD products,” these sources tended to be older and more ambiguous about whether the analyzed materials fit the definition of FGD gypsum. Because capturing available information on the composition and behavior of gypsum was a primary goal of the literature search, search terms related to the specific beneficial use were not used. The literature search resulted in a total of 121 unique sources, of which 70 were determined to contain potentially relevant information based on preliminary review of abstracts and tables. Further review of this subset identified two sources that had conducted analyses with the available data that might be used to draw conclusions about FGD gypsum:

- Roper et al., 2013: *Analysis of Naturally-Occurring Radionuclides in Coal Combustion Fly Ash, Gypsum, and Scrubber Residues.*
- U.S. EPA, 2014b: *Final Human and Ecological Risk Assessment of Coal Combustion Residuals.*

3.2. Review of Data Quality in Existing Evaluations

EPA reviewed existing evaluations identified in the literature according to the recommendations in *Summary of General Assessment Factors for Evaluating the Quality of Scientific and Technical Information* (U.S. EPA, 2003a). The focus of this review was to determine whether the quality of these historical evaluations was sufficient to form a defensible basis for conclusions about FGD gypsum in agriculture. The review determined whether the existing evaluations pertain to or can be used to draw conclusions about FGD gypsum (i.e., applicability and utility), clearly and sufficiently explain the data and assumptions relied upon (i.e., clarity and completeness), use analytical methods that are both reasonable and relevant to the intended application of the data (i.e., soundness), considered potential key sources of variability and uncertainty (i.e. variability and uncertainty), and had undergone sufficient external review (i.e., evaluation and review). A summary of this review for both of the existing evaluations identified during the literature review is provided in the following subsections.

3.2.1. Roper et al. (2013)

Roper et al. (2013) measured the activity of naturally occurring radionuclides in the uranium, thorium and potassium decay chains from samples of FGD gypsum collected across the United States. The study found through a direct comparison that both typical and high-end activities in FGD gypsum were similar to or lower than those reported from extensive sampling of European mined gypsum. Based on this comparison, the author concluded that levels of naturally occurring radionuclides in FGD gypsum are lower than those in mined gypsum.

Applicability and Utility:

Roper et al. (2013) explicitly measured activities in FGD gypsum collected from coal-fired utilities in the United States. As a result, these data are directly applicable to the current beneficial use evaluation. However, the study compared the FGD gypsum to mined gypsum samples collected from around Europe (El Afifi et al., 2006; Trevisi et al., 2012). Therefore, EPA reviewed the literature for supplementary information to determine whether gypsum mined in Europe could differ in composition from gypsum mined in North America.

EPA identified several studies that measured the activity of mined gypsum and wallboard from the United States (LRL, 1962; LLL, 1977; Zikovsky and Kennedy, 1992; SFDTET, 2009). Both the average and high-end values reported in these studies are similar to those measured in European samples, and so there is minimal concern that the use of European mined gypsum will substantially skew the comparison results. Therefore, the findings of Roper et al. (2013) are considered fully applicable to the current beneficial use evaluation.

Clarity and Completeness:

All of the methods and instruments used to assemble the data relied upon in both Roper et al. (2013) and the supplementary evaluations are well documented. The data relied upon are either presented in the text or documented through reference to other publicly available literature sources. Therefore, the existing evaluation is considered clear and complete.

Soundness:

The methods employed to collect and analyze the samples in Roper et al. (2013) and the supplementary evaluations conform with standard laboratory methods. Roper et al. (2013) ensured data quality through use of standards from the National Institute of Standards and Technology to calibrate equipment. Although some of the instruments used in older, supplementary studies are not the most current, the principal difference is sensitivity and ability to detect constituents at low concentrations. Use of less sensitive instruments may result in higher detection limits, but this is not a major source of uncertainty when the majority of samples have detected values. Therefore, the data used in Roper et al. (2013) and supplemental evaluations are considered sound.

Variability and Uncertainty:

The 20 FGD gypsum samples analyzed in Roper et al. (2013) are the same samples collected and analyzed for inorganic constituents in the two EPA characterizations relied upon in the current beneficial use evaluation (U.S. EPA, 2008, 2009). These samples reflect a range of coal types, pollution control technologies, and wash status found across the United States. As a result, there is reasonable confidence that these samples are representative of the variability in FGD gypsum. This conclusion is further corroborated by an uncertainty analysis discussed in **Section 7 (Uncertainty and Sensitivity Analyses)** that found that average and high-end concentrations of inorganic constituents from this set of samples are consistently either comparable to or higher than those from other data sources. Therefore, the potential to underestimate concentrations with these data is considered minimal.

Roper et al. (2013) compared the FGD gypsum data to summary statistics from over 500 mined gypsum samples collected across Europe. The supplemental sources provided data on an additional 38 samples. Therefore, there is reasonable confidence that the variability of North American mined gypsum has generally been captured by the available data. Although some of the mined gypsum data sources are one or more decades old, this is not a major source of uncertainty because geological background levels are unlikely to shift dramatically over time without significant anthropogenic disturbance.

Evaluation and Review:

Roper et al. (2013) and all of the supplemental evaluations have been published in peer-reviewed journals. There are no known conflicts of interest for the authors of Roper et al. (2013) that might diminish their capacity to provide an impartial, technically sound, and objective analysis. EPA did not review the backgrounds of each author in supplemental evaluations, as these studies were used only as sources of raw data. While there may be the potential for bias in sample collection in a particular study, the large number of samples available from across Europe makes it unlikely that this would impact the conclusions in Roper et al. (2013). Therefore, these evaluations are considered to have undergone sufficient evaluation and review.

3.2.2. U.S. EPA (2014b)

U.S. EPA (2014b) evaluated the potential risks to human health and the environment associated with the range of known disposal practices for different CCRs, including FGD gypsum. As part of this assessment, EPA conducted a screening analysis on uncontrolled releases of windblown dust from landfills at the point of highest air concentration. Constituent concentrations in the dust were estimated using high-end values drawn from all different CCR types (e.g., fly ash, FGD gypsum). This analysis found that the risks from inhalation of chemical constituents present in the dust are below levels of concern, but potential risks are possible from inhalation of fine particulate matter with a diameter less than 2.5 µm ($PM_{2.5}$). These small particulates pose greater health risks because of the potential to be inhaled more deeply into the lungs. Based on this screening-level evaluation, EPA concluded that uncovered CCR landfills have the potential to result in $PM_{2.5}$ concentrations nearly twice the relevant national ambient air quality standards (NAAQS).

Applicability and Utility:

The same inhalation exposure pathway exists for FGD gypsum used in agriculture; however, there is a high degree of confidence that the releases from agriculture will be less frequent and of lower magnitude than those from uncovered landfills. High-end constituent concentrations in FGD gypsum are often lower than those modeled in U.S. EPA (2014b) due to consideration of fly ash in that evaluation. The potential for generation of dust is also much lower from agricultural practices. Dust generation by wind or mechanical disturbance will be highest during application, which will be infrequent and relatively short in duration. During the remainder of the year, releases will be at a much lower magnitude because the gypsum is not piled high above the ground surface and the presence of plant cover will further limit wind erosion. Because the releases of inorganic elements from FGD gypsum used in agriculture will be lower than those from CCR placed in uncovered landfills, EPA determined that the findings of U.S. EPA (2014b) can be extrapolated to draw conclusions about the inorganic constituents in FGD gypsum.

U.S. EPA (2014b) identified the potential for adverse impacts associated with particulate matter. EPA reviewed the literature for supplementary information to determine whether the use of FGD gypsum in agriculture may result in similar risks. Studies have found that the distribution of particle sizes in FGD gypsum is dominated by particles with a diameter greater than 10 µm. Under the controlled conditions in the power plant, the gypsum particles tend to precipitate fairly uniformly with respect to particle size and shape. In addition, the smallest particles tend to be washed out during dewatering (Henkles and Gaynor, 1996; U.S. DOE, 2005). This results in a distribution skewed towards particle sizes larger than those modeled in U.S. EPA (2014b), with median and low-end (10th percentile) particle diameters closer to 50 µm and 20 µm, respectively. In contrast, several samples of fly ash had median particle diameters approaching 10 µm and all had low-end diameters less than 10 µm. Because potential releases of fine respirable particulates from FGD gypsum used in agriculture will be lower than those from CCR placed in uncovered landfills, EPA determined that the findings of U.S. EPA (2014b) can be extrapolated to draw conclusions about the inorganic constituents in FGD gypsum.

Clarity and Completeness:

All of the methods used to assemble and analyze the data relied upon in U.S. EPA (2014b) are well documented. All data and assumptions used in the screening model are summarized in the text. Therefore, the information contained in U.S. EPA (2014b) is considered clear and complete for the intended use.

Soundness:

U.S. EPA (2014b) relied on data assembled from a number of sources to conduct the evaluation. These samples were collected and analyzed through various methods, all of which are validated consistent with current standards. The principal difference between the instruments is the relative sensitivity, which determines the ability to detect constituents at low concentrations. This is not a major concern, because the evaluation relied on upper percentile concentrations that reflect detected values. Therefore, all of the data used in this evaluation are considered sound.

The American Meteorological Society/EPA Regulatory Model (AERMOD) was used in U.S. EPA (2014b) to screen potential exposures from inhalation. AERMOD is a regulatory steady-state plume model that estimates the amount of atmospheric dispersion and deposition during windblown transport. This model has undergone validation and been found to provide reasonable estimates of downwind air concentrations and deposition rates (U.S. EPA, 2003b). Therefore, the use of this model in the evaluation is considered sound.

Variability and Uncertainty:

The data used in U.S. EPA (2014b) reflect the range of coal types and pollution control technologies found across the United States. There is a high degree of confidence that these data provide a good estimate of the variability of these secondary materials. In addition, the bulk content (i.e., mg/kg) of inorganic constituents and small particulates used in U.S. EPA (2014b) include all types of CCR, including fly ash. Measured levels of inorganic constituents and small particulates are frequently lower in FGD gypsum than fly ash. Therefore, EPA concluded that application of the data from U.S. EPA (2014b) adequately address variability and uncertainty by providing an upper bound on potential exposures.

Evaluation and Review:

The database used in U.S. EPA (2014b) contains data collected over a series of regulatory activities between 1998 and 2010. These data were either collected and analyzed by EPA or were provided by States, public advocacy groups, or regulated facilities. Many of the samples provided from outside parties were originally collected as part of regulatory compliance activities. Given the large overall number of samples and the reliance on high-end concentrations to draw conclusions, there is minimal concern that the data relied upon is biased in a way that would underestimate exposures.

The draft of U.S. EPA (2014b) (*Draft Human and Ecological Risk Assessment of Coal Combustion Residuals*; U.S. EPA, 2009a) was submitted for both peer review and public comment as part of a proposed Agency rulemaking. In response to those comments, EPA revised the risk assessment and replied to the peer and substantive public comments in two separate documents (U.S. EPA 2014c,d). Because the full extent of the data and analyses have been subject to review and comment by independent experts and the public, this evaluation is considered to have undergone sufficient evaluation and review.

3.3. Application of Findings from Existing Evaluations

After the existing evaluations were identified and determined to be of adequate quality, the findings were considered in light of all supporting information to reach conclusions about FGD gypsum used in agriculture. If the available information was adequate to demonstrate that the potential for adverse impacts is comparable to or lower than from an analogous product, or at or below relevant regulatory and health-based benchmarks, then no further evaluation is warranted for that constituent or exposure pathway. A summary of the conclusions is provided in the following subsections.

3.3.1. Roper et al. (2013)

Roper et al. (2013) found that the activity of naturally occurring radionuclides in FGD gypsum is comparable to or lower than in European mined gypsum. Supplemental information shows that North American mined gypsum is comparable to European gypsum, so there is little concern that this comparison overestimates mined gypsum activity. Based on the review of all available information, EPA concluded that the activity of naturally occurring radionuclides in FGD gypsum is comparable to or lower than that in mined gypsum. Therefore, direct exposure to radiation from FGD gypsum applied to the soil was not retained for further evaluation.

3.3.2. U.S. EPA (2014b)

U.S. EPA (2014b) found that windblown dust from uncovered landfills does not pose concern from inorganic constituents, but might for small particulates. Knowledge of how the gypsum will be used is sufficient to demonstrate that the magnitude of releases of FGD gypsum will be far lower when used in agricultural applications. Supplemental information shows that the many inorganic constituents and small particulates are present at lower levels in FGD gypsum than in the fly ash modeled in U.S. EPA (2014b). Given the relatively low exceedance of NAAQS identified for uncovered landfills, the potential reductions in releases associated with use of FGD gypsum in agriculture will be sufficient to reduce the potential exposures to below levels of concern, provided that the gypsum is managed as a valuable product. Therefore, EPA concluded that exposures to inorganics and small particulates from windblown dust do not pose a concern to human health. Therefore, these constituents were not retained for further evaluation. However, constituents that may volatilize (i.e., mercury) were retained for the air pathway.

3.4. Review of Available Literature

At this stage of the beneficial use evaluation, EPA reviewed all of the remaining literature sources assembled and identified the constituents that could be carried forward for quantitative evaluation for each exposure pathway based on the availability of constituent data needed to characterize releases and toxicological data needed to characterize the risks from exposure:

- EPA assembled a database of all the available constituent data. This database includes information on the identity and concentration of the various constituents that may be present in and released from both FGD and mined gypsums. A more detailed discussion about the development and management of this database can be found in **Appendix A (Constituent Data)**.
- EPA identified toxicity values for human health and ecological receptors according to a hierarchy of data sources. A more detailed discussion about the selection of these values can be found in **Appendix B (Benchmarks)**.

A total of 23 unique constituents were identified that had not been addressed by existing evaluations and that had sufficient constituent data and toxicity values to characterize the risks from potential exposures. Constituents that may be present in FGD gypsum, but that could not undergo a quantitative evaluation are discussed further in **Section 7 (Uncertainty and Sensitivity Analyses).** The list of constituents that were carried forward for quantitative analysis is presented in **Table 3-1**.

Table 3-1. Constituents Retained for Comparison with Analogous Product

Constituent	CASRN	Human Health				Ecological		
		Soil	Ground Water	Surface Water	Air	Soil	Surface Water	Sediment
Aluminum	7429-90-5	×	×	×	--	--	×	--
Antimony	7440-36-0	×	×	--	--	×	×	×
Arsenic	7440-38-2	×	×	×	--	×	×	×
Barium	7440-39-3	×	×	×	--	×	×	×
Beryllium	7440-41-7	×	×	×	--	×	×	--
Boron	7440-42-8	×	×	--	--	×	×	--
Cadmium	7440-43-9	×	×	×	--	×	×	×
Chloride	16887-00-6	--	--	--	--	--	×	--
Chromium	7440-47-3	×	×	×	--	×	×	×
Cobalt	7440-48-4	×	×	--	--	×	×	×
Copper	7440-50-8	×	×	--	--	×	×	×
Iron	7439-89-6	×	×	×	--	--	×	--

Table 3-1. Constituents Retained for Comparison with Analogous Product

Constituent	CASRN	Human Health				Ecological		
		Soil	Ground Water	Surface Water	Air	Soil	Surface Water	Sediment
Lead	**7439-92-1**	×	×	--	--	×	×	×
Magnesium	**7439-95-4**	--	--	--	--	--	×	--
Manganese	**7439-96-5**	×	×	×	--	×	×	×
Mercury	**7439-97-6**	×	×	×	×	×	×	×
Molybdenum	**7439-98-7**	×	×	×	--	×	×	--
Nickel	**7440-02-0**	×	×	×	--	×	×	×
Selenium	**7782-49-2**	×	×	×	--	×	×	×
Strontium	**7440-24-6**	×	×	×	--	×	×	--
Thallium	**7440-28-0**	×	×	×	--	×	×	--
Vanadium	**7440-62-2**	×	×	×	--	×	×	×
Zinc	**7440-66-6**	×	×	×	--	×	×	×

× - Retained for further evaluation

4. Comparison with Analogous Product

FGD gypsum has been shown to function as a replacement for mined gypsum in some agricultural applications. However, this secondary material may contain higher concentrations of inorganic constituents, which accumulate in the limestone spray along with sulfur dioxide gas. To determine if the beneficial use of this secondary material may result in higher releases of and subsequent exposures to these constituents, EPA conducted a comparison of FGD and mined gypsums with the data available. This section details the data relied upon, the approaches used to qualitatively compare the materials, the other lines of evidence considered, and the Agency's interpretation of results.

4.1. Comparison Approach

This section describes the primary statistical approach used to compare FGD and mined gypsum. All of the data used in these comparisons were drawn from the gypsum database discussed in **Appendix A (Constituent Data)**. When mined gypsum data were not available to conduct a quantitative comparison, the constituents were automatically retained for further evaluation. Additional quantitative and qualitative lines of evidence were also considered when available; these pathway-specific considerations are discussed in the subsections dedicated to that type of release.

4.1.1. Handling of Non-detect Data

The comparison of some constituents was complicated by a large number of non-detects in one or both of the datasets. These non-detect values were not always the lowest values reported for a given constituent due to the variable detection limits found across different studies. To best address each comparison based on the amount of detected data, EPA binned the constituents for each release pathway into one of three groups:

- **Group 1:** Where non-detects account for less than 20% of both datasets, there was reasonable confidence that the number of non-detects would not interfere with the conclusions of the statistical tests. Selected statistical tests were conducted with non-detects set to the reported detection limit based on the requirements of the statistical tests.

- **Group 2:** Where non-detects account for 20% to 50% of either dataset, EPA used bootstrapping to fill data gaps prior to comparison. This involved fitting the detected data to a gamma, lognormal or Weibull distribution and selecting the distribution with the best agreement based on log-likelihood statistics. The selected distribution was then randomly sampled 1,000 times for each non-detect value observed in the dataset at values below each of the corresponding detection limits, as this was the highest value that may be present. All the sampled values were then arranged in order from smallest to largest and the median of consecutive sets of 1,000 random numbers was calculated. Each non-detect value was then replaced with one of these median values. The resulting distribution was used in the comparison with all values treated as detections. Similar methods have previously been described by Frey and Zhao (2004) and Zhao and Frey (2004).

- **Group 3:** Where non-detects account for greater than 50% of either dataset, a statistical test was not conducted because it is unlikely that a statistical test would provide a reliable estimate of comparability. Because bootstrapping and other available methods used to fill the data gaps rely on the detected data, there is too much uncertainty introduced by the fitted distribution when non-detects represent a majority of the dataset. Therefore, statistical comparisons were not conducted for these constituents. Instead, EPA weighed other available lines of evidence. When these lines of evidence involved a comparison between datasets, EPA set all non-detect data to half the reported detection limit according to the recommendations in *Risk Assessment Guidance for Superfund (RAGS) Part A* (U.S. EPA, 1989) and *EPA Region 3 Guidance on Handling Chemical Concentration Data near the Detection Limit in Risk Assessments* (U.S. EPA, 1991).

4.1.2. Separation of Washed and Unwashed FGD Gypsum

EPA separated the available FGD gypsum data into three sets based on wash status: washed, unwashed and unknown. The Agency compared washed and unwashed data to determine whether there was a substantial difference between these two materials that might skew the results of the evaluation. Because unknown data could not be reliably sorted into one of these two categories, it was included in this comparison as a separate category. When washed and unwashed samples were found to be substantially different, the three datasets were kept separate and unknown data were excluded from the evaluation. When washed and unwashed data were found to be comparable, EPA combined all three sets of data into a single dataset. When both washed and unwashed measurements were available for a sample in this single dataset, the data were averaged to avoid biasing subsequent analyses toward those with multiple measurements.

4.1.3. Statistical Tests

Distributions for environmental data are often positively skewed, with a longer tail in the direction of higher concentrations (U.S EPA, 2006). The parametric distributions that best describe the concentrations present in and released from gypsum are likely to differ among constituents. Nonparametric tests were selected because these tests avoid assumptions about both the parametric form of distributions and the exact values of non-detect samples (U.S. EPA, 2010). EPA used one or more of the following statistical tests, depending on the amount of data available:

- The Kolmogorov-Smirnov (KS) test compares continuous distributions to test the hypothesis that the distributions are the same. This test can detect differences anywhere along the range of the data, although it is most sensitive to differences around the median (Darling, 1957).

- The Anderson-Darling (AD) test is similar to the KS test, but is more sensitive to differences at the tails of the distributions (Engmann and Cousineau, 2011).
- The Wilcoxon Signed-Ranks (WSR) test is analogous to the parametric paired t-test. This test compares the similarity of distribution medians (Hollander et al., 2013).

When sufficient data were available, EPA used both the KS and AD tests to compare samples because these tests compare entire distributions, rather than just medians. Both tests were used because neither was more robust for the purposes of this evaluation. In addition, agreement between these two tests provides greater certainty in the results obtained. Yet the KS and AD tests both require a sufficient sample size to compute the percentiles used in these tests. If the sample size was too small, then too many percentiles would require interpolation and the comparison would depend too heavily on the interpolation method used. Therefore, when datasets were judged to be too small to support these tests, EPA used the WSR test instead.

For all tests, the calculated p-value was compared to a confidence level representing the acceptable likelihood of incorrectly rejecting the null hypothesis (i.e., distributions are the same). When the p-value is lower than the specified confidence level, the null hypothesis is rejected in favor of the alternative hypothesis (i.e., distributions are different). For this evaluation, a confidence level of 90% ($\alpha = 0.10$) was selected because of the two-tailed test. This results in around a 5% chance of concluding that concentrations in FGD gypsum are either higher or lower than mined gypsum when the two distributions are actually the same. When p-values fell below this value, EPA inspected the distributions to determine which had the higher concentrations. Where releases of mined gypsum were found to be significantly higher than those of FGD gypsum, the constituent was removed from further consideration. When more than one test was used, the results were compared for agreement. And, because the potential for error cannot be entirely eliminated, EPA considered other quantitative and qualitative lines of evidence to corroborate the statistical results.

4.2. Comparison for Releases to Soil

EPA reviewed the data available in the constituent database to determine how best to compare the exposures that may result from mixing gypsum with surface soil. The exposures pathways from the conceptual model considered in the comparison were direct contact with and ingestion of soil, as well as ingestion of produce and animal products raised on the soil. The magnitudes of these exposures are directly proportional to the bulk content of constituents in the soil column. The factor driving accumulation of these constituents in the soil is the bulk content in the gypsum. Thus, EPA concluded that a comparison of bulk content in gypsum would provide a suitable surrogate for exposures that may result from releases to soil.

4.2.1. Comparison of Washed and Unwashed FGD Gypsum

EPA identified seven sample pairs in the available data that had been collected from the same facilities both before and after washing. These samples provide the most direct comparison of the changes that can result from washing. In addition, these samples represent a range of coal sources and pollution control technologies (U.S. EPA, 2009). Due to the small sample size, EPA used the WSR test to compare the bulk content of constituents in washed and unwashed FGD gypsum. The results of this comparison are presented in **Table 4-1**, with instances of higher unwashed concentrations highlighted.

Table 4-1. Comparison of Washed and Unwashed FGD Gypsum Bulk Content

Constituent	Unwashed		Washed		WSR p-value	Result
	Detection Frequency	Median Value (mg/kg)	Detection Frequency	Median Value (mg/kg)		
Group 1						
Aluminum	7 / 7	959	7 / 7	1,836	0.297	Comparable
Antimony	7 / 7	1.6	7 / 7	1.9	0.866	Comparable
Arsenic	7 / 7	3.5	7 / 7	2.3	0.176	Comparable
Barium	7 / 7	10.0	7 / 7	14.0	0.462	Comparable
Cadmium	7 / 7	0.30	7 / 7	0.40	0.834	Comparable
Chloride	**7 / 7**	**1,639**	**7 / 7**	**275**	**0.016**	**Unwashed Higher**
Chromium	7 / 7	9.1	7 / 7	7.8	0.938	Comparable
Cobalt	7 / 7	2.3	7 / 7	2.6	0.578	Comparable
Iron	7 / 7	1,610	7 / 7	1,583	0.578	Comparable
Lead	7 / 7	0.90	7 / 7	1.6	0.681	Comparable
Mercury	7 / 7	0.54	7 / 7	0.49	0.295	Comparable
Molybdenum	7 / 7	3.1	7 / 7	3.7	0.375	Comparable
Selenium	7 / 7	4.9	7 / 7	4.5	0.208	Comparable
Strontium	7 / 7	289	7 / 7	281	0.529	Comparable
Thallium	7 / 7	0.60	7 / 7	0.60	0.423	Comparable
Group 3						
Beryllium	0 / 0	--	0 / 0	--	--	--
Boron	0 / 0	--	0 / 0	--	--	--
Copper	0 / 0	--	0 / 0	--	--	--
Manganese	0 / 0	--	0 / 0	--	--	--
Nickel	0 / 0	--	0 / 0	--	--	--
Vanadium	0 / 0	--	0 / 0	--	--	--
Zinc	0 / 0	--	0 / 0	--	--	--

Chloride is the only constituent in this dataset that exhibited both a large and consistent difference between washed and unwashed samples. These results are supported by the fact that a primary goal of washing is to reduce the amount of soluble salts, such as chlorides (Gustin and Ladwig, 2010). There is no indication that current washing practices substantially decrease the bulk content of other constituents. In fact, comparison of individual sample pairs in the constituent database shows that measured concentrations in washed samples can be higher than those in corresponding unwashed samples. These increases are not isolated to certain sample pairs or constituents, making them unlikely to be the result of sampling or analytical error. Instead, this indicates that losses from washing are minor enough to be masked by a combination of sample heterogeneity, matrix interference, and other sources of measurement uncertainty. Because further comparison of washed and unwashed samples collected from different sources and analyzed in different laboratories would only compound this uncertainty, EPA did not conduct a direct comparison with the full dataset.

As a secondary line of evidence, EPA calculated the constituent mass lost during washing as a percentage of the bulk content for all available samples of unwashed FGD gypsum. EPA compared both the 90th and 50th percentile values for both variables to determine if mass lost to leaching represented a substantial and consistent fraction of the initial mass present. Relative mass loss was calculated using LEAF Method 1316, which measures leaching (mg/L) as a function of the liquid-to-solid (L/S) ratio, multiplied by the relevant L/S ratio (L/kg). EPA selected samples at a L/S ratio of 2:1 based on recommended wash ratios of between 1.5:1 and 2.5:1 to remove high solute concentrations (Genck et al., 2008). The results of this comparison are presented in **Table 4-2**, with instances of high and consistent losses highlighted. Reported values are rounded to the nearest whole percent. This comparison does not rely on statistical tests and so samples were not divided into separate groups.

Table 4-2. Relative Mass Loss from Unwashed Gypsum

Constituent	Detection Frequency		Percent Mass Lost	
	Unwashed Bulk Content	Method 1316 Mass Loss	90th Percentile	50th Percentile
Aluminum	21 / 21	4 / 11	0%	0%
Antimony	21 / 21	5 / 11	0%	1%
Arsenic	27 / 29	3 / 11	0%	0%
Barium	21 / 21	11 / 11	0%	1%
Beryllium	7 / 10	0 / 11	5%	14%
Boron	**11 / 11**	**11 / 11**	**65%**	**33%**
Cadmium	21 / 21	6 / 11	4%	3%
Chloride	**18 / 18**	**11 / 11**	**100%**	**100%**
Chromium	21 / 21	9 / 11	0%	0%
Cobalt	20 / 21	6 / 11	3%	1%
Copper	14 / 16	10 / 11	2%	2%
Iron	28 / 28	4 / 11	0%	0%
Lead	20 / 21	6 / 11	1%	1%
Manganese	**10 / 10**	**11 / 11**	**40%**	**34%**
Mercury	35 / 35	7 / 11	0%	0%
Molybdenum	21 / 21	11 / 11	15%	9%
Nickel	10 / 10	11 / 11	39%	5%
Selenium	29 / 29	11 / 11	11%	11%
Strontium	21 / 21	11 / 11	2%	1%
Thallium	19 / 19	9 / 11	2%	7%
Vanadium	10 / 10	10 / 11	2%	1%
Zinc	10 / 10	11 / 11	5%	5%

The results of this comparison indicate high and consistent mass loss for chloride relative to the bulk content of these constituents, which agrees well with the chloride results presented in **Table 4-1.** Similarly, high and consistent loss was observed for boron and manganese. Based on the available lines of evidence, EPA concluded that there is a clear difference between the bulk content of boron, chloride and manganese in washed and unwashed gypsum. For these three constituents, washed and unwashed data were kept separate in subsequent analyses.

The results of this comparison indicate moderate mass loss for molybdenum, nickel and selenium. However, these losses were not consistent between the 90th and 50th percentiles. The results also disagree with the results presented in **Table 4-1**, which did not identify any differences between washed and unwashed samples for molybdenum and selenium. A review of the raw data found that the upper percentile losses for these constituents were driven by a single high measurement, which may skew results. In addition, the unwashed sample with the highest molybdenum loss had a corresponding washed sample with a higher measured bulk content. Therefore, it is likely that differences for these three constituents fall within the bounds of measurement uncertainty and cannot be reliably identified. Based on these lines of evidence, EPA concluded that it would be inappropriate to base recommendations for the management of FGD gypsum on these differences. Thus, all available data were combined into a single set for these constituents.

For the remaining constituents, there is no evidence that current washing practices substantially reduce bulk content. These results agree well with those presented in **Table 4-1.** While there are isolated reports in the literature of larger reductions for some constituents, such as mercury, the same studies note this behavior is unusual (Gustin and Ladwig, 2010). However, because such losses cannot be reliably predicted, EPA concluded it would be inappropriate to base recommendations for the management of FGD gypsum on these differences. Thus, all available data were combined into a single dataset.

4.2.2. Comparison of Mined and FGD Gypsum

EPA assembled all the available data for mined and FGD gypsum for a direct statistical comparison. Because of the greater number of samples available, EPA relied on both the KS and AD tests. This comparison will tend to underpredict the relative constituent mass applied from mined gypsum. The purity of this material can be as low as 66% and is frequently less than that of FGD gypsum, which is consistently at or above 95% (Henkels and Gaynor, 1996; OSU-E, 2011). Somewhat higher mined gypsum application rates would be needed to achieve the same calcium sulfate loading onto agricultural fields, which will result in higher mass loading of chemical constituents from mined gypsum than assumed by a direct comparison of the materials. EPA directly compared mined and FGD gypsum because it was difficult to incorporate variable mass loading in the comparisons and there is an added degree of confidence when concentrations in mined gypsum are found to be comparable or higher than FGD gypsum. Results of this comparison are presented in **Table 4-3**, with instances of higher FGD concentrations highlighted. Because some p-values are extremely small, the reported values are truncated for values < 0.001 for ease of presentation.

Table 4-3. Comparison of FGD and Mined Gypsum Bulk Content

Constituent	Washed Status	FGD		Mined		KS p-value	AD p-value	Result
		Detection Frequency	Median (mg/kg)	Detection Frequency	Median (mg/kg)			
Group 1								
Aluminum	--	53 / 53	380	15 / 15	1,516	0.015	0.032	Mined Higher
Antimony	--	50 / 54	0.60	15 / 17	0.44	0.475	0.475	Comparable
Barium	--	55 / 55	12.0	17 / 17	12.0	0.644	0.604	Comparable
Boron	**Unwashed**	**11 / 11**	**51.0**	**14 / 14**	**8.9**	**0.001**	**< 0.001**	**FGD Higher**
	Washed	20 / 20	8.6	14 / 14	8.9	0.818	0.830	Comparable
Cadmium	**--**	**50 / 55**	**0.14**	**13 / 16**	**0.05**	**0.001**	**0.006**	**FGD Higher**
Chromium	**--**	**57 / 60**	**2.9**	**17 / 18**	**1.5**	**0.042**	**0.003**	**FGD Higher**
Iron	--	65 / 65	1,000	18 / 18	1,133	0.481	0.152	Comparable
Manganese	Unwashed	10 / 10	27.0	16 / 16	28.0	0.417	0.688	Comparable
	Washed	20 / 20	7.5	16 / 16	28.0	0.002	0.005	Mined Higher
Mercury	**--**	**94 / 96**	**0.34**	**17 / 19**	**0.002**	**< 0.001**	**< 0.001**	**FGD Higher**
Molybdenum	--	52 / 56	0.95	16 / 18	0.77	0.143	0.213	Comparable
Nickel	--	41 / 47	1.3	16 / 18	1.6	0.526	0.379	Comparable
Strontium	--	50 / 50	161	15 / 15	1,140	< 0.001	< 0.001	Mined Higher
Vanadium	--	38 / 41	2.0	16 / 17	3.0	0.554	0.358	Comparable
Zinc*	--	43 / 44	7.0	18 / 18	5.9	0.121	0.043	Mined Higher
Group 2								
Arsenic	**--**	**56 / 69**	**2.8**	**11 / 16**	**1.6**	**< 0.001**	**< 0.001**	**FGD Higher**
Beryllium	**--**	**23 / 41**	**0.04**	**8 / 14**	**0.01**	**0.038**	**0.026**	**FGD Higher**
Cobalt	--	40 / 55	0.45	16 / 18	0.62	0.265	0.243	Comparable
Copper	--	38 / 50	1.5	17 / 18	1.6	0.532	0.206	Comparable
Lead	--	45 / 53	1.1	14 / 18	1.7	0.608	0.398	Comparable
Selenium	**--**	**64 / 69**	**5.6**	**11 / 15**	**0.21**	**< 0.001**	**< 0.001**	**FGD Higher**
Thallium	**--**	**43 / 45**	**0.02**	**12 / 15**	**0.01**	**0.080**	**0.010**	**FGD Higher**
Group 3								
Chloride	Unwashed	18 / 18	833	0 / 0	--	--	--	--
	Washed	14 / 14	219	0 / 0	--	--	--	--

(ND): The reported value is the same as half the detection limit.

* Though the median FGD gypsum concentration is higher, the distribution for mined gypsum has a tail with the highest concentrations. This explains the difference between the results of the KS and AD tests.

The results of this comparison indicate that there is a potential for higher concentrations of arsenic, beryllium, boron (unwashed), cadmium, chromium, mercury, selenium and thallium in FGD gypsum. Many of these constituents have higher volatility and, as a result, are more likely to pass through particulate control technologies and become entrained in the limestone slurry. Previous studies have shown that an appreciable fraction of boron, mercury, selenium and the halogen group (e.g., bromide, chloride, fluoride, iodide) can be released from flue gas stacks as vapor (Cheng et al., 2009). Other studies have found that a much smaller fraction (around 1%) of arsenic can also be released as vapor (Meij and Alderliesten, 1989). These findings are further corroborated by the larger relative differences seen between median FGD and mined gypsum concentrations for the more volatile constituents (i.e., boron, mercury, selenium) compared to the less volatile constituents (i.e., arsenic, beryllium, cadmium, chromium, thallium). **Figure 4-1** presents a list of constituents found in coal ranked by volatility, with the constituents that may become concentrated in FGD gypsum based on this comparison and reports in the literature highlighted in yellow.

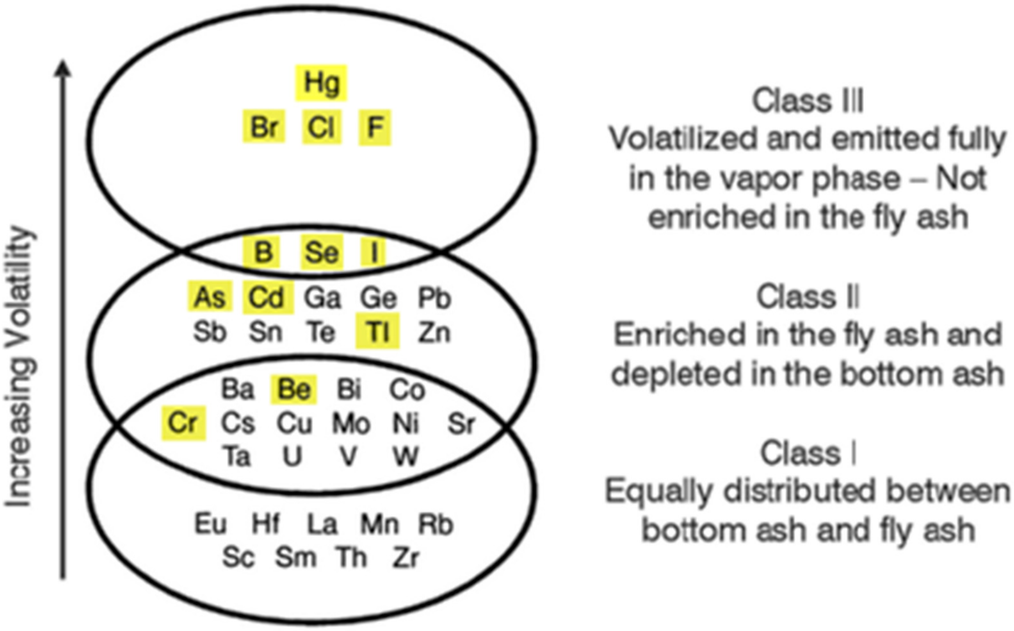

Figure 4-1: Categorization of trace elements based on partitioning in flue gas (adapted from Clarke and Sloss, 1992)

The more semi-volatile constituents may be present in the vapor phase initially but will condense and/or nucleate out of the flue gas as very fine particles (i.e., below 1 µm in diameter) due to large drops in temperature following combustion. Particulate control devices are the least effective at removal of these fine particulates, so there is the potential for the entrainment of these particulates in the limestone slurry to contribute additional constituent mass to the gypsum. However, substantial enrichment was not observed for all of the semi-volatile constituents, including lead and antimony. Because metals with similar volatility are expected to partition similarly across air pollution control systems (U.S. EPA, 1996a), the entrainment of small particulates alone does not explain differences observed for arsenic, beryllium, cadmium, chromium or thallium.

Studies have shown that the slurry sprayed into the scrubber can account for over 90% of the mass in FGD gypsum for many semivolatile constituents, such as arsenic, cadmium, lead and zinc

(Gutberlet, 1984; Gutberlet et al., 1985). Because much of the constituent mass is geogenic in origin, it is reasonable that, even for the constituents found to be statistically different, there is considerable overlap with distributions for mined gypsum. The highest measured concentrations of chromium, cadmium and zinc were found in mined gypsum. Thus, it is possible that some of the observed differences are driven more by natural variations in the parent minerals, rather than contributions from the combustion of coal.

Based on the results of this comparison, there is a high degree of confidence that the bulk content of boron, mercury and selenium can be higher in FGD gypsum. Although mined gypsum data were not available for any of the halogens, there is also a high degree of confidence these constituents will be higher in FGD gypsum based on the tendency for halogens to concentrate in the vapor phase. There is uncertainty whether coal combustion results in higher levels of arsenic, beryllium, cadmium, chromium or thallium in FGD gypsum, but these constituents were retained for further evaluation out of an abundance of caution.

4.3. Comparison of Releases to Water

EPA reviewed all the data available in the constituent database to determine how best to compare the exposures that may result from releases through leaching. The exposure routes from the conceptual model considered in the comparison were ingestion of impacted ground water and fish caught from nearby surface water bodies. Due to the trace levels of most constituents in leachate, it is assumed that releases from both FGD and mined gypsums will behave the same once mixed with environmental media, where ambient conditions will dictate fate and transport. Thus, EPA believes that a comparison of leachate concentrations provides a suitable surrogate for exposures that may result from releases to ground or surface water.

4.3.1. Comparison of Washed and Unwashed FGD Gypsum

Based on the comparison discussed in **Section 4.2.1** for bulk content, EPA determined that the seven sample pairs would also provide the best comparison of washed and unwashed leaching behavior. These leachate samples were analyzed with EPA Method 1313 and provide data over the entire pH range of interest. The available data were pooled into a single distribution because a similar amount of data are available for each sample across the pH range. Because of the relatively small number of source samples (n = 7), EPA used the WSR test for this comparison. The results of this comparison are presented in **Table 4-4**, with instances of higher unwashed concentrations highlighted. Because some p-values are extremely small, the reported values are truncated for values below 0.001 for ease of presentation.

Table 4-4. Comparison of Washed/Unwashed FGD Gypsum Median Leachate Concentrations

Constituent	Unwashed		Washed		WSR p-value	Result
	Detection Frequency	Median (µg/L)	Detection Frequency	Median (µg/L)		
Group 1						
Aluminum	24 / 24	473	21 / 22	546	0.921	Comparable
Barium	**24 / 24**	**98**	**23 / 23**	**80**	**0.015**	**Unwashed Higher**
Boron	**24 / 24**	**5,296**	**19 / 23**	**264**	**< 0.001**	**Unwashed Higher**
Manganese	**23 / 23**	**1,413**	**21 / 23**	**274**	**0.001**	**Unwashed Higher**
Nickel	**24 / 24**	**125**	**23 / 23**	**46**	**0.001**	**Unwashed Higher**
Selenium	**24 / 24**	**171**	**23 / 23**	**42**	**0.004**	**Unwashed Higher**
Strontium	24 / 24	775	23 / 23	547	0.307	Comparable
Vanadium	20 / 24	24	19 / 23	9.0	0.110	Comparable
Zinc	24 / 24	212	23 / 23	188	0.282	Comparable
Group 2						
Chromium	18 / 23	10	17 / 23	15	0.173	Comparable
Copper	13 / 24	7	14 / 23	10	0.617	Comparable
Iron*	17 / 24	343	12 / 23	44	0.124	Comparable
Molybdenum	**16 / 24**	**13**	**13 / 23**	**8**	**0.036**	**Unwashed Higher**
Group 3						
Antimony	3 / 24	5.6 (ND)	0 / 23	5.6 (ND)	--	--
Arsenic	7 / 24	6.4 (ND)	4 / 24	6.4 (ND)	--	--
Beryllium	0 / 24	6.4 (ND)	0 / 23	6.4 (ND)	--	--
Cadmium	13 / 23	3.0	8 / 23	1.7	--	--
Chloride	15 / 17	237,596	5 / 17	4,130 (ND)	--	--
Cobalt	17 / 24	10.0	7 / 24	4.1 (ND)	--	--
Lead	5 / 24	2.3 (ND)	0 / 23	2.3 (ND)	--	--
Mercury	7 / 24	0.0036 (ND)	10 / 23	0.0036 (ND)	--	--
Thallium	15 / 24	7.5	0 / 23	5.1 (ND)	--	--

(ND): The reported value is the detection limit.

* Though the median unwashed concentration is much higher, the distribution for washed gypsum has a tail with the highest concentrations.

The results of this comparison indicate that barium, boron, manganese, molybdenum, nickel and selenium may be released from unwashed gypsum at higher rates than washed gypsum. Statistical comparisons were not conducted for antimony, arsenic, beryllium, cadmium, chlorine, cobalt, lead, mercury or thallium because of the high proportion of non-detect samples in these datasets. However, it is notable that cadmium, chloride, cobalt and thallium all fall into Group 3 because of a decrease in the detection frequency after washing. This is a strong indication that substantial differences also exist for these constituents.

As a secondary line of evidence, EPA conducted a comparison of the 90th percentile leachate concentrations from all samples. These values were chosen because the extreme values are the most likely to shift as a result of changes in leaching behavior and most likely to reflect detected values for Group 3 constituents. To ensure a direct comparison, the percentiles were calculated from the raw data with non-detects set to half the detection limit interpolated between 0.2 pH intervals for a total of 16 comparison points. The percent differences were then averaged across a pH range of 5 to 8, except where both datasets were entirely non-detects. In these cases, the resulting 0% difference was excluded to avoid skewing the calculated percentages. If the average percent difference was greater than 44%, then washed and unwashed samples were judged to be substantially different and not the result of measurement uncertainty. This cutoff represents the maximum difference expected to occur between multiple measurements of a single sample based on the repeatability of EPA Method 1313 observed during inter-laboratory validation (U.S. EPA, 2012a). The results of this comparison are presented in **Table 4-5**, with values rounded to the nearest whole percent. Negative values reflect an average concentration increase measured for washed samples.

Table 4-5. Percent Difference Between Washed/Unwashed FGD Gypsum 90th Percentile Leachate Concentrations

Constituent	90th Percentile Detection Frequency		Percent Difference	Result
	Unwashed	Washed		
Aluminum	16 / 16	16 / 16	- 42%	Comparable
Antimony	**9 / 16**	**3 / 16**	**57%**	**Unwashed Higher**
Arsenic	16 / 16	16 / 16	21%	Comparable
Barium	16 / 16	16 / 16	13%	Comparable
Beryllium	0 / 16	0 / 16	--	--
Boron	**16 / 16**	**16 / 16**	**97%**	**Unwashed Higher**
Cadmium	**16 / 16**	**10 / 16**	**66%**	**Unwashed Higher**
Chloride	**16 / 16**	**16 / 16**	**95%**	**Unwashed Higher**
Chromium	16 / 16	16 / 16	- 18%	Comparable
Cobalt	**16 / 16**	**8 / 16**	**81%**	**Unwashed Higher**
Copper	16 / 16	16 / 16	- 24%	Comparable
Iron	16 / 16	16 / 16	- 24%	Comparable
Lead	**16 / 16**	**0 / 16**	**79%**	**Unwashed Higher**
Manganese	**16 / 16**	**16 / 16**	**92%**	**Unwashed Higher**
Mercury	16 / 16	13 / 16	- 2%	Comparable
Molybdenum	**16 / 16**	**16 / 16**	**54%**	**Unwashed Higher**
Nickel	**16 / 16**	**16 / 16**	**57%**	**Unwashed Higher**
Selenium	**16 / 16**	**16 / 16**	**58%**	**Unwashed Higher**
Strontium	16 / 16	16 / 16	- 8%	Comparable
Thallium	**16 / 16**	**0 / 16**	**73%**	**Unwashed Higher**
Vanadium	16 / 16	16 / 16	1%	Comparable
Zinc	**16 / 16**	**16 / 16**	**47%**	**Unwashed Higher**

* Value not presented because all data were non-detects.

The results of this comparison indicate that antimony, boron, cadmium, chloride, cobalt, lead, manganese, molybdenum, nickel, selenium, thallium and zinc can be released from unwashed gypsum at higher rates than washed gypsum. These results agree well with those presented in **Table 4-4** and are consistent with the decrease in detection frequency observed for constituents in Group 3 after washing. Based on these lines of evidence, washed and unwashed leachate data for these 11 constituents were kept separate in all subsequent analyses.

The results for barium disagreed between the two comparisons. The results presented in **Table 4-4** indicate that differences between washed and unwashed barium are statistically significant; however, the results in **Table 4-5** indicate that the magnitude of these differences are well within the range of measurement variability. Because the observed differences for barium are so small in magnitude, EPA concluded that any differences that do exist will not substantively change the results of the evaluation. Thus, EPA combined washed and unwashed data for this constituent.

The results for zinc also disagreed between the two comparisons. The results presented in **Table 4-5** indicate that the magnitude of differences for zinc were somewhat higher than the typical range of measurement variability; however, the results in **Table 4-4** indicate that these differences are not statistically significant. This indicates that there is high variability in both the washed and unwashed samples across the pH range, but no consistent shift in leachate concentration. Because the observed differences for zinc were not significant, EPA combined washed and unwashed data for this constituent.

For all the remaining constituents, there is no evidence that current washing practices substantially alter leaching behavior. Although a comparison could not be conducted for beryllium, all the LEAF data available for this comparison are non-detects. Therefore, EPA concluded that it was most appropriate to combine all available leachate data into a single dataset for the remaining 12 constituents.

4.3.2. Comparison of Mined and FGD Gypsum

After reviewing the leachate data for FGD and mined gypsum, EPA determined that insufficient information was available to conduct a reliable comparison of these materials. Method 1313 data are available for only two mined gypsum samples, resulting in a total of eight data points over the relevant pH range. Due to the small number of samples, no statements can be made about the representativeness of these data. EPA considered merging the Method 1313 data with Synthetic Precipitation Leaching Procedure (SPLP) and Toxicity Characteristic Leaching Procedure (TCLP) data into a single distribution for this comparison, but ultimately determined that this would increase the overall uncertainty. Because SPLP and TCLP measure the concentration released at a single pH, no information is provided about broader leaching behavior. As a result, it is not possible to determine whether the measured concentrations are limited by solubility or available content. In addition, the pH values of the available single-pH data tend to be clustered at or above a pH of 7. Combining the leachate data from these different methods may bias the overall distribution and lead to erroneous conclusions about whether the two materials are comparable. Because of these uncertainties, EPA chose to retain all constituents with FGD gypsum data for further evaluation.

4.4. Comparison of Releases to Air

EPA reviewed all the data available in the constituent database to determine how best to compare the exposures that could result from volatilization of mercury from the gypsum. The exposure routes for mercury vapor from the conceptual model considered in the comparison were inhalation of ambient air, contact with surface water and ingestion of fish caught from the water bodies. Due to the trace levels of mercury in the air, it is assumed that releases from both FGD and mined gypsums will behave the same once mixed with environmental media where ambient conditions will dictate fate and transport.

EPA identified multiple studies that measured emission rates from FGD gypsum (Pekney et al. 2009; Gustin and Ladwig, 2010; Cheng et al. 2012; Briggs et al., 2014). However, there is strong evidence in the literature that mixing gypsum with agricultural soil will alter emission rates. Mercury present in the soil column is typically associated with organic content, forming complexes with sulfur-containing functional groups (e.g., thiol and disulfide) (Meili, 1991; Yin et al., 1997; Xia et al., 1999; Skyllberg et al., 2006; Oswald et al., 2014). It has been shown that mercury applied to the soil will form these complexes within days or weeks of application (Hintleman et al., 2002; Harris et al., 2007). Given the high sulfur content of gypsum and organic content of agricultural soils, it is reasonable to expect that mercury in the applied gypsum will behave similarly to unamended soils soon after application. This conclusion is corroborated by studies that have shown that, after controlling for bulk mercury concentration, the emission rates from FGD gypsum mixed with soil are far closer to those from the unamended soil than those from the original FGD gypsum (Cheng et al., 2012; Briggs et al., 2014). Studies on the mercury emission rates from natural soils have found a strong and consistent relationship between the mercury content of soils and the resulting emission rates (Eckley et al., 2011; 2015; 2016). Thus, EPA determined that comparison of bulk mercury concentrations in the gypsum would provide a suitable surrogate for exposures that result from volatilization of mercury to ambient air.

4.4.1. Comparison of Washed and Unwashed FGD Gypsum

EPA determined that comparison of the bulk mercury concentration in washed and unwashed FGD gypsum was the most appropriate comparison for emission of mercury to ambient air. The comparison of bulk content discussed in **Section 4.2.1** previously demonstrated that mercury concentrations in washed and unwashed FGD gypsum are comparable. Therefore, EPA did not conduct further comparisons for these materials and combined available washed and unwashed emission data into a single dataset.

4.4.2. Comparison of Mined and FGD Gypsum

EPA determined that comparison of the bulk mercury concentration in mined and FGD gypsum was the most appropriate comparison for emission of mercury to ambient air. The comparison discussed in **Section 4.2.2** previously demonstrated that bulk content of mercury in FGD gypsum can be higher than in mined gypsum. These results agree with analyses in the Agency's 2014 *Coal Combustion Residual Beneficial Use Evaluation* that previously demonstrated the potential for higher mercury concentrations in wallboard made with FGD gypsum (U.S. EPA, 2014a). Therefore, EPA did not conduct further comparisons and retained mercury vapor for further evaluation.

4.5. Summary of Comparisons

EPA first compared washed and unwashed FGD gypsum to determine whether any differences exist between these materials that could have a substantial impact on the results and conclusions of this evaluation. When washed and unwashed gypsum were found to be substantially different, the data for these samples were kept separate for all future analyses. Otherwise, all available data were combined into a single dataset. For bulk content, the comparisons found that substantial differences exist for boron, chloride, and manganese. For leachate, the comparisons found that substantial differences exist for antimony, boron, cadmium, chloride, cobalt, lead, manganese, molybdenum, nickel, selenium and thallium. For volatile mercury emissions, no differences were identified.

Differences in leaching behavior of washed and unwashed samples were observed for constituents that are highly soluble over some or all of the relevant pH range, resulting in the quick depletion of leachable mass and a sharp drop in leachate concentration. Differences in the bulk content were observed for some, but not all, of these highly soluble constituents. There are two main reasons for this discrepancy. First, the solubility of constituents can vary by orders of magnitude across the pH range. While a given constituent may exhibit highly soluble behavior across part of the pH range, it might not at the prevailing pH during washing. Second, even when a constituent is highly soluble over the entire pH range, the constituent mass that is available to leach may be only a small fraction of the total mass present. The remainder may be so tightly adsorbed or complexed that it is unable to dissolve under the specified environmental conditions. Although it is likely that there are some differences for these remaining constituents between washed and unwashed samples, they are so minor that they cannot be reliably differentiated from noise. Therefore, EPA concluded that these differences would not provide a sound basis for recommendations about the appropriate use of FGD gypsum.

After separating washed and unwashed data for the relevant constituents, EPA compared mined and FGD gypsum to determine which constituents may be present in and released from FGD gypsum at higher levels. For bulk content, the comparison indicated that levels of arsenic, beryllium, boron (unwashed only), cadmium, chromium, mercury, selenium and thallium may be higher in FGD gypsum. For leachate, insufficient data for mined gypsum precluded a comparison, and so all constituents with sufficient data for FGD gypsum were retained. For volatile mercury emissions, the comparison indicated that releases of mercury can be higher from FGD gypsum. Because the potential exists for higher releases of these constituents, EPA retained each of them for a screening analysis.

The comparison of mined and FGD gypsum found many constituents to be present at comparable levels in these materials. This makes sense, given that the limestone used in FGD gypsum has been shown to account for a majority of the mass in FGD gypsum for some constituents. Both limestone and mined gypsum are excavated from the earth with minimal processing that would further concentrate inorganic constituents. The greatest differences between FGD and mined gypsum were identified for the constituents most likely to volatilize at the high temperature present during coal combustion (e.g., mercury, selenium). These are the constituents most likely to pass through particulate control devices and be captured in the limestone spray. **Table 4-6** provides a summary of the constituents carried forward for each impacted medium.

Table 4-6. Constituents Retained for Screening Analysis

Constituent	CASRN	Human Health				Ecological		
		Soil	Ground Water	Fish	Air	Soil	Surface Water	Sediment
Aluminum	**7429-90-5**	—	×	×	—	—	×	—
Antimony	**7440-36-0**	—	×	—	—	—	×	×
Arsenic	**7440-38-2**	×	×	×	—	×	×	×
Barium	**7440-39-3**	—	×	×	—	—	×	×
Beryllium	**7440-41-7**	×	×	×	—	×	×	—
Boron	**7440-42-8**	×	×	—	—	×	×	—
Cadmium	**7440-43-9**	×	×	×	—	×	×	×
Chloride	**16887-00-6**	—	—	—	—	—	×	—
Chromium	**7440-47-3**	×	×	×	—	×	×	×
Cobalt	**7440-48-4**	—	×	—	—	—	×	×
Copper	**7440-50-8**	—	×	—	—	—	×	×
Iron	**7439-89-6**	—	×	×	—	—	×	—
Lead	**7439-92-1**	—	×	—	—	—	×	×
Manganese	**7439-96-5**	—	×	×	—	—	×	×
Mercury	**7439-97-6**	×	×	×	×	×	×	×
Molybdenum	**7439-98-7**	—	×	×	—	—	×	—
Nickel	**7440-02-0**	—	×	×	—	—	×	×
Selenium	**7782-49-2**	×	×	×	—	×	×	×
Strontium	**7440-24-6**	—	×	×	—	—	×	—
Thallium	**7440-28-0**	×	×	×	—	×	×	—
Vanadium	**7440-62-2**	—	×	×	—	—	×	×
Zinc	**7440-66-6**	—	×	×	—	—	×	×

× - Retained for further evaluation

5. Screening Analysis

A screening analysis is a streamlined approach that reduces the complexity of the modeled system through a combination of high-end data and simplifying assumptions, which ensure that exposure estimates that may range anywhere from a reasonable upper bound to unrealistically extreme to ensure that risks are not underestimated. If a potential exposure is found to be below levels of concern based on this screening, it can be eliminated from further consideration with a high degree of confidence. The screening for FGD gypsum considered each exposure pathway carried forward from **Section 4 (Comparison with Analogous Product)**. This section provides a summary of the approach used to estimate exposures and the results of comparison with relevant benchmarks.

5.1. Data Preparation

All of the data used in this screening analysis were drawn from the gypsum database discussed in **Appendix A (Constituent Data)**. This subsection details additional steps taken to prepare the constituent data for the screening to ensure that the calculated exposures reflect anywhere from a high-end to an upper-bound estimate of what may result from FGD gypsum. Additional pathway-specific considerations are discussed in subsequent sections dedicated to each medium.

5.1.1. Non-Detect Data

Non-detect measurements in the dataset represent constituent concentrations below the level that an analytical methodology can differentiate from background noise. These measurements do not provide definitive evidence that a constituent is or is not present but do indicate that constituents are not present at concentrations any higher than the detection limit. Thus, eliminating non-detects outright may unduly bias the remaining, truncated data set toward the higher, detected values. Non-detect values were replaced with half of the reported detection limit according to the recommendations in *Risk Assessment Guidance for Superfund (RAGS) Part A* (U.S. EPA, 1989) and *EPA Region 3 Guidance on Handling Chemical Concentration Data near the Detection Limit in Risk Assessments* (U.S. EPA, 1991).

5.1.2. Available Content

Available content (also commonly referred to as “leachable content” or “soluble content”) is the total 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 L/S ratio of that sample (in L/kg). This is considered a reasonable estimate because the highly acidic pH will dissolve iron hydroxides and other complexations that may initially limit the release of constituents, and the high L/S ratio will ensure that all of the trace constituent mass can be dissolved (U.S. EPA, 2014e). The resulting available content (in mg/kg) should be less than or equal to the total constituent mass, but for some highly soluble constituents, the calculated available content may be slightly higher than the measured bulk content as a result of measurement uncertainty. In these instances, the available content was set to the measured total constituent mass. Sufficient data were not available to calculate the available content of beryllium, boron, copper, manganese, nickel, vanadium and zinc. Therefore, EPA made the assumption that the entire bulk content was leachable, which will tend to overestimate releases of these constituents.

Available content could not be calculated for samples without measured leachate concentrations over the full pH range (e.g., Methods 1311 and 1312). Therefore, to make the best use of all available data, EPA calculated the fraction of the total bulk content that is leachable for every Method 1313 sample by dividing the available content by the bulk content. For each constituent, the leachable fractions were assembled into a distribution that was applied to all samples to estimate the available content.

5.1.3. Available Content-Limited Behavior

When the solubility of a constituent in water is greater than the total mass available to be released from FGD gypsum, this can result in the rapid release of all the constituent mass present. Because the total mass that can be released in a given year is limited by the application rate, the dissolved concentration is strongly dependent on the amount of water present. Laboratory tests typically specify the ratio of water and solids. The L/S ratio used in a particular test can differ from what occurs in the field because of the amount of rainfall. Therefore, the measured concentration must be adjusted to ensure that releases are not underestimated (U.S. EPA, 2014e). For constituents found to exhibit leaching behavior limited by the available content, the measured leachate concentrations were adjusted based on **Equation 5-1**:

(5-1)
$$C_F = (C_M)\frac{(LS_M)(r)(n)}{(LS\ )(\rho)(d)}$$

Where:

C_F = Field leachate concentration [mg/m^3]
C_M = Measured leachate concentration [mg/m^3]
LS_M = Measured L/S ratio [Unitless; 10 for Method 1313, 20 for Methods 1311 and 1312]
LS = Saturated soil L/S ratio [Unitless; 0.5]
r = FGD gypsum application rate [$kg/m^2 \cdot yr$]
n = Years of application [yr]
ρ = Soil density [1,400 kg/m^3]
d = Soil mixing depth [0.2 m]

EPA determined that the constituents identified in **Section 4 (Comparison with Analogous Product)** with different washed and unwashed leaching behavior are those that exhibit behavior limited by available content (i.e., antimony, beryllium, boron, cadmium, chloride, cobalt, lead, manganese, molybdenum, nickel, selenium and thallium). This is because the loss of some fraction of the available content during washing limits the amount that can be released in subsequent leaching events. In contrast, leaching that is limited by the solubility of a constituent is anticipated to continue at approximately the same concentration until the available content is depleted. For this screening, EPA assumed that these 12 constituents exhibited available content-limited behavior in every sample over the pH range relevant to agricultural soils (i.e., 5 to 8).

Because this screening combined high-end values for bulk content, leachate concentration and leachable fraction that had been calculated independently from available data, a low L/S ratio might result in an unrealistic scenario where the available content of these 12 constituents does not deplete within a year. Therefore, EPA adjusted the leachate concentrations based on an assumed 100 years of application. While this will result in higher leachate concentrations than will actually occur in the field, it will not result in a dramatic overestimation. Any concentration higher than that needed to deplete available content will result in faster depletion and the same annualized concentration. It is possible that this adjustment could push dissolved concentrations above solubility limits; however, past studies have found that similar adjustments provided a reasonable estimate of field leaching (U.S. EPA, 2014e).

5.1.4. Depletion of Constituents

Due to the relatively low annual application rates identified for FGD gypsum, there is the potential that even constituents constrained by solubility limits will be depleted from the soil by runoff and infiltration prior to the next round of application. This can result in periods when no constituent mass remains to be released. All of the fate and transport models considered for this beneficial use evaluation require a leachate concentration that is constant throughout the year. Therefore, when constituents were found to deplete before the next round of application, an annualized leachate concentration was calculated. EPA first identified the minimum amount of water required to deplete all of the available constituent mass applied. If the amount of infiltration or runoff was greater than this minimum amount, the leachate concentration was multiplied by the ratio of the two values to estimate the average dissolved concentration over the course of the year. If the amount of infiltration or runoff was less than this minimum amount, then the measured leachate concentration was used without any additional adjustment.

5.1.5. Solubility Limits

In fresh surface waters, it is unlikely that dissolved constituent concentrations will exceed the solubility limits for the common solid phases of these elements. It is assumed that concentrations any higher than these solubility limits will precipitate out of solution as a solid. EPA calculated solubility limits for aluminum and iron because these are two major constituents found in FGD gypsum that are the most likely to exceed respective solubility limits. The geochemical speciation model MINTEQA2 was used to estimate the solubility limits for both constituents (U.S. EPA, 2001). This model assumes typical values for concentrations of dissolved ions and organic matter in freshwater bodies to identify the dominant solid phase for constituents and the corresponding solubility limits as a function of water pH. The highest limit for each constituent over the pH range was used as an upper bound on surface water concentrations in this screening.

The most soluble solid phases for each constituent that limits dissolved concentrations are aluminum hydroxide [$Al(OH)_3$] and iron carbonate [$FeCO_3$]. While iron(II) is likely to oxidize to the far less soluble iron(III) in oxygenated surface water, EPA considered the possibility of reducing conditions to ensure that the estimated upper bound did not underestimate potential water concentrations. **Table 5-1** presents the results of the MINTEQA2 modeling, with the upper bound of solubility limits highlighted. For both constituents, the highest solubility limit was identified between a pH of 5.5 to 6. These upper bounds were used as limits on the dissolved concentration in the surface water. Further discussion of this modeling effort can be found in **Appendix D (Screening Analysis)**.

Table 5-1. Aluminum and Iron Solubility in Surface Water

pH Range	Aluminum Solubility (µg/L)	Iron Solubility (µg/L)
5.5 to 6.0	**11**	**No Limit Found**
6.0 to 6.5	1	5,100,000
6.5 to 7.0	0.7	250,000
7.0 to 7.5	0.9	25,000
7.5 to 8.0	2	2,900
8.0 to 8.5	7	320

5.2. Screening Results

A single scenario was applied to all exposure pathways associated with each environmental medium. In this scenario, FGD gypsum is applied to an agricultural field at 3 tons/acre, which reflects one of the highest annual rates identified from the literature and summarized in **Appendix C (Use Characterization)**. The field covers the entirety of a 1,728,000-m^2 (427-acre) watershed that drains into an adjacent lake with a volume of 144,317 m^3 (117 acre-ft). These dimensions are based on a real-world watershed included in the Agency's Food Quality Protection Act Index Reservoir Screening Tool (FIRST), which was determined to be a high risk for surface water contamination. Applications occur over a 100-year timeframe. It is assumed the FGD gypsum is well distributed within the top 20 cm (8 in) of the soil column based on standard tilling depths (U.S. EPA, 2005). Although the same scenario was applied to each pathway, there was no attempt to account for mass balance between the pathways, to ensure that each pathway result would reflect the high-end estimate of potential exposures. Assumptions specific to a particular pathway are discussed in the relevant subsection. The calculated exposure concentrations were compared directly to the lowest of the relevant benchmarks identified for each receptor in **Appendix B (Benchmarks)**.

5.2.1. Soil Pathways

As a preliminary screen for soil pathways, constituents were applied at the 90th percentile bulk content and allowed to accumulate in the soil column without losses. Exposures to soil were assumed to occur within the field boundary. Although the resulting mixture of soil and FGD gypsum may be transported to downgradient soils by wind or overland runoff, the resulting downgradient concentrations will inevitably be lower than at the point of application.

To estimate the FGD gypsum that may be transported as a solid from the point of application to downgradient sediments through both wind and overland runoff, a100-year soil concentration was calculated assuming no loss of constituent mass after 100 years of mixing within the top 20 cm of soil and then divided by a single dilution and attenuation factor (DAF) of 10. This CCR-specific DAF was previously used in *Human and Ecological Risk Assessment of Coal Combustion Residuals* (U.S. EPA, 2014b) and represents unmitigated transport of CCRs from uncovered, above-grade landfills through wind and overland runoff. A DAF of 10, lower than the smallest value identified for any constituent, was applied to all constituents for ease of calculation and to ensure that sediment concentrations were not underestimated.

The calculated soil concentrations were compared to available benchmarks. For human receptors, these benchmarks addressed ingestion of soil, consumption of crops grown on the field, and consumption of beef and milk from cattle both fed on crops grown from the field and allowed to graze. For ecological receptors, these benchmarks addressed ingestion and direct contact with soil and sediment. **Table 5-2** presents the results of this preliminary round of screening**.**

Table 5-2. Preliminary Screening Results for Soil Pathways

Constituent	Wash Status	Human				Ecological	
		Soil	Produce*	Beef	Milk	Soil	Sediment
Arsenic	Combined	Screen Out	Screen Out	Screen Out	Screen Out	Screen Out	Screen Out
Beryllium	Combined	Screen Out	Screen Out	Screen Out	Screen Out	Screen Out	--
Boron	**Unwashed**	Screen Out	Screen Out	Screen Out	Screen Out	**Retain**	Screen Out
Cadmium	Combined	Screen Out	Screen Out	Screen Out	Screen Out	Screen Out	Screen Out
Chromium	**Combined**	Screen Out	Screen Out	Screen Out	Screen Out	**Retain**	Screen Out
Mercury	**Combined**	Screen Out	Screen Out	Screen Out	Screen Out	**Retain**	Screen Out
Selenium	**Combined**	Screen Out	Screen Out	Screen Out	Screen Out	**Retain**	Screen Out
Thallium	**Combined**	Screen Out	Screen Out	**Retain**	**Retain**	Screen Out	--

-- No screening benchmark identified for comparison.
* Same results for all individual crop categories.

The results of the preliminary screening identified potential concerns for human and ecological receptors. Concentrations of thallium were found to be above benchmarks for human receptors. Concentrations of boron (unwashed), chromium, mercury and selenium were found to be above benchmarks for ecological receptors. Therefore, these five constituents were retained for a second, refined round of screening.

This final round of screening used the same inputs as the preliminary round, but also accounted for moderate losses of constituent mass over the 100 years of application. EPA first calculated the soluble content by multiplying the 90th percentile bulk content by the 50th percentile available fraction to retain more constituent mass in the soil. Losses from leaching were estimated for each constituent at the 50th percentile leachate concentration. The infiltration depth was set to 5 cm/yr (2 in/yr) and the overland runoff depth was set to 17 cm/yr (6.7 in/yr) for a total annual water depth of 23 cm/yr (8.7 in/yr). The infiltration depth was drawn from the default low-end values modeled with the Hydrologic Evaluation of Landfill Performance (HELP) model for climate stations within the evaluation area (U.S. EPA, 1994). The annual overland runoff depth represents the annual amount of overland runoff from the 1,728,000-m^2 (427-acre) watershed required to maintain a constant water level in the reservoir. Any soluble mass that remained at the end of a year was summed with the insoluble mass to calculate accumulation. The calculated media concentrations were compared directly to the same screening benchmarks. The results of the final round of screening are presented in **Table 5-3.** For selenium and thallium, which were found to have comparable washed and unwashed bulk content, but different leaching behavior, the results are now presented separately.

Table 5-3. Final Screening Results for Soil Pathways.

Constituent	Wash Status	Human		Ecological
		Beef	Milk	Soil
Boron	Unwashed	--	--	Screen Out
Chromium	**Combined**	--	--	**Retain**
Mercury	**Combined**	--	--	**Retain**
Selenium	**Unwashed**	--	--	**Retain**
	Washed	--	--	**Retain**
Thallium	**Unwashed**	**Retain**	**Retain**	--
	Washed	**Retain**	**Retain**	--

-- Screened out in the preliminary screening

These results identified potential concerns associated with thallium (both washed and unwashed) for human receptors and with chromium, mercury and selenium (both washed and unwashed) for ecological receptors. Mercury and thallium in particular often had low available fractions that result in the majority of constituent mass remaining in the soil, regardless of the magnitude of leachate concentrations. In contrast, boron was modeled with high available content and frequently exhibits available content-controlled behavior. Thus, it is reasonable that this constituent would deplete from the soil column relatively quickly. Based on these results, chromium, mercury, selenium and thallium were carried forward for further evaluation in **Section 6 (Risk Modeling)**.

5.2.2. Ground Water Pathways

As a preliminary screen for ground water pathways, EPA assumed that receptors were exposed to leachate as it was released from the fields with no dilution or attenuation. Annualized leachate concentrations were calculated using the 90th percentiles for leachate concentration, leachable fraction, and bulk content. An infiltration rate of 5 cm/yr (2 in/yr) was selected as a floor based on a lower bound on values calculated with the HELP model for each soil type at climate stations within the evaluation area (U.S. EPA, 1994). If the leachable mass of a constituent was depleted by release of the 90th percentile leachate concentration into this amount of infiltration, then the leachate concentration was scaled to reflect the average leachate concentration over the year. Otherwise the 90th percentile leachate concentration was used directly for comparison. These leachate concentrations were compared to available benchmarks for ingestion, dermal contact, and inhalation of vapor. The results of this preliminary screen are presented in **Table 5-4**.

Table 5-4. Preliminary Screening Results for Ground Water Pathways

Constituent	Wash Status	Human Receptors		
		Ingestion	Dermal	Inhalation
Aluminum	Combined	Screen Out	--	--
Antimony	**Unwashed**	**Retain**	Screen Out	--
	Washed	Screen Out	Screen Out	--
Arsenic	**Combined**	**Retain**	Screen Out	--
Barium	Combined	Screen Out	--	--
Beryllium	Combined	Screen Out	Screen Out	--
Boron	Unwashed	Screen Out	--	--
	Washed	Screen Out	--	--
Cadmium	Unwashed	Screen Out	Screen Out	--
	Washed	Screen Out	Screen Out	--
Chloride	Unwashed	Screen Out	--	--
	Washed	Screen Out	--	--
Chromium	**Combined**	**Retain**	Screen Out	--
Cobalt	**Unwashed**	**Retain**	--	--
	Washed	Screen Out	--	--
Copper	Combined	Screen Out	Screen Out	--
Iron	Combined	Screen Out	--	--
Lead	Unwashed	Screen Out	--	--
	Washed	Screen Out	--	--
Manganese	**Unwashed**	**Retained**	Screen Out	--
	Washed	Screen Out	Screen Out	--
Mercury	Combined	Screen Out	Screen Out	Screen Out
Molybdenum	Unwashed	Screen Out	--	--
	Washed	Screen Out	--	--
Nickel	Unwashed	Screen Out	Screen Out	--
	Washed	Screen Out	Screen Out	--
Selenium	**Unwashed**	**Retain**	Screen Out	--
	Washed	**Retain**	Screen Out	--
Strontium	Combined	Screen Out	--	--
Thallium	**Unwashed**	**Retain**	Screen Out	--
	Washed	**Retain**	Screen Out	--
Vanadium	Combined	Screen Out	Screen Out	--
Zinc	Combined	Screen Out	Screen Out	--

-- No benchmark or complete exposure pathway identified for comparison.

The results of this first round of screening identified the potential for concern associated with the concentrations of antimony (unwashed), arsenic, chromium, cobalt (unwashed), manganese (unwashed), selenium (both washed and unwashed) and thallium (both washed and unwashed) for human receptors. These seven constituents were carried forward for a second, refined round of screening that accounted for some of the dilution and attenuation that may occur in the environment between the point of release (i.e., fields) and the point of exposures (i.e., private wells). To estimate well concentrations, EPA used the land application module in the Industrial Waste Evaluation Model (IWEM; U.S. EPA, 2015) with the following inputs:

- **Soil Type** is the type of subsurface materials in the unsaturated zone immediately below the field and in the saturated zone below the water table. Both soil types were set to "unknown" for this screen. For the unsaturated zone, this results in a probabilistic sampling of the different soil types associated with the selected geographic location. For the saturated zone, this selection provides values representative of the average aquifer characteristics across the United States.
- **Infiltration Rate** is the amount of precipitation that percolates into the field in a given year. The annualized constituent mass flux is greatest when the infiltration rate is equal to the rate required to deplete a given constituent. Thus, different infiltration rates were chosen for each constituent. However, if the calculated rate was less than the floor of 5 cm/yr, this floor was used instead. In addition, a ceiling was selected through trial and error to prevent flooding of the field based on the other inputs used in the model. If the calculated rate was greater than the ceiling of 29 cm/yr, this value was used instead.
- **Climate Stations** are facilities with instruments that measure local atmospheric conditions. These stations provide local precipitation data and determine the infiltration rate into soils surrounding the field. Of the 102 stations available in the HELP model, EPA selected Grand Junction, Colorado, as the climate station to limit infiltration outside the field boundary and minimize the amount of dilution that may occur in the water table.
- **Distance to Receptor** is the shortest straight-line distance to the closest drinking water well. The nearest well was set 75 m (250 ft) from the edge of the field. This distance was selected based on recommendations for minimum offset distances from large contamination sources (U.S. EPA, 2002a). This value was selected to reflect a large and continuous source.

IWEM outputs a single concentration for each constituent that represents the 90th percentile from 10,000 model runs. These concentrations were compared to benchmarks for the ingestion of ground water. The results of the final round of screening are presented in **Table 5-5**.

Table 5-5. Final Screening Results for Ground Water Pathways

Constituent	Wash Status	Human Ingestion
Antimony	**Unwashed**	**Retain**
Arsenic	**Combined**	**Retain**
Chromium	**Combined**	**Retain**
Cobalt	Unwashed	Screen Out
Selenium	Unwashed	Screen Out
	Washed	Screen Out
Thallium	**Unwashed**	**Retain**
	Washed	**Retain**

This final round of screening identified potential concerns for human receptors. Concentrations of antimony (unwashed), arsenic, chromium and thallium (both washed and unwashed) were found above benchmarks for human consumption of ground water. Based on these results, these four constituents were retained for further evaluation in **Section 6 (Risk Modeling)**.

5.2.3. Surface Water Pathways

A single round of screening was conducted for surface water pathways because it would be difficult to ensure that results would still reflect a reasonable high-end scenario after altering the empirical conceptual model. In this scenario, EPA assumed that all the water in the reservoir originated from overland runoff, with a minimum value set to 17 cm/yr (6.7 in/yr) to represent the annual amount of overland runoff from the watershed needed to maintain a constant water level in the reservoir. Contributions from ground water were not considered because that would only reduce concentrations in the water body due to dilution in the subsurface. If the leachable mass of a constituent was depleted by release of the 90th percentile leachate concentration into this amount of infiltration, then the leachate concentration was scaled to reflect the average leachate concentration over the year. Otherwise the 90th percentile leachate concentration was used directly for comparison. The calculated water concentrations were compared to benchmarks for the ingestion of fish caught from the water body by human receptors and both direct contact with and ingestion of surface water and sediment by aquatic ecological receptors. The results of the screening are presented in **Table 5-6**.

Table 5-6. Final Screening Results for Surface Water Pathways

Constituent	Wash Status	Human	Ecological	
		Fish Ingestion	Surface Water	Sediment
Aluminum	Combined	Screen Out	Screen Out	--
Antimony	**Unwashed**	Screen Out	Screen Out	**Retain**
	Washed	Screen Out	Screen Out	**Retain**
Arsenic	**Combined**	**Retain**	Screen Out	Screen Out
Barium	Combined	Screen Out	Screen Out	Screen Out
Beryllium	Combined	Screen Out	Screen Out	--
Boron	Unwashed	Screen Out	Screen Out	--
	Washed	Screen Out	Screen Out	--
Cadmium	**Unwashed**	**Retain**	**Retain**	**Retain**
	Washed	Screen Out	Screen Out	Screen Out
Chloride	Unwashed	Screen Out	Screen Out	--
	Washed	Screen Out	Screen Out	--
Chromium	**Combined**	Screen Out	**Retain**	**Retain**
Cobalt	Unwashed	--	Screen Out	Screen Out
	Washed	--	Screen Out	Screen Out
Copper	Combined	--	Screen Out	Screen Out
Iron	**Combined**	Screen Out	**Retain**	--
Lead	**Unwashed**	--	**Retain**	**Retain**
	Washed	--	Screen Out	**Retain**
Manganese	**Unwashed**	Screen Out	**Retain**	**Retain**
	Washed	Screen Out	Screen Out	Screen Out
Mercury	**Combined**	**Retain**	Screen Out	**Retain**
Molybdenum	Unwashed	Screen Out	Screen Out	--
	Washed	Screen Out	Screen Out	--

Table 5-6. Final Screening Results for Surface Water Pathways

Constituent	Wash Status	Human	Ecological	
		Fish Ingestion	Surface Water	Sediment
Nickel	**Unwashed**	Screen Out	Screen Out	**Retain**
	Washed	Screen Out	Screen Out	**Retain**
Selenium	**Unwashed**	**Retain**	**Retain**	Screen Out
	Washed	**Retain**	**Retain**	Screen Out
Strontium	Combined	Screen Out	Screen Out	--
Thallium	**Unwashed**	**Retain**	Screen Out	--
	Washed	**Retain**	Screen Out	--
Vanadium	Combined	Screen Out	Screen Out	--
Zinc	**Combined**	Screen Out	Screen Out	**Retain**

-- Benchmark value not available

The surface water screening identified potential concerns to both human and ecological receptors. Concentrations of arsenic, cadmium (unwashed), mercury, selenium (both washed and unwashed) and thallium (both washed and unwashed) were above benchmarks for human consumption of fish. Concentrations of cadmium (unwashed), chromium, iron, lead (unwashed), manganese (unwashed), mercury and selenium (both washed and unwashed) were above benchmarks for ecological exposure to surface water. Concentrations of antimony (both washed and unwashed), cadmium (unwashed), chromium, lead (both washed and unwashed), mercury, nickel (both washed and unwashed) and zinc were above benchmarks for ecological exposure to sediment. Therefore, all of these constituents were retained for further evaluation in **Section 6 (Risk Modeling)**.

5.2.4. Air Pathways

As a screen for mercury in air (other constituents do not volatize under ambient conditions), EPA assumed that the entire constituent mass applied to the soil each year volatilizes prior to the next round of application. Calculated with the 90th percentile bulk content, this resulted in a continuous emission rate of 77 ng/m^2-hr. This emission rate was input into AERMOD to estimate air dispersion and deposition. Further information about the inputs to the model can be found in **Appendix D (Screening Analysis)**. The maximum ambient air concentration was compared directly to the benchmark for inhalation. The maximum deposition rate of vapor onto soil was used to calculate a dissolved mercury concentration based on equations outlined in *Human Health Risk Assessment Protocol for Hazardous Combustion Facilities* (U.S. EPA, 2005). The calculated surface water concentrations were compared to benchmarks for human ingestion of fish and ecological exposure to both surface water and sediment. **Table 5-7** presents the results of this final round of screening.

Table 5-7. Final Screening Results for Air Pathways (Mercury Only)

Constituent	Wash Status	Human		Ecological	
		Inhalation	Fish Ingestion	Surface Water	Sediment
Mercury	Combined	Screen Out	Screen Out	Screen Out	Screen Out

These results indicate that potential exposures from volatilization of mercury fall over an order of magnitude below levels of concern for all exposure routes. Because no concerns were identified in this scenario even with these high-end assumptions, EPA did not further refine the emission estimates. Therefore, this pathway was not carried forward for further evaluation.

5.3. Summary

EPA applied the constituent concentration data assembled in **Appendix A (Constituent Data)** to provide a point estimate of exposures that falls somewhere between the high-end and worst-case of possible exposures to each media. The concentrations modeled in environmental media were compared directly to benchmarks identified in **Appendix B (Benchmarks)**, which were developed to protect human and the environment. Where higher concentrations than these benchmarks were identified, the screening scenario was refined to the extent possible to reflect fate and transport that will occur in the environment. Constituents still above relevant benchmarks were retained for further evaluation in **Section 6 (Risk Modeling)**. The screening results for each medium are summarized in **Table 5-8**.

Table 5-8. Constituents Retained for Risk Modeling

Constituent	CASRN	Human Health				Ecological		
		Soil	Ground Water	Surface Water	Air	Soil	Surface Water	Sediment
Aluminum	7429-90-5	--	--	--	--	--	--	--
Antimony	**7440-36-0**	--	×	--	--	--	--	×
Arsenic	**7440-38-2**	--	×	×	--	--	--	--
Barium	7440-39-3	--	--	--	--	--	--	--
Beryllium	7440-41-7	--	--	--	--	--	--	--
Boron	7440-42-8	--	--	--	--	--	--	--
Cadmium	**7440-43-9**	--	--	×	--	--	×	×
Chloride	16887-00-6	--	--	--	--	--	--	--
Chromium	**7440-47-3**	--	×	--	--	×	×	×
Cobalt	7440-48-4	--	--	--	--	--	--	--
Copper	7440-50-8	--	--	--	--	--	--	--
Iron	**7439-89-6**	--	--	--	--	--	×	--
Lead	**7439-92-1**	--	--	--	--	--	×	×
Manganese	**7439-96-5**	--	--	--	--	--	×	×
Mercury	**7439-97-6**	--	--	×	--	×	--	×
Molybdenum	7439-98-7	--	--	--	--	--	--	--
Nickel	**7440-02-0**	--	--	--	--	--	--	×
Selenium	**7782-49-2**	--	--	×	--	×	×	--
Strontium	7440-24-6	--	--	—	--	--	--	--
Thallium	**7440-28-0**	×	×	×	--	--	--	--
Vanadium	7440-62-2	--	--	--	--	--	--	--
Zinc	**7440-66-6**	--	--	--	--	--	--	×

× - Retained for further evaluation.

6. Risk Modeling

This step of the beneficial use evaluation consists of a national-scale evaluation designed to capture the variability in constituent properties, environmental conditions and receptor characteristics that may impact the fate and transport of constituents released from FGD gypsum during use. The modeled results are intended to provide a best estimate of the long-term (i.e., chronic) risks that may result from use of FGD gypsum in agriculture. The full-scale model considered each exposure pathway and associated constituents carried forward from **Section 5 (Screening Analysis)**. The remainder of this section describes the handling of the available data, the model design and the model results.

6.1. Model Inputs

Where data availability permitted, EPA compiled data for each model input into distributions that could be probabilistically sampled. Multiple distributions were created for some model inputs based on location to better capture any relevant geographic correlations (e.g., soil type, field size, precipitation rate). The spatial resolution at which the data were aggregated were selected to best capture the variability of data, while also minimizing the computational intensity necessary for the probabilistic model results to converge (i.e., independent runs of the model will return the equivalent results). Local-scale distributions were compiled at either the 10 or 12-digit hydrologic unit codes (HUC10, 12).[2] Regional-scale distributions were compiled at different scales wider than a HUC10 (e.g., HUC8, state-wide). National-scale distributions were compiled for the entire country. Further details about data collection and preparation are discussed in the relevant appendices discussed.

6.1.1. Constituent Data

A detailed discussion of the efforts to identify, review and assemble the constituent data used in this beneficial use evaluation is provided in **Appendix A (Constituent Data)**. This subsection details the data management to define empirical distributions and prepare the data for use in fate and transport models. A summary of the model inputs is presented in **Table 6-1**. The majority of the data available in the literature were blinded, meaning the reported constituent concentrations could not be linked to a specific utility. As a result, it was not possible for EPA to link constituent concentrations present in and released from FGD gypsum with specific geographic areas where it might be applied. Instead, EPA aggregated all of the available data into national-scale distributions and applied the same distributions to all agricultural fields.

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2) HUCs map the areal extent of surface water drainage across the United States with a hierarchical system of nested hydrologic units at different spatial scales that range from region (HUC2) to sub-watershed (HUC12). The size of the drainage area is indicated by the number of digits, with larger numbers representing smaller areas.

Table 6-1. Summary of Constituent Data in the Probabilistic Analysis

Type	Spatial Resolution	Variability	Appendix
Bulk Content	National : Country-Wide	Distribution	Appendix A
Available Content	National : Country-Wide	Distribution	
pH-Dependent Leachate Concentration	National : Country-Wide	Distribution	

Non-Detect Data

Non-detect measurements in the dataset represent constituent concentrations below the level that an analytical methodology can differentiate from background noise and do not provide definitive evidence that a constituent is or is not present. However, non-detect measurements can provide useful information because it is known that the constituent is not present at concentrations any higher than the detection limit. Eliminating non-detect values entirely may unduly bias the remaining, truncated data set toward the higher, detected values. Instead, non-detect values were replaced with half of the reported detection limit according to the recommendations in *Risk Assessment Guidance for Superfund (RAGS) Part A* (U.S. EPA, 1989) and *EPA Region 3 Guidance on Handling Chemical Concentration Data near the Detection Limit in Risk Assessments* (U.S. EPA, 1991).

Washed/Unwashed Data

When constituent concentrations present in or released from washed and unwashed samples were found to be comparable (See: **Section 4: Comparison with Analogous Product**), all the available data were combined into a single empirical distribution regardless of wash status. If data for both washed and unwashed versions of the same sample were available, concentrations were averaged to avoid bias towards a particular sample source. This was not possible for leachate samples when the final pH of washed and unwashed samples were different. These samples were determined to not be duplicative because they capture the leaching behavior of the sample at different portions of the relevant pH range.

Leachate pH

The available leachate data are compiled from tests at a single pH (i.e., TCLP, SPLP) and at multiple pH (i.e., LEAF Method 1313). While it is possible to interpolate among the data from Method 1313 to estimate leachate concentrations at any given pH within the relevant range, interpolation is not possible for single pH tests. A consequence of interpolation is that probabilistically sampling distributions of leachate data based on a specific pH assigned to each model run would heavily bias against the selection of single pH data. Instead, EPA divided the available leachate data into 6 bins of 0.5 pH increments. If a value fell on the cusp of two bins, it was placed in both. EPA incorporated the Method 1313 data into these bins after interpolation at 0.25 pH increments to ensure even coverage of the pH range while not overwhelming single-point data. This coverage is important because dramatic shifts in leachate concentration can occur over a small pH range. Although

Method 1313 data in the distribution will be sampled more frequently, this reflects the better coverage of sample variability within each pH bin. The specific pH assigned to each model run was then used to select the pH bin sampled.

Data Sampling

EPA designed a framework to sequentially pull data for each model run. This framework is meant to make the best use of all available data while maintaining real-world connections that already exist within the dataset. First, the wash status for the particular model run was assigned. Both washed and unwashed FGD gypsum were modeled with the same frequency. The bulk content of each constituent in FGD gypsum was then sampled from the relevant empirical distributions. Next, leachate concentrations and available fraction were sampled. If the FGD gypsum associated with the bulk content also had data on the available fraction and/or leachate concentration, these data were assigned by default for every model run associated with that sample. If data for one or both of these variables were not available for the selected FGD gypsum sample, values were sampled probabilistically from a distribution of all available data.

Available Content

To determine whether a constituent exhibits leaching behavior limited by the available content over a specific range of pH values, EPA compared the leachate concentration measured at each interpolated pH point to the maximum pH concentration measured for that sample. If the two values fell within 44% (See **Section 5: Screening Analysis**), then the constituent is labeled as available content-limited. Otherwise, the constituent was labeled as solubility limited. Available content-limited leachate concentrations were adjusted based on **Equation 5-1** to reflect differences between the L/S ratios used in laboratory tests and those present in the field.

Because the full-scale model independently sampled values for bulk content, leachate and available fraction, a low L/S ratio might result in an unrealistic scenario where the available content does not deplete within the year. Therefore, to ensure that exposures were not underestimated, EPA adjusted the leachate concentrations based on 100 years of application. While this will result in higher leachate concentrations than will actually occur in the field, it will not result in a dramatic overestimation. Any concentration higher than that needed to deplete the leachable content will result in faster depletion and the same annual average concentration. It is possible that this adjustment could push the dissolved concentration above solubility limits; however, past studies have found that similar adjustments provided a reasonable estimate of field leaching (U.S. EPA, 2014e).

Annualized Leachate Concentration

It is assumed that FGD gypsum is applied only once in any given year. Due to the relatively low annual application rates identified for FGD gypsum, there is a potential that constituents will be depleted from the soil by runoff and infiltration prior to the next round of application, even for constituents with leaching behavior limited by solubility. This can result in parts of the year when no constituent mass remains to be released. The fate and transport models considered for this beneficial use evaluation require a fixed leachate concentration provided in annual time steps (U.S. EPA, 2003d,e). Therefore, when constituents were found to deplete before the next application, an annualized leachate concentration was calculated. To calculate this concentration, EPA first calculated the amount of rainfall that contributes to the ground and surface water pathways through infiltration and runoff in a single year. If the amount of water required to deplete a constituent was less than this amount, the leachate concentration was multiplied by the ratio of the two values to estimate the fraction of water that would be free of that constituent. If the ratio was greater than one, then the leachate concentration was used without any additional adjustment.

6.1.2. Exposure Factors

A detailed discussion of the data used to characterize the rate at which receptors are exposed to environmental media and the resulting likelihood of adverse health effects is provided in **Appendix B (Benchmarks)**. These data include information about receptor physiology, mobility, dietary habits, and susceptibility. A summary of the model inputs for exposure and toxicity is presented in **Table 6-2**. The available data were often based on national surveys and it was not possible for EPA to link receptor characteristics to specific geographic areas. Thus, EPA aggregated all of the available data into national-scale distributions. In instances where a full distribution could not be developed, constant values intended to capture reasonable high-end exposures were used instead. This approach makes the best use of available data and ensures that the potential exposures are not underestimated.

Table 6-2. Summary of Exposure and Toxicity Data in the Probabilistic Analysis

Type	Spatial Resolution	Variability	Appendix
Exposure Averaging Time	National : Country-Wide	Constant	Appendix B
Exposure Frequency	National : Country-Wide	Constant	
Fraction of Media Contaminated	National : Country-Wide	Constant	
Fraction of Fish Consumed from Tropic Levels	National : Country-Wide	Constant	
Bioconcentration and Biotransformation Factors	National : Country-Wide	Constant	
Ecological Benchmarks	National : Country-Wide	Constant	
Human Toxicity Values	National : Country-Wide	Constant	
Cattle Ingestion Rate of Soil and Crops	National : Country-Wide	Constant	
Human Ingestion Rate of Fish	National : Country-Wide	Constant	
Human Ingestion Rate of Water, Beef, and Milk	National : Country-Wide	Distribution	
Body Weight	National : Country-Wide	Distribution	
Exposure Duration	National : Country-Wide	Distribution	

6.1.3. Extent of FGD Gypsum Use

A detailed discussion of the data used to characterize the rate and extent to which FGD gypsum may be used for different purposes is provided in **Appendix C (Use Characterization)**. EPA first defined the maximum geographic area over which FGD gypsum might be applied based on economic feasibility. A maximum area was defined for each potential use utilizing the available data on farmer willingness to pay; the relative locations of coal-fired utilities that generate FGD gypsum and quarries that mine natural gypsum; and the costs associated with the production, transport, and application of both types of gypsum. EPA then further defined the boundaries of agricultural fields within the economic feasibility zone where FGD gypsum might provide specific benefits. This required an initial delineation of agricultural fields that was accomplished with a combination of satellite imagery and survey data collected from both USDA and EPA. Given that crop patterns change over time as fields are left fallow and later resewn, data from 2010 to 2015 were used to capture total acreage that may be farmed. EPA used available data on soil conditions and crop types in those fields to determine which are likely to benefit from application of FGD gypsum. A summary of the relevant model inputs for gypsum use is provided in **Table 6-3**.

Table 6-3. Summary of FGD Gypsum Use Data in the Probabilistic Analysis

Type	Spatial Resolution	Variability	Appendix
Total Field Area	Local : HUC12	Constant	Appendix C
Distance to Surface Water	Regional : HUC8	Distribution	
Distance to Drinking Water Wells	Regional : State-Wide	Distribution	
Percent Field Area with Gypsum Application	National : Country-Wide	Distribution	
Years of Application	National : Country-Wide	Distribution	

6.1.4. Environmental Data

A detailed discussion of the data that characterize the properties of environmental media that impact constituent fate and transport is provided in **Appendix E (Probabilistic Modeling)**. EPA used the locations of fields for each potential use to identify data on soil type, hydrogeologic environment and climate. A summary of relevant model inputs for environmental characteristics is provided in **Table 6-4**. Site-based data on soil type and distance to receptors were drawn based on prevalence within the boundaries of fields to capture local variability. When field data were not available, regional data were collected by assigning fields to the nearest reference point in national databases, such as the HELP model climate stations and U.S. Geological Survey (USGS) hydrological regions. When environmental parameters could not be linked based on location, EPA sampled from national-scale distributions.

Table 6-4. Summary of Environmental Data in the Probabilistic Analysis

Type	Spatial Resolution	Variability	Appendix
Water Body Geometry	Local : Headwater - Mainstem - HUC12	Constant	Appendix E
Annual Average Water Flow	Local : Headwater - Mainstem - HUC12	Constant	
Base Flow Index	Local : Headwater - Mainstem - HUC12	Constant	
Stream Annual Flow Mixing Volume	Local : Headwater - Mainstem - HUC12	Constant	
Climate Center	Local : HUC10	Constant	
Soil Composition	Local : HUC10	Distribution	
Soil pH	Local : HUC10	Distribution	
Hydrogeological Environment	Local : HUC10	Distribution	
Total Suspended Solids	Regional : HUC2	Distribution	
Bed Sediment Particle Concentration	National : Country-Wide	Constant	
Bed Sediment Porosity	National : Country-Wide	Constant	
Depth of Upper Benthic Layer	National : Country-Wide	Constant	
Saturated and Unsaturated Soil K_d Values	National : Country-Wide	Distribution	
Bed and Suspended Sediment K_d Values	National : Country-Wide	Distribution	

6.2. Model Design

For each model run, EPA first used the partitioning module of the land application unit (LAU) model to determine what fraction of annual precipitation infiltrates to ground water or runs off overland directly to surface water (U.S. EPA, 2003f). The calculated depth of precipitation was used together with the constituent data (e.g., bulk content, available content) and soil properties (e.g., pH) to calculate an annualized leachate concentration. The calculated concentrations were used in both ground and surface water models to conserve mass among pathways. The following subsections summarize how fate and transport was modeled for different environmental media that may be impacted by FGD gypsum. A more detailed discussion of how the data were derived is provided in **Appendix E (Probabilistic Modeling)**.

6.2.1. Soil Pathways

To estimate accumulation in surface soil, EPA performed a simple mass balance on the soil column. Annual additions were calculated based on the FGD gypsum application rate and bulk content assigned to each model run. Annual losses were calculated based on the combined rate of runoff and infiltration, the available fraction and the leachate concentration for each model run. Each year, the insoluble fraction of the bulk content accumulated in the soil without any losses. The mass loss from leachate was subtracted from the accumulated soluble mass. Any soluble mass remaining at the end of the year was added to the insoluble mass to calculate accumulation. In every model iteration, the soil concentration was recorded over a 200-year time interval starting at the first year of application. The recorded concentrations were averaged over the subset of the time interval relevant to the modeled receptor, centered on the year of highest concentration.

6.2.2. Ground Water Pathways

The EPA Composite Model for Leachate Migration with Transformation Products (EPACMTP; U.S. EPA, 2003c,d,e) was used to model fate and transport through the subsurface environment and estimate concentrations at a specified downgradient point (i.e., private well or water body). The source area for each model run was defined as the total area of cropland within the relevant area with FGD gypsum applied. **Figure 6-2** depicts an aerial view of the conceptual model.

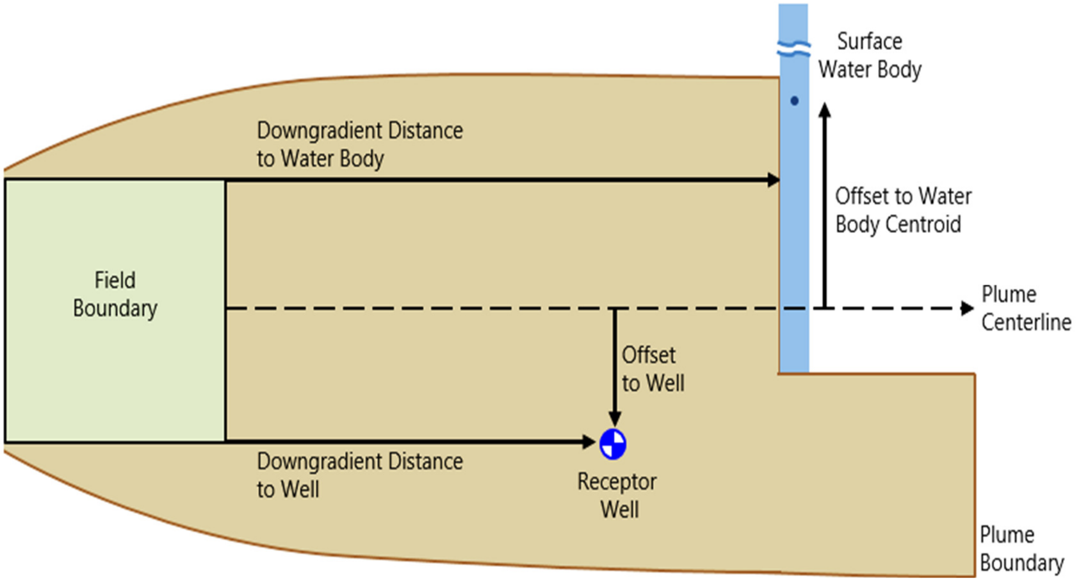

Figure 6-1: Aerial view of conceptual model for ground water plumes.

Ground water wells were treated as discrete points in the landscape, while surface water bodies were treated as straight lines with lengths set equal to the longest National Hydrography Dataset Plus (NHDPlus) flowline located in the modeled catchment or HUC12. The downgradient distance to nearest receptor from the field boundary was drawn from empirical distributions aggregated at the state level for ground water wells and at the HUC8 level for surface water bodies to minimize computational intensity while still capturing spatial variability. Once a distance was selected, a random number generator was used to offset the well or water body centroid randomly within the plume width, as estimated before each model run based on predicted dispersion coefficients. Any model runs that placed the well or water body entirely outside the modeled plume boundary were omitted. This ensured that the model results effectively captured highly exposed receptors.

For ground water, the well was assumed to be screened at a discrete point beneath the water table. In each model run, this point was allowed to vary to either a depth of 10 m below the water table or to the bottom of the aquifer, whichever was shallower. For surface water, the concentration along the width of the plume that intersected with the water body was used to calculate a mass flux from ground water to surface water. Because the focus of this evaluation is the risks related to application of FGD gypsum, the water discharging to the remainder of the water body length was assumed to not contribute additional constituent mass. For headwaters, the centroid of the water body was allowed to vary within the plume in each model run. For higher order streams, the water body intersected the entire plume width.

In each model iteration, the ground water concentration at the point of exposure was averaged over the selected exposure interval. For drinking water wells, this time interval was recorded around the peak concentration. For ground water discharge to surface water, concentrations were recorded at both the final year of FGD gypsum application (maximum of 100 years) and the peak ground water concentration at any point in the future. These two timeframes were chosen to reflect near-term exposures, which are assumed to occur concurrently with overland runoff, and far-term exposures, which may take several hundreds or thousands of years to occur. In cases where the ground water concentration was found to still be increasing after 10,000 years, EPACMTP stopped modeling and reported the ground water concentration at that time as the peak. This value was held constant over the relevant time interval.

6.2.3. Surface Water Pathways

EPA grouped water bodies into two sets for this evaluation based on the relationship of Strahler stream order and hydrologic unit code (HUC).[3] The first set consists of first- and second-order streams that are almost entirely contained within the boundaries of individual HUC12, referred to in this evaluation as "headwater streams." EPA used the concentrations modeled at each headwater outfall to estimate ecological exposure to surface water and sediment. The majority of land initially drains to these streams and so they collectively provide extensive habitat for wildlife. Although ecological receptors will also be present in higher-order streams, EPA focused on these smaller streams in part to manage computational intensity. The second set consisted of streams at or above third order that flow across HUC12 boundaries, referred to in this evaluation as "mainstem streams." EPA first modeled the cumulative mass loading to each HUC12 outfall until the stream order reached sixth order or greater. EPA then calculated the resulting constituent concentrations at each HUC10 outfall, which were used to estimate human exposure to fish. Although fish can be present in smaller streams, these water bodies are unlikely to support a population that could sustain fishing rates that correspond with the high-end ingestion rates that were considered in this evaluation.

The drainage area upgradient of each outfall is the total land area that contributes runoff through that discrete point in the landscape. For headwater streams, the drainage area was defined as the sum of all NHDPlus catchments between the point of origin and the outfall to a higher-order water bodies. For mainstem streams, the drainage area was defined as the total land area within the HUC12 boundary. The fraction of each drainage area covered in cropland was recorded and held

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3) Strahler stream order is used to define stream size based on a hierarchy of the tributaries. Initial streams without any upstream tributaries are first-order. Each time two streams of the same order intersect, the number increases.

constant.[4] This value was multiplied by the fraction of cropland with gypsum applied. Little data are available to estimate and so this fraction was allowed to vary in each run based on a flat distribution that ranged from none to all of the cropland. The concentration in runoff was calculated by multiplying the leachate concentration from the field with the fraction of the drainage area with FGD gypsum applied. This approach accounts for the mixing of runoff from the entire drainage area once it enters the water body. **Figure 6-3** presents an example of a HUC10 and all of the outfalls located within it.

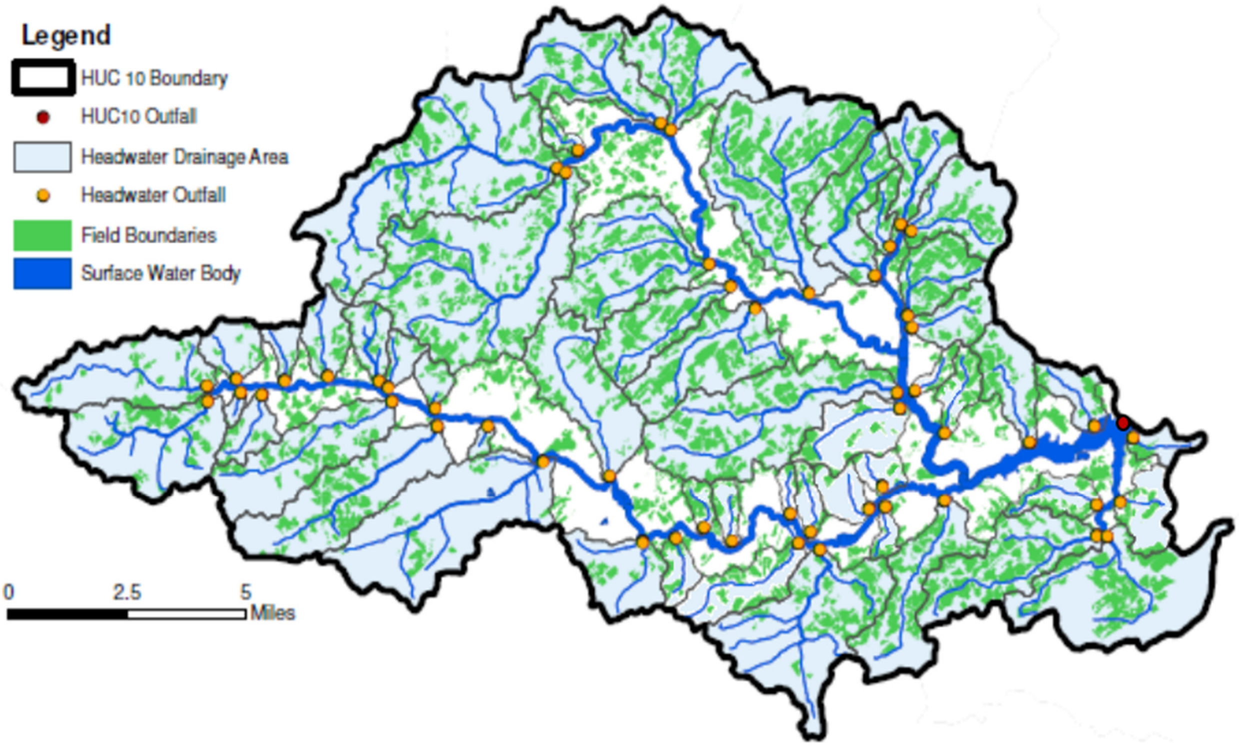

Figure 6-2: Map of drainage areas within a sample HUC10 boundary.

The contributions from overland runoff and ground water discharge to any outfall were related through the base flow index (BFI). This value reported in the NHDPlus dataset represents the cumulative fraction of water flow at a given point that originates from base flow (i.e., ground water discharge) compared to other sources (i.e., runoff). The modeled concentrations in ground water and overland runoff were weighted based on the BFI to approximate the contributions from each and obtain a concentration at each headwater or HUC12 outfall.

Larger water bodies flow through multiple HUC12s. Therefore, the flow rate at any given point may include drainage from one or more upgradient HUC12s. Because there can be a great deal of

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4) As described in **Appendix C (Use Characterization)**, the total cropland area represents the cumulative land area used to grow crops between the years 2010 and 2015. This total area will not be in active use in any given year, as some fraction will inevitably be left fallow. However, it represents a best estimate of the total area over which FGD gypsum may be applied over time.

variability among different HUC12s, it is not realistic to assign the entire concentration for these water bodies based on the characteristics of a single HUC12. Therefore, EPA mapped out the flow path of each stream through successive HUC12 outfalls and subtracted the annual average flow rate reported for each HUC12 outfall from those immediately downstream to obtain contributions from individual HUC12 to the overall flow. In each model run, the inputs for every HUC12 in the flow path were sampled independently from relevant distributions to calculate the constituent mass load to the local stream segment. The mass loading contributed by each successive HUC12 was summed along the flow path. At each HUC10 outfall, the cumulative mass loading and the annual flow rate were used to calculate the surface water concentration. Concentrations were calculated for each successive HUC10 until the stream either terminated (e.g., outfall to ocean) or intersected with a stream of sixth-order or more. This limit was selected both to limit the computational intensity from extremely long travel distances and the fact that the large drainage areas for these higher-order streams makes BFI measurements less reliable.

In each model iteration for both headwater and mainstem streams, the surface water concentration at the relevant outfall was recorded at the year of peak concentration. Concentrations were recorded at both the final year of FGD gypsum application (maximum of 100 years) and the peak ground water concentration at any point in the future. These two timeframes were chosen to reflect near-term exposures, which are assumed to occur concurrently with overland runoff, and far-term exposures, which may take several hundreds or thousands of years to occur. In cases where the ground water concentration was found to still be increasing after 10,000 years, EPACMTP stopped modeling and reported the ground water concentration at that time as the peak.

6.3. Model Results

The concentrations modeled in this beneficial use evaluation are intended to account for potential sources of variability associated with FGD gypsum, environmental media, and exposed receptors. In total, the fate and transport models were run one hundred times within every HUC10, resulting in up to two million individual model runs across the country for a single use. This subsection summarizes the model results for use to limit phosphorus runoff. This use was selected because it results in the highest annual mass loading to the environment. Because the model is not site-specific, the combination of model inputs inevitably results in some combinations that are outside of what will realistically occur in the field. As a result, chronic exposures above levels of concern indicate that further evaluation is required to determine if risks are driven by a particular subset of modeled scenarios. Therefore, constituents found above levels of concern were carried forward to **Section 7 (Uncertainty and Sensitivity Analyses)**.

The modeled concentrations in soil, ground water and surface water were used together with the long-term exposure and toxicity data discussed in **Appendix B (Benchmarks)** to calculate the likelihood that adverse health effects will occur. These effects can be divided into two broad types: carcinogenic and noncarcinogenic. Carcinogenic effects are those that ultimately result in the occurrence of cancer somewhere in the individual. The likelihood of carcinogenic effects is expressed as the increased lifetime probability of cancer that results from an incremental change in exposure. A risk of 1×10^{-5} was selected as the point at which further evaluation was warranted for human receptors. Noncarcinogenic effects are those that result in adverse health effects other than cancer. The likelihood of noncarcinogenic effects is expressed as a ratio of the exposure level and the level below which no adverse effects are known or anticipated to occur. This ratio is known as a hazard quotient (HQ). An $HQ > 1$ was selected as the point at which further evaluation was warranted for human and ecological receptors.

6.3.1. Soil Pathways

The full-scale model results for soil pathways are presented in **Table 6-5**. The reported results reflect the most sensitive receptors for noncarcinogens (i.e., children 1 to 5 years) and the most mobile and/or toxic valence states. None of the constituents carried forward to this stage of the evaluation for this exposure pathway had an identified carcinogenic endpoint. Values that exceed the risk criteria (i.e., $HQ > 1$) are shown in **bold**.

Table 6-5. National Risk Results for Soil Pathways

Constituent	Wash Status	Human				Ecological	
		Beef Ingestion		Milk Ingestion		Soil	
		90th	50th	90th	50th	90th	50th
Noncancer Hazard Quotient							
Chromium	**Combined**	—	—	—	—	**1.3**	0.19
Mercury	Combined	—	—	—	—	0.65	0.13
Selenium	Unwashed	—	—	—	—	0.88	0.03
	Washed	—	—	—	—	0.86	0.08
Thallium	**Unwashed**	**1.4**	0.03	**1.3**	0.03	—	—
	Washed	**1.7**	0.04	**1.5**	0.04	—	—

— Screened out in a previous step.

The model results show potential concerns associated with thallium (both washed and unwashed) for human receptors and chromium for ecological receptors. It is notable that slightly higher risks for thallium and selenium were sometimes identified for washed samples compared to unwashed samples. This occurred because it was not possible to differentiate between the bulk content of washed and unwashed FGD gypsum as a result of measurement uncertainty. However, differences were identified for washed and unwashed leachate of these two constituents. The use of a single distribution for bulk content and separate distributions for leachate resulted in slightly higher estimates of accumulation in the soil due to reduced leaching from washed samples. Because the differences identified between washed and unwashed soil results are small and primarily an artifact of data limitations, the two values can be considered effectively the same. Based on these results, EPA carried chromium and thallium forward for further evaluation in **Section 7 (Uncertainty and Sensitivity Analyses)**.

6.3.2. Ground Water Pathways

The full-scale model results for ground water pathways are presented in **Table 6-6**. The reported results reflect the most sensitive receptors for carcinogens (i.e., adults) and for noncarcinogens (i.e., children 1 to 5 years) and the most mobile and/or toxic valence states. Values that exceed the risk criteria (i.e., risk $> 1\times10^{-5}$ or HQ > 1) are shown in **bold**. Because some modeled values are extremely small, reported values are truncated below an HQ < 0.01 and risks $< 1\times10^{-7}$ for ease of presentation.

Table 6-6. National Results for Ground Water Pathways

Constituent	Wash Status	Human Drinking Water Ingestion	
		90th	50th
Cancer Risk			
Arsenic	Combined	1.1×10^{-6}	$< 1.0\times10^{-7}$
Chromium	Combined	$< 1.0\times10^{-7}$	$< 1.0\times10^{-7}$
Noncancer Hazard Quotient			
Antimony	Unwashed	< 0.01	< 0.01
Arsenic	Combined	0.03	< 0.01
Chromium	Combined	< 0.01	< 0.01
Thallium	Unwashed	0.05	< 0.01
	Washed	< 0.01	< 0.01

These results indicate that all risks from potential exposures to ground water fall well below levels of concern. Because no concerns were identified for this pathway, even at high-end exposures, EPA did not retain any constituents for further evaluation. Given that the use with the highest mass loading to ground water did not pose concern, EPA did not model the remaining uses.

6.3.3. Surface Water Pathways

The results of the full-scale modeling for surface water pathways are presented in **Table 6-7**. The reported results reflect the most sensitive receptors for carcinogens (i.e., adult recreational fishers > 21 years) and for noncarcinogens (i.e., children 1 to 5 years). The results also reflect the most mobile and/or toxic valence states for each constituent. Values that exceed the risk criteria (i.e., risk $> 1\times10^{-5}$ or HQ > 1) are shown in **bold**. Reported values reflect exposures from combined runoff and ground water discharge. All risks from peak ground water discharge only fall below levels of concern. Because some modeled values are extremely small, reported values are truncated below an HQ < 0.01 and risks $< 1\times10^{-7}$ for ease of presentation.

Table 6-7. National Risk Results for Surface Water Pathways

Constituent	Wash Status	Human		Ecological			
		Fish Ingestion		Surface Water		Sediment	
		90th	50th	90th	50th	90th	50th
Cancer Risk							
Arsenic	Combined	1.0×10^{-7}	$< 1.0\times10^{-7}$	—	—	—	—
Noncancer Hazard Quotient							
Antimony	Unwashed	—	—	—	—	< 0.01	< 0.01
	Washed	—	—	—	—	< 0.01	< 0.01
Arsenic	Combined	< 0.01	< 0.01	—	—	—	
Cadmium	Unwashed	< 0.01	< 0.01	0.03	< 0.01	0.1	< 0.01
Chromium	Combined	—	—	0.07	< 0.01	< 0.01	< 0.01
Iron	Combined	—	—	0.02	< 0.01	—	
Lead	Unwashed	—	—	0.02	< 0.01	0.01	< 0.01
	Washed	—	—	—	—	0.01	< 0.01
Manganese	Unwashed	—	—	0.2	0.01	0.03	< 0.01
Mercury	Combined	0.8	0.03	—	—	0.05	< 0.01
Nickel	Unwashed	—	—	—	—	0.01	< 0.01
	Washed	—	—	—	—	0.01	< 0.01
Selenium	**Unwashed**	**1.1**	0.04	**1.3**	0.09	—	—
	Washed	**1.0**	0.04	**1.3**	0.09	—	—
Thallium	Unwashed	0.2	< 0.01	—	—	—	—
	Washed	0.04	< 0.01	—	—	—	—
Zinc	Combined	—	—	—	—	0.2	< 0.01

— Previously screened out.

These results identified potential concerns to both human and ecological receptors. Concentrations of selenium (both washed and unwashed) were found to be at or above benchmarks for human consumption of fish and for ecological exposure to surface water. Therefore, EPA carried selenium forward for further evaluation in **Section 7 (Uncertainty and Sensitivity Analyses)**.

6.4. Summary

EPA refined the screening analysis discussed in **Section 5 (Screening Analysis)** to incorporate sources of variability and provide a best estimate of exposures that could result from use of FGD gypsum in agriculture at a national scale. The concentrations modeled in each environmental medium were used to probabilistically calculate risks to human and ecological receptors. Where risks were identified above levels of concern, constituents were retained for further evaluation in **Section 7 (Uncertainty and Sensitivity Analyses)**. The results of the full-scale model are summarized in **Table 6-8**.

Table 6-8. Constituents Retained for Uncertainty and Sensitivity Analyses

Constituent	CASRN	Human Health			Ecological		
		Soil	Ground Water	Fish Ingestion	Soil	Surface Water	Sediment
Antimony	7440-36-0	—	—	—	—	—	—
Arsenic	7440-38-2	—	—	—	—	—	—
Cadmium	7440-43-9	—	—	—	—	—	—
Chromium	**7440-47-3**	—	—	—	×	—	—
Iron	7439-89-6	—	—	—	—	—	—
Lead	7439-92-1	—	—	—	—	—	—
Manganese	7439-96-5	—	—	—	—	—	—
Mercury	7439-97-6	—	—	—	—	—	—
Nickel	7440-02-0	—	—	—	—	—	—
Selenium	**7782-49-2**	—	—	×	—	×	—
Thallium	**7440-28-0**	×	—	—	—	—	—
Zinc	7440-66-6	—	—	—	—	—	—

7. Uncertainty and Sensitivity Analyses

This step consists of a review of the uncertainties associated with this beneficial use evaluation and identification of any sensitive model inputs that might drive the identified risks. In any evaluation, there will always be some sources of uncertainty. Characterization of uncertainties associated with the data and modeling approach used in the evaluation can provide a better understanding of the potential impacts on the analytical results and conclusions. The identification of sensitive inputs can help define measures that may be targeted to effectively mitigate the identified risks. This discussion focuses primarily on constituents and exposure pathways previously found to pose potential risk in **Section 6 (Risk Modeling)**.

7.1. Uncertainty Analyses

Uncertainty exists to some degree in any evaluation, and it may bias model results higher or lower than actual values. It is important to understand both the direction and magnitude of uncertainties present in the evaluation. The direction of an uncertainty is the tendency for it to push a predicted value higher or lower than the true value, while the magnitude of an uncertainty is the extent to which it may push the predicted value away from the true value. There are three primary causes of uncertainty:

- Variability is the extent to which characteristics of environmental systems are heterogeneous. Uncertainty is introduced if the distributions used as inputs for the models do not fully capture the extent of real-world variability. 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 model input. This 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.

Potential sources of uncertainty were mitigated to the extent practicable prior to running the full-scale model. For example, uncertainties about the exact distribution of certain model inputs were addressed through point values or distributions intended to reasonably bound the true range while remaining protective. However, it is still useful to characterize the remaining uncertainties to understand whether and how analytical results might change if these uncertainties could be fully addressed in the model. The following text details the current understanding of the magnitude and direction of major uncertainties identified for this beneficial use evaluation, grouped by topic area.

7.1.1. Material Characterization

As part of this evaluation, EPA reviewed the available literature and assembled data on constituent concentrations present in or released from FGD and mined gypsum. **Appendix A (Constituent Data)** details the collection and review of this type of data. When individual data points or entire studies were found to introduce unacceptable levels of uncertainty into the evaluation, these data were removed prior to any quantitative analysis. The following text discusses the potential sources of uncertainty identified in the remaining data.

Additional Treatment:

Pelletization is a treatment that involves tumbling gypsum with a binding agent, resulting in more uniformly sized pellets. Known examples of binding agents include sodium lignosulfonate and black liquor (U.S. EPA, 2012b).[5] The advertised benefit of pelletized gypsum is that the material is more uniform in size, which produces less dust and is easier to both transport and spread. Pelletized mined gypsum is already available on the market (Chen et al, 2014; EPRI, 2008b; 2012; 2013; Kost et al., 2014; U.S. EPA, 2012b). EPA did not identify any references in the literature for pelletized FGD gypsum. However, this may be a result of a burgeoning market for this secondary material. There were not enough data available to determine whether the pelletization process contributes additional constituent mass to the mined gypsum or alters the leaching behavior of the gypsum. Therefore, EPA treated pelletized gypsum as a separate material from untreated gypsum that fell outside the scope of this evaluation. Further evaluation of pelletized gypsum may be warranted if the same treatment is applied to FGD gypsum.

Bulk Characterization Data:

A number of the studies relied upon to characterize FGD gypsum blinded the source of the samples. In some cases, this information was unknown even to the authors. As a result, despite attempts to reduce bias through data management, there remains the potential for some uneven weighting of the gypsum dataset toward certain regions of the country. Regardless, there is confidence that the full range of coal characteristics have been captured in the available dataset. Samples collected by EPA reflect a range of coal types, pollution control technologies, and wash status found across the United States (U.S EPA, 2008; 2009b). **Table 7-1** compares the data assembled by EPA with those from all other literature sources for the constituents that were collected by EPA and were found to have comparable washed and unwashed bulk content.

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[5] Black liquor is a secondary material generated by the kraft pulping process. This liquid contains a mixture of pulping residues (e.g., lignin, hemicellulose) and inorganic compounds (e.g., sodium hydroxide).

Table 7-1. Comparison of Data Collected by EPA and from Other Sources

Constituent	U.S. EPA (2008a; 2009b)				All Other Literature			
	Detection Frequency	50th Percentile	90th Percentile	Max	Detection Frequency	50th Percentile	90th Percentile	Max
Antimony	13 / 13	1.8	5.5	8.2	37 / 39	0.33	8.3	23.9
Arsenic	13 / 13	2.9	5.5	10	41 / 54	2.8	6.3	11
Barium	13 / 13	27.6	55.8	67	40 / 40	10	49.3	81.8
Cadmium	13 / 13	0.30	0.50	0.58	37 / 40	0.11	0.47	1.9
Chromium	13 / 13	7.7	13.4	14.9	42 / 45	3.6	7.4	15.0
Cobalt	13 / 13	2.5	3.4	4.3	27 / 40	0.25	0.66	3.1
Lead	13 / 13	2.4	3.8	6.5	32 / 40	1.0	2.0	8.3
Mercury	13 / 13	0.40	1.3	3.1	79 / 81	0.30	1.0	2.3
Molybdenum	13 / 13	3.7	6.3	12	38 / 41	0.7	2.5	6.2
Selenium	13 / 13	11.5	34.4	46	49 / 55	5.5	19.6	32
Strontium	13 / 13	177	383	530	37 / 37	154	338	405
Thallium	13 / 13	0.60	1.1	2.3	28 / 30	0.01	0.10	2.8

This comparison shows that the concentrations measured by EPA tend to be somewhat higher than the remaining dataset. However, there is considerable overlap in the range reported by both sets, with the major exceptions of cobalt and thallium. Although maximum concentrations are similar for both of these constituents, both the median and high-end concentrations are an order of magnitude different. It is unlikely that the differences result from analytical error in the EPA data, as Agency quality assurance and quality control (QA/QC) protocols were followed and this type of error would be expected to propagate to additional constituents. Instead, it is more likely that the samples collected by EPA reflect a wider swath of the FGD gypsum generated in the United States. EPA aimed to capture different coal types and pollution control technologies with these samples, while other studies focused on samples that are commercially available now. It is possible that higher concentrations in EPA samples reflect FGD gypsum that is not currently marketed for use, but that may be in the future. Based on these considerations, EPA concluded that it was appropriate to combine all the available data in the current evaluation. While some uncertainty remains about the exact shape of the distributions, the amount of overlap provides confidence that high-end concentrations have been adequately captured. The general agreement between EPA data and other sources also provides confidence that pH-dependent leachate data drawn from U.S. EPA (2008a; 2009b) also adequately capture high-end leaching behavior.

Constituents Without Characterization Data:

There are several constituents for which human or ecological benchmarks were identified, but for which sufficient bulk content or leachate data were not available to reliably characterize potential exposures. The full-scale results presented in **Section 6 (Risk Modeling)** indicate that constituents most likely to pose environmental concerns are those that volatilize in the flue gas and concentrate in FGD gypsum. The only other constituents that are known to be particularly volatile are members of the halogen group. Of these elements, one or more relevant benchmarks were identified for bromide, fluoride and iodide. Therefore, EPA focused on these constituents for further consideration. The following discussion relies on all available sources of information to fill data gaps and, as a result, includes a greater amount of uncertainty than the main evaluation. Therefore, the concentrations estimated in this section should not be used outside of the context discussed in this document:

- **Bromide:** EPA identified an ecological soil benchmark at 10 mg/kg (ORNL, 1997). Leachate data, but no usable bulk content data, were found for this constituent. EPA instead used the relationship between chlorine and bromide in coal to estimate a high-end concentration. The typical ratio of chlorine/bromide in coal is 0.02 (U.S. DOI, 2012). Multiplying the 90th percentile chlorine concentration in unwashed FGD gypsum by this ratio yields a bromide concentration of 52 mg/kg. Accumulation in the soil over 100 years under the screening scenario discussed in **Section 5 (Screening Analysis)** with no losses would result in a soil concentration of 13 mg/kg. However, this is not realistic because bromide is highly soluble. Accumulation with losses set at the 50th percentile unwashed leachate concentration of 160 ug/L results in depletion of the bromide added to the soil each year. Therefore, bromide is unlikely to drive environmental concerns for soil.
- **Fluoride:** EPA identified an ecological surface water benchmark of 2,700 µg/L (MIDEQ, 2007) and an MCL of 4,000 µg/L. Bulk content data, but no usable leachate data, were found for this constituent. The comparison of washed and unwashed samples revealed that concentrations of some washed samples were measured at higher levels than corresponding unwashed samples. This indicates that losses during washing are within the range of measurement uncertainty and so the two types of samples were combined. The 90th percentile concentration of combined samples is 1,350 mg/kg. Assuming complete washout of fluoride mass under the screening scenario discussed in **Section 5 (Screening Analysis)**, the resulting water concentration would be approximately 5,300 µg/L, which is almost twice the identified ecological benchmark. However, fluoride in FGD gypsum is typically associated with fluorite (CaF_2) and so releases are controlled by the solubility of this mineral (Álvarez-Ayuso and Querol, 2007). Thus, leaching of fluoride is not anticipated to sustain such high concentration. Furthermore, combined with the order-of-magnitude or more decrease in concentration observed for other constituents between screening and full-scale analyses, indicates fluoride is unlikely to drive environmental concerns for ground or surface waters.

  EPA also identified an ecological soil benchmark of 200 mg/kg (ORNL, 1997) and a human health screening value of 3,100 mg/kg for fluoride. Bulk content data, but no usable leachate data, were found for this constituent. The Agency identified one study from Spain that reported a single sample of FGD gypsum with a fluoride leachate concentration in deionized water mixed at a L/S ratio of 10:1 resulted in release of around 20% of the bulk constituent mass present (Álvarez-Ayuso et al., 2006), which supports the conclusion that fluoride will exhibit solubility-limited behavior. The leachate of other constituents in this sample generally fell at or below median values in the FGD database, making it unlikely these data will substantially overestimate potential losses. Accumulation in the soil over 100 years under the screening scenario discussed in **Section 5 (Screening Analysis)** with no losses results in a concentration of 325 mg/kg. Accumulation with leachate set to 20% of applied mass results in a peak concentration of 150 mg/kg, which is below the identified benchmark. Therefore, fluoride is unlikely to drive environmental concerns for soil.

- **Iodide:** EPA identified an ecological benchmark for iodide in soil of 4 mg/kg (ORNL, 1997). However, no usable bulk content or leachate data were found for this constituent and no other means to estimate values was identified. Therefore, no further evaluation for this constituent is possible.

Available data indicates that bromide and other halogens have low toxicity (WHO, 2009). Indeed, some are essential nutrients. However, it has been documented in laboratory and field studies that higher levels of halogens in surface water can increase formation of disinfection byproducts (DBPs) during water treatment (Luong et al., 1982; Heller-Grossman et al., 1993; Pourmoghaddas et al., 1993; Cowman and Singer 1996; Chang et al., 2001; U.S EPA, 2002b; Duong et al., 2003; Liang and Singer 2003; Ates et al., 2007; McTigue et al., 2014; Regli et al., 2015). Bromate (BrO^{-3}) can form when ozone reacts directly with bromide. Hypobromite (BrO^{-}) can form when chloride reacts with bromide, which can then react with organic matter to form a range of brominated and mixed chloro-bromo trihalomethanes. Although MCLs have been promulgated for total trihalomethanes and other DBPs, there are currently no surface water benchmarks for halogens that address subsequent formation of DBPs. There is also insufficient information available to reliably estimate the extent to which DBPs may form. As a result, this may result in an underestimation of potential risk, but the magnitude of this uncertainty is unknown. However, available leachate data demonstrate that washing can reduce releases of both chloride and bromide by an order of magnitude or more. Washing is sufficient to reduce leaching of chloride over an order of magnitude and bromide to below detection limits in the most samples. Therefore, if the formation of DBPs is a concern in a given area, then washing the FGD gypsum is an effective method to substantially reduce releases of halogens to the environment.

7.1.2. Farming Practices

EPA reviewed the available literature to assemble data on where and how gypsum might be applied across the country. **Appendix C (Use Characterization)** details the collection and review of this type of data. There was little information available to define how gypsum is currently used in many regions of the country. There is also the potential for practices to change over time as barriers are removed. Therefore, EPA aimed to define the maximum extent that FGD gypsum might be used without consideration of limits, such as regional availability of the material. This allowed EPA to evaluate each of the different uses, but likely overestimated the area over which FGD gypsum will actually be applied. The following text discusses the sources of uncertainty associated with where and how FGD gypsum is applied.

Extent of Use:

This beneficial use evaluation defined the maximum range that FGD gypsum might be used based on the location of coal-fired utilities that the Energy Information Administration (EIA) database reported as generating this secondary material (EIA, 2017). This range represents a snapshot in time and will be subject to change as older plants close and new plants open. There is no reliable way to forecast where a new plant will open and the type of pollution control technology it would install. However, it is possible to review the current landscape to determine if any existing facilities may retrofit and install forced oxidation scrubbers with limestone. EPA found that 293 of the 597 power plants operating in 2017 already produced FGD gypsum or have some other form of sulfur dioxide control. **Figure 7-1** provides a comparison of the relative locations of the current economic feasibility zone for FGD gypsum, coal-fired plants without sulfur dioxide controls, and non-attainment areas for sulfur dioxide to determine where new sulfur dioxide controls are most likely to occur in the future.

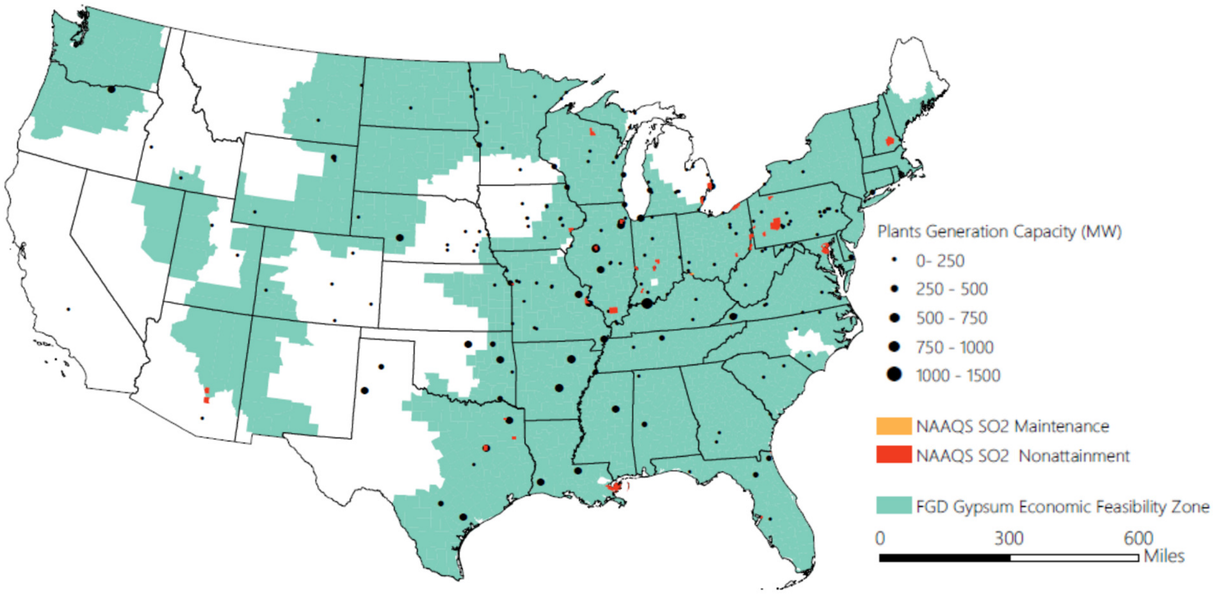


Figure 7-1: Locations of coal-fired plants without scrubbers and 2017 SO_2 NAAQS exceedances

Based on this map, the vast majority of non-attainment areas for sulfur dioxide are located within the current economic feasibility zone. The facilities located outside the modeled zone tend to have smaller generating capacities, which make them less likely to be a major future source of sulfur dioxide. Installation of a scrubber system onto one the facilities located near the non-attainment areas may extend the feasibility zone slightly. However, further expansion would be limited by the larger number of gypsum mines in surrounding areas. Based on this analysis, there is little concern that the extent that potential FGD gypsum use was underestimated. Instead, it is far more likely that the feasibility zone overestimates the area that gypsum will actually be used.

Extent of Application:

EPA estimated the total extent of agricultural land in the United States with data from satellite imagery and USDA surveys collected between 2010 and 2015. Multiple years of data were used to capture the periodic rotation of land use. EPA did not identify any reliable way to estimate how much of this total land will be utilized in any given year and so EPA assumed that application of FGD gypsum could range anywhere from 0 to 100% of the field area. In each model run, the extent of application was independently varied. This allowed consideration of variable mass loadings from different regions to the watershed outfalls, though some application still occurred in nearly every region in a given model run. This assumption ensures that the evaluation does not underestimate risks from releases that have the potential to compound from different parts of a watershed. EPA did not identify any data that could be used to further refine the model.

It is considered unlikely for a number of reasons that FGD gypsum will be applied every year at high-end rates over all the cropland in a watershed. First, application over the full area may simply not be needed. Beneficial use of FGD gypsum application are only needed for specific purposes that rarely extend to the full area. For example, the highest rate of FGD gypsum applications modeled was for the reduction of phosphorus in runoff, but this use would only be applied on individual fields where the potential for excessive phosphorus in runoff had been identified (i.e., high soil test phosphorus or use of manures as fertilizers). Also, in any given year, the actual extent of farmed land will be less than the total possible because some fields will be left fallow or transitioned to crops that do not require FGD amendments as a result of either agronomic or economic conditions. Second, it is possible that the benefits of application will extend beyond a single year. Finally, there may also physical limitations to how much FGD gypsum can be applied in a given area based on generation rates. The ACAA estimates that nearly 18 million tons of FGD gypsum were generated in 2020, but that nearly two-thirds of that were already diverted to wallboard production or other uses (ACAA, 2021). Even if all of the remaining FGD gypsum were directed to agricultural fields, that would only allow application of 3 tons/acre on less than 1% of the nearly 193 million acres of modeled cropland. Altogether, this is expected to result in an overestimation of potential risks. The magnitude of this uncertainty is generally expected to be large and to be even larger for higher-order streams evaluated for human exposure through fish ingestion. This is because the total land area that feeds into these streams is larger, sometimes spanning across multiple states. As the contributing land area increases, it becomes progressively less likely that a majority of the land area would have FGD gypsum applied at high-end rates in a given year.

Duration and Frequency of Application:

EPA selected 100 years as a reasonable upper bound on the duration of application. This value has been used in previous evaluations of agricultural amendments (U.S. EPA, 1992a,b). In each model run, the number of years was varied between 1 and 100 based on a flat distribution. It is unknown how much FGD gypsum will be generated or otherwise available for use that far into the future.

Even if the use of this secondary material is still common at that point, it is unlikely that it will be applied every year over that maximum duration. A combination of fallow periods needed for soil health and economic drivers that rotate crops or take fields out of production will periodically halt applications for one or more years at a time. It is also possible that application of FGD gypsum will not be as frequent in a given area because the benefits provided are sustained for more than a single year, further reducing need for annual application. Based on these considerations, the modeled duration and frequency of application is likely to result in an overestimation of potential risks. The magnitude of this uncertainty is expected to be large.

Tilling Practices:

Conservation tilling is a general term for a varied set of practices that minimize disturbance to the soil during farming. It is estimated that nearly half of all farms in the United States currently implement some form of conservation tilling (USDA, 2014a). Potential benefits include reduced soil erosion, reduced nutrient runoff, and increased retention of both moisture and organic matter in the soil (USDA, 2015b). However, changes in tillage system management often change over time depending on land ownership/management and crops being grown. Cropping systems that maintain "no-tillage" operations over many years are relatively rare. Most conservation tillage systems instead involve some limited tillage operations to prepare a seed bed for planting or to distribute fertilizers and other amendments into the soil profile. Also, some crops, such as peanuts and potatoes grow underground and require soil disturbance in order to harvest the crop. Based on consultation with USDA staff, EPA assumed that soil tillage intervals between 5 and 20 years would occur as part of normal farm production. Thus, long-term exposures were assumed to be better reflected by concentrations mixed in the soil column.

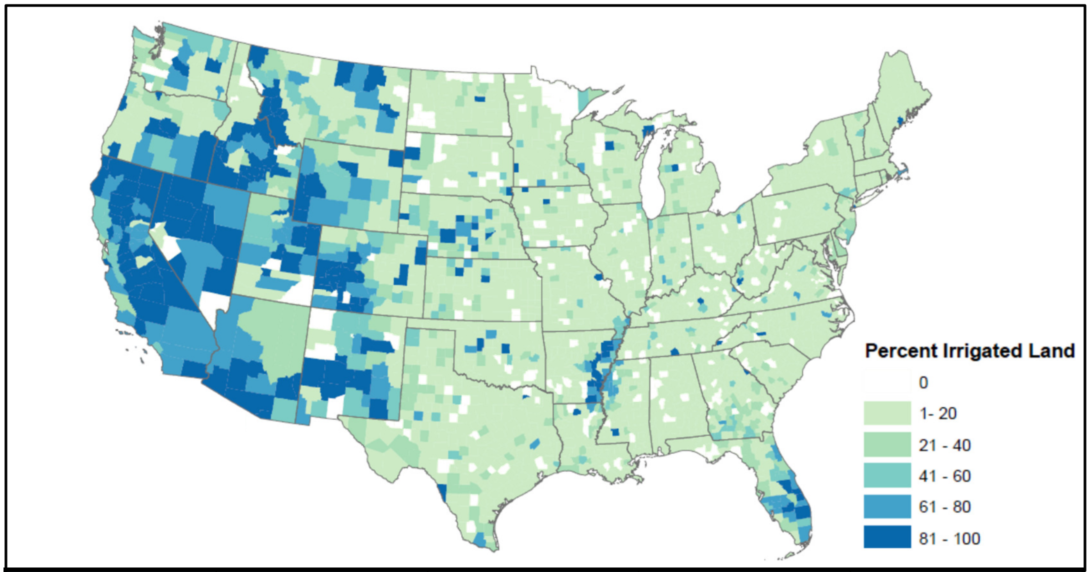
In the absence of tilling, insoluble constituent mass will accumulate in the topmost soil at higher rates than modeled in this evaluation. However, the lowest soil benchmarks identified for several constituents (e.g., selenium) are protective of plants. Higher temporary accumulation in the soil surface is unlikely to impact uptake across the root zone. Potential risks for other receptors in tilled soil were only identified after nearly 100 years of consecutive application. A shorter period without tilling is unlikely to result in exposures appreciably higher than those modeled in this evaluation. Furthermore, it is unlikely that the applied gypsum will remain at the soil surface for 20 years or more. Other additions to the soil, such as plant residue and manure, will further dilute and limit exposures to the topmost soil. Therefore, EPA concluded that the magnitude of this uncertainty is small.

7.1.3. Water Budget

EPA used precipitation data from climate stations together with regional soil properties to model infiltration rates across the country. Any water not lost to infiltration or evapotranspiration was assumed to run off into nearby water bodies. **Appendix E (Probabilistic Modeling)** details the methodology used to estimate this mass balance. The Agency is aware of other potential sources and sinks for water, but it was not possible to incorporate each one quantitatively with the data available. The following text discusses the methodology used to model water flow through the environment and potential sources of uncertainty associated with the available data.

Irrigation:

In some areas of the country, precipitation is not sufficient in volume and/or frequency to meet crop needs. In these areas, rain water may be supplemented through irrigation from nearby surface or ground water. EPA did not identify a reliable means to estimate the additional volume of water that might be applied on an annual basis. This amount is likely to vary each year based on rainfall, irrigation water availability, and the type of crops grown. The rate and time (day and year) of application will influence how much of the irrigation water evaporates, infiltrates, or runs off. To better understand the effect this uncertainty might have on calculated risks, EPA multiplied the total cropland in each county by the percent irrigation reported in the 2012 Census of Agriculture (USDA, 2014b) and then calculated the fraction of total cropland irrigated in each use zone. **Figure 7-2** presents data on the prevalence of irrigation as a fraction of total field area both in each county and for each use. Percentages were calculated on a county basis and so do not align exactly with the HUCs considered in this evaluation.



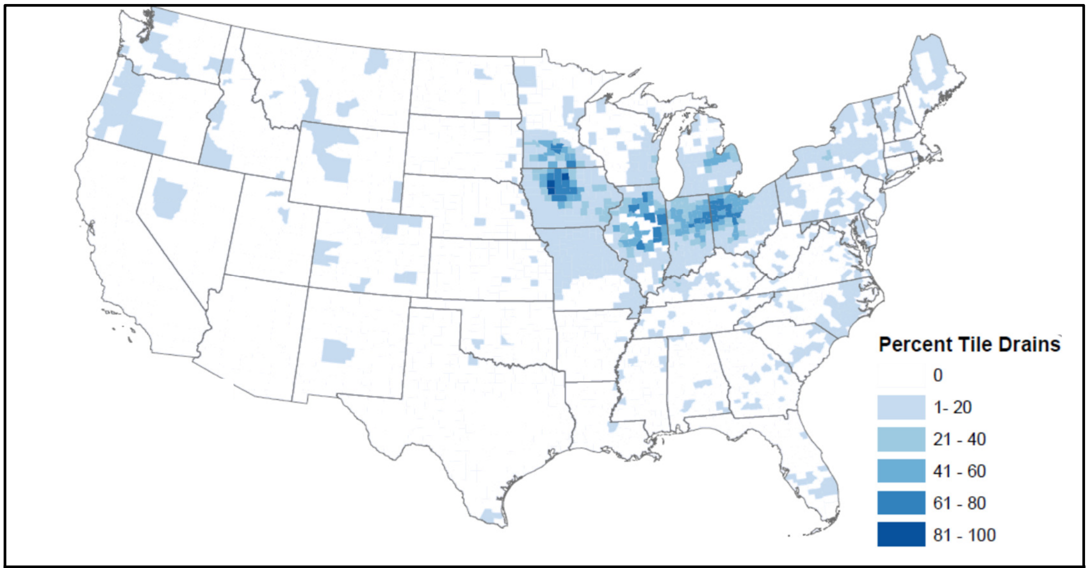
Phosphorus	Ca or S Nutrient	Infiltration	Sodic Soils	Aluminum Toxicity
10.2%	11.7%	11.2%	13.2%	18.3%

Figure 7-2: Percentage of agricultural land irrigated in each use area

The greatest density of irrigated fields occurs in the western United States, in areas that fall largely outside of the economic feasibility zone for FGD gypsum. Given the arid environment in these locations, it is likely that irrigation rates are more closely tied to plant requirements with an aim to minimize losses to runoff or infiltration. Higher density irrigation on the east coast occurs in areas in Florida and along the Mississippi river that already receive a substantial amount of precipitation. In these areas, the available constituent mass applied each year was often depleted over the course of the year by precipitation alone. Therefore, any further infiltration or runoff from irrigation is expected to only further dilute concentrations in the environment. As a result, the effects of this uncertainty on calculated risks are anticipated to be minimal.

Tile Drains:

Subsurface drainage tiles are used in areas of flat terrain and poorly drained soil to drain away high ground water tables and prevent the inundation of fields. Tile drains have been used since the early 1900s, primarily in the Midwestern United States. There remains a fair amount of uncertainty about the exact location and spacing of tile drains in the United States (Williams et al., 2015). However, even if the location of these tile drains were well known, some are quite old and may have become so clogged with sediment over time that the capacity to transmit water is greatly diminished. To better understand the effect this uncertainty may have on calculated risks, EPA estimated prevalence of tile drains based on data reported by the World Resources Institute (WRI, 2007). **Figure 7-3** presents data on the prevalence of tile drains as a fraction of total field area in each county and in each use zone. Percentages were calculated on a county basis and so do not align exactly with the HUCs modeled in this evaluation.



Phosphorus	Ca or S Nutrient	Infiltration	Sodic Soils	Aluminum Toxicity
8.6%	7.6%	7.5%	1.0%	4.0%

Figure 7-3: Percentage of agricultural land with tile drains in each use area

EPA used the base flow index (BFI) to estimate the relative fraction of surface water flow that originates from ground water and overland flow (or near surface discharge). The USGS calculates BFI with the approach proposed by the British Institute of Hydrology (Institute of Hydrology, 1980). The method uses measured flow minimums to estimate the annual volume of base flow to water bodies and calculates a ratio of the base flow to the total flow volume for a given year based on multiple years of data. Therefore, to the extent that the existing tile drains still divert infiltration directly into water bodies, the effects should already be reflected in this ratio. As a result, the effects of this uncertainty on calculated risks are anticipated to be minimal.

Precipitation Data:

The weather data used in this risk assessment was collected for a period from 1961 through 1990. Therefore, some uncertainty is introduced because any changes in weather patterns that have occurred since 1990 are not reflected in this data set. The National Climate Assessment Report documents region-specific changes in rainfall, temperature and episodic rainfall events over recent decades (Melillo et al., 2014). In general, this report identified a trend towards greater amounts of rainfall that are more concentrated in discrete events, particularly in the Northeast and Midwest. More intense storms could result in a larger fraction of the precipitation directed to overland runoff than predicted in this evaluation because storm events would be more likely to exceed the rate that water can infiltrate into the soil. This might increase or decrease concentrations in water bodies in different circumstances. For example, concentrations may increase from the greater constituent mass that would flow directly into the water body, but concentrations may be balanced out by greater runoff from the remainder of the watershed or decrease from higher total flow from contributed by upstream watersheds. Thus, the overall effects of this uncertainty are unknown.

7.1.4. Fate and Transport

EPA used data from the LEAF test methods to estimate the initial release of constituent mass from FGD gypsum. EPA then used a combination of EPACMTP and other models to simulate the subsequent movement of these constituents through the environment. **Appendix E (Probabilistic Modeling)** details the methodology used to model fate and transport. The following text discusses potential sources of uncertainty associated with the data and models used to estimate the fate and transport of constituents in this evaluation.

Leaching Behavior:

EPA made an initial determination about constituent leaching behavior based on a comparison of mean washed and unwashed concentrations measured with LEAF Method 1313 across the relevant pH range, as discussed in **Section 4 (Comparison with Analogous Material)**. A second, more refined analysis of leaching behavior was based on measured concentrations at each pH, as discussed in **Section 6 (Risk Modeling)**. The agreement between the approaches is generally good, with the exceptions of antimony and lead. A comparison of washed and unwashed samples in **Section 4** indicated that these two constituents are availability-limited over the relevant pH range, while the comparison in **Section 6** indicated the constituents are solubility-limited. **Figure 7-4** presents the pH-dependent leaching behavior of two sets of samples for lead. These two sample pairs were chosen because the unwashed data were detected over the majority of the pH range. WAU/WAW and TAU/TAW are the sample IDs for unwashed/washed sample pairs.

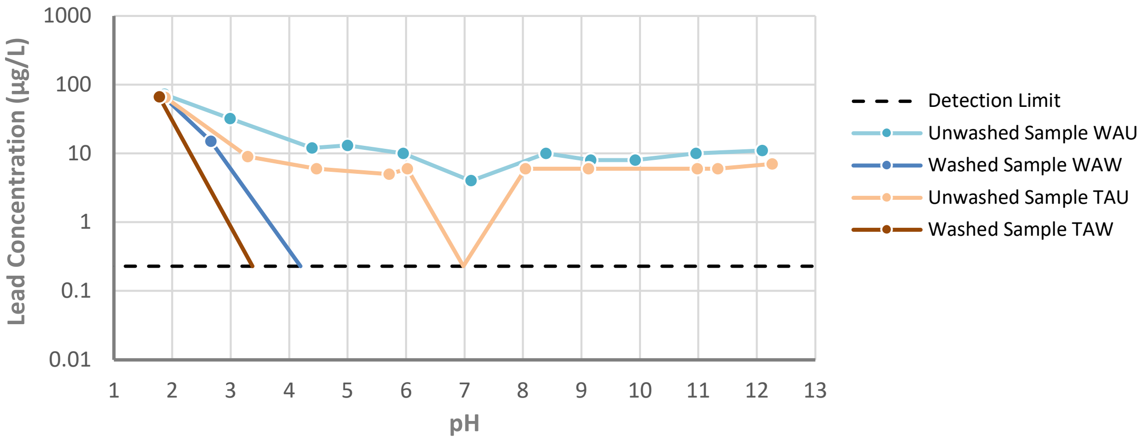


Figure 7-4: Comparison of lead leached from washed and unwashed samples.

At a highly acidic pH values, lead appears to be solubility limited because washed and unwashed concentrations are nearly identical. Yet, as the pH increases (i.e., becomes more basic), differences between washed and unwashed samples become apparent. After washing, samples measured above a pH of 3 are all non-detect, which indicates washout is occurring. It is likely that this discrepancy is caused by the presence of different lead compounds within the gypsum. The first is more soluble and readily washes out around a neutral pH during washing. The second is only soluble at a highly acidic pH and is retained during washing. Because the method used to determine leaching behavior in **Section 6** is based on a comparison with the highest measured leachate concentration, it appears that the constituent is solubility-controlled over the full pH range. It is more likely that, under typical environmental conditions, a small fraction of the constituent mass would quickly wash out, followed by solubility-controlled leaching at a far lower rate for any remaining leachable mass.

Calculating the available content based on the maximum concentration over the relevant pH range, rather than the full pH range, results in a 90th percentile available content closer to 5% of the total mass. This is far lower than 100% of the total mass used in this evaluation. Thus, the current evaluation overestimates exposures to these two constituents due to leaching. However, because neither antimony nor lead were found to be risk drivers in this evaluation, the magnitude of this uncertainty is considered negligible.

Field Distribution:

Due to model limitations, EPA had to assume that all the farm fields with FGD gypsum applied in a given watershed formed a continuous parcel of land and, thus, a single source of leachate. In reality, fields can be dispersed widely and non-continuously across the landscape. The greater the distance between individual fields, the greater the opportunity for dilution and attenuation in the environment before a release reaches downgradient wells or water bodies. Additionally, some farm fields may be located downgradient of or entirely outside the flow path of some private wells, limiting the impact to some water supplies. This is likely to result in an overestimation of risk for groundwater and surface water to some degree; however, the magnitude of this uncertainty is unknown.

Temporal Variability of Releases:

This beneficial use evaluation focused on the potential for adverse impacts associated with longer-term exposures, which are based on environmental concentrations averaged over the course of a year. This approach is believed to reasonably reflect exposures that may result from groundwater pathways, such as discharge to surface water, because transport is a gradual and continuous process. However, runoff events are intermittent throughout the year. As a result, there is potential for leaching of most soluble constituents at higher concentrations following individual runoff events. This might result in shorter periods of higher concentration in nearby streams than modeled in this evaluation.

EPA did not identify sufficient models or time-dependent data to support estimates of shorter-term exposures on a national scale. Such modeling would require information on both the specific time of application at different fields across a watershed, the specific location of the fields relative to the water bodies, and the relative timing, intensity, and duration of individual runoff events. The current model instead assumes that mixing of precipitation and FGD gypsum is uniform, that contact between the two occurs long enough to achieve near equilibrium concentrations in the runoff, and that there are no losses of dissolved constituent mass as runoff flows to nearby water bodies. However, there are a number of reasons why these assumptions may not always hold.

FGD gypsum is unlikely to be applied to all modeled fields at the same time or even in the same year. Thus, releases to runoff from different parts of the watershed can occur at different times, resulting in greater dilution of runoff from individual fields. Applications will not occur if the ground is already saturated due to difficulty operating spreading equipment on water-logged soil. As a result, some of the initially released mass would first infiltrate to the subsurface and smaller precipitation events may not exceed the initial abstraction at all (i.e., water diverted to infiltration, evaporation, or other pathway prior to runoff). Once runoff begins, the duration of contact between the flowing water and the soil will decrease. As a result, runoff may not always have enough sustained contact with the FGD gypsum or mixing with the intermingled water for leachate to achieve equilibrium concentrations. Even after release, there is potential for sorption and other interactions between runoff and the soil matrix that could limit immediate transport to surface water and may further promote infiltration to groundwater. Finally, not all runoff will reach the water body at the same time or same location. Travel times from the furthest point of a HUC10 watershed to a higher-order stream can take multiple days. Combined, all of these considerations will considerably limit shorter-term concentrations in water bodies.

There is also some potential for the methods used to estimate the magnitude of selenium in leachate to overestimate short-term concentrations. Batch leaching tests, such as EPA Method 1313 and 1316, measure dissolved concentrations under equilibrium conditions. As a result, these tests do not provide any information about factors that would affect the rate of dissolution prior to reaching equilibrium. For example, the selenium captured in FGD gypsum may associate with calcium. It has been shown that shown that selenate can substitute for sulfate in the gypsum structure (Fernández-González et al., 2006). Because this selenium is incorporated within FGD gypsum, rather than sorbed to the surface, releases would be limited by the rate at which the bulk gypsum dissolves. Therefore, it is unlikely all of this selenium would be released until all of the applied FGD gypsum had also been depleted from the fields.

Some studies have analyzed runoff from fields treated with FGD gypsum (Torbert and Watts, 2014; Watts and Torbert, 2017; Schomberg et al., 2018; Torbert et al., 2018). Washed FGD gypsum was applied to field plots and the runoff generated by simulated rainstorms was collected at 10-minute intervals. The studies reported that cadmium was not detected in any samples (<2 μg/L), manganese was detected only in initial runoff events (65 to 290 μg/L), and selenium was detected only in initial runoff events at low concentrations (5 μg/L). The reported values for cadmium and manganese align well with median leachate measured for washed FGD gypsum over the relevant pH range (0.85 μg/L and 65 μg/L, respectively), but the values for selenium are considerably lower than the median washed leachate (45 μg/L). These studies provide some confirmation that measured leachate concentrations can provide a reasonable estimate of runoff concentrations and may, in some cases, overestimate these releases. However, it is difficult to draw broader conclusions from the studies because of the lack of data on leaching potential of the FGD gypsum prior to application, different soil types, and other environmental conditions evaluated. Therefore, the magnitude of this uncertainty is not known.

Water Body Size:

To estimate ecological exposure to surface water and sediment, EPA modeled concentrations at the outfall of 1st and 2nd order streams (“headwater streams”) to any higher-order streams. Stream order is based on Strahler number, which assigns an order of 1 to initial headwater streams and increases each time two streams of the same order intersect, and was used as a metric for relative stream size and flow. The rationale for this approach is that a majority of runoff first flows through headwater streams, which provides a best estimate of immediate releases prior to further mixing and dilution during flow through successive watersheds. The cumulative land area that drains to these streams provides an extensive amount of habitat for wildlife, though is possible that some of the smaller streams are too small or ephemeral to sustain a complex ecological community. It is not known whether or to what extent this approach may overestimate risk; however, it is unlikely to underestimate risk.

To estimate human exposure to fish, EPA modeled concentrations at successive HUC10 outfalls until the stream order reached 6th order or above, as well as any HUC12 outfalls that discharge directly into high-order streams. The rationale for this approach is that:

- Streams below 3rd order are unlikely to support a fish population that could sustain fishing at the rates that correspond with the ingestion rates modeled in this evaluation;
- That data used to characterize some variables, such as BFI, may become less reliable in high-order streams due to long travel times and large cumulative drainage areas;
- The probabilistic modeling of surface water concentrations over the great distances covered by high-order streams was prohibitively time and resource intensive; and
- The addition of HUC12 outfalls captures releases from land area that would have otherwise been omitted due to the dominance of high-order streams, such as the Mississippi River, in certain regions of the country.

The current evaluation may overestimate risks to some degree by excluding high-order streams. It is anticipated that concentrations will generally decrease as stream order increases because the total drainage area contains proportionally less agricultural land. Yet, streams between 3rd and 5th orders represent nearly 90% of the flowlines above 2nd order (U.S. EPA, 2008b). Therefore, the exclusion of even higher-order streams may not have a substantial impact on overall risks.

Water Body Type:

As part of this evaluation, EPA modeled the transport and accumulation of constituent mass in surface water as it flowed through multiple HUC10 watersheds. This evaluation relied on the National Hydrography Dataset (NHD) to define the direction and magnitude of flow in each segment of the water body. However, there were not sufficient data to model every water body within the defined use zones. In particular, EPA was not able to model "terminal water bodies," which are those with an NHD flowline that has a terminal flag (i.e., unidirectional flow over the ground surface stops). For example, fish ponds and other relatively small and static (i.e., lentic) water bodies. Because there is no flow path through these water bodies, there is no information that could be used to estimate the associated volume or turnover rate. Modeling these water bodies would require a number of additional assumptions that would introduce a significant amount of uncertainty into the evaluation. It is possible that the exclusion of these water bodies underestimates potential risks because the relatively small volume combined with longer hydraulic residence time could result in longer exposures to higher concentrations. In addition, EPA developed a separate water quality benchmark for selenium in these lentic water bodies to account for the effects of prolonged exposure (U.S. EPA, 2016c). Therefore, further evaluation may be warranted before FGD gypsum is applied in the vicinity of these types of water bodies.

7.1.5. Exposures

EPA used the constituent concentrations modeled in each medium together with available data on receptor characteristics, behavior and sensitivity to estimate potential exposure and resulting risks. **Appendix B (Benchmarks)** details the data and approach to develop benchmarks used to calculate risk. The Agency is aware of other potential receptors and types of exposures beyond those evaluated, but could not quantitatively account for them with available data. The following text discusses the potential sources of uncertainty associated with the data and methodology used to calculate exposure and subsequent risk.

Fish Ingestion Rates:

Results presented in **Section 6 (Risk Modeling)** reflect modeled risks for recreational fishers. These receptors were selected because they are more likely to consume fish caught from a single water body. Therefore, these individuals and their families represent a sensitive subpopulation that is more likely to be exposed through fish ingestion (U.S. EPA, 2011). Subsistence fishers are another sensitive subpopulation that could be exposed at levels higher than the general population. This subpopulation is not well defined or characterized and may include a diverse range of rural and urban receptors that rely on fresh-caught fish as a major portion of their overall diet. Given the far greater uncertainty associated with these subsistence fishers, EPA considered these receptor as part of an uncertainty analysis to provide further context for results. The primary difference between recreational and subsistence fishers in this evaluation is the rate of fish ingestion, with subsistence fishers consuming about three times more fish. However, there are uncertainties associated with the ingestion rates used for both recreational and subsistence populations.

The full-scale model relied on a fixed, high-end ingestion rate to characterize fish ingestion for each age cohort due to a lack of data that would allow for a broader characterization of these subpopulations. This can overestimate exposures because a single value does not reflect the full variability of the modeled population. This uncertainty will be greater in areas where diets may vary throughout the year based on seasonal access to fish and the availability of other protein sources, such as wild game. The model also assumed that all the fish consumed were caught from a single affected waterway. This could overestimate exposures to the extent that the diet also incorporates fish sourced from beyond local waters. As a result, the data used to characterize fish exposure is expected to overestimate potential risks. However, the magnitude of this uncertainty is not known.

Constituents Without Benchmarks:

There are some constituents for which human or ecological benchmarks were not identified. It was not possible to quantitatively evaluate these constituents in either the screening or full-scale modeling. For other constituents, toxicity values were identified for some, but not all, relevant exposure pathways. In these cases, the potential risks to receptors in these media could not be fully quantified. The absence of a toxicity value is not necessarily equivalent to the absence of risk. Constituents may pose pathway-specific risks or may influence the fate and transport or toxicity of another constituent, resulting in an underestimation or overestimation of risk. The magnitude of this potential underestimation is unknown.

Aggregate Exposures:

Aggregate exposure is the combined exposure to a single constituent through multiple exposure pathways. Aggregate exposures may be simultaneous or sequential, but all occur within the critical window for the health effect. This beneficial use evaluation considered potential risks to human health from individual constituents and individual pathways. It is possible that individuals could be exposed simultaneously through ingestion of ground water, soil, produce, livestock and fish. However, it is highly unlikely that receptors would be exposed to high-end concentrations through every route. Even if receptors are exposed to high-end concentrations through every pathway, the constituents found to accumulate in each media are different. Therefore, the overall magnitude of this uncertainty is considered minimal.

Cumulative Exposure:

Cumulative exposure is the combined exposure to multiple stressors that produce the same health effect. These different stressors may interact with one another in antagonistic or synergistic ways that serve to mitigate or exacerbate potential health effects. The extent of these interactions may change based on the level of the stressors present and the order of exposure. The toxicity values used in the current evaluation do not account for these types of interactions. Where the potential for simultaneous exposure to multiple constituents exists, current EPA policy is to assume that the risks resulting from these exposures are additive (U.S. EPA, 2000).

The only constituents carried forward to the full-scale evaluation that share a similar health endpoint are arsenic and chromium in ground water (cancer) and mercury and selenium in fish (neurological). Arsenic and chromium are both far below levels of concern in all media and so the uncertainty associated with this endpoint is minimal. However, mercury and selenium were both identified near levels of concern for fish ingestion. There is no relationship between the levels of mercury and selenium in FGD gypsum, so it is unlikely that receptors would be exposed to high-end concentrations of both constituents at the same time. Furthermore, numerous investigations have found selenium can mitigate the toxicity of mercury (HHS, 2003). Recent studies have proposed several mechanisms for detoxification, such as mercury sequestration in metabolically inert compounds, formation of selenium-based antioxidants, demethylation of methylmercury, or replenishment of selenium-containing enzymes needed for metabolism (Bjørklund et al., 2017; Ralston and Raymond, 2018). Despite this evidence from the literature, it is not possible to quantify whether and to what extent selenium will reduce mercury toxicity in all circumstances. However, it is unlikely that cumulative exposure will compound the risk of the two constituents. Therefore, the uncertainty associated with these endpoints is considered minimal.

Sulfate does not share any known health endpoints with selenium; however, available research indicates the presence of dissolved sulfate can reduce the bioavailability and toxicity of selenium (Banuelos et al., 1990; Banuelos & Mayland, 2000; Bell et al., 1992; Brix et al., 2001; Chaney et al., 2014; Hopper & Parker, 1999; Qin et al., 2013; and Yang, 1995). This is attributed to the fact that selenium and sulfur have similar ionic structures and so the two elements can be transported by the same membrane proteins. As a result, sulfate can compete with selenium for cell uptake. The potential risks from selenium identified in this evaluation are driven by bioaccumulation in the food chain. Thus, competition between sulfate and selenium for uptake by algae and other lower trophic level organisms in surface water (e.g., Daphnia) would also result in larger reductions in uptake by fish and other higher-order receptors. Sulfate is a primary component of FGD gypsum ($CaSO_4$) and leaches at concentrations at or above 1,000 mg/L from both washed and unwashed samples. Brix et al. (2001) studied selenium in aquatic environments and found that sulfate strongly inhibited selenium uptake even at much lower concentrations. Yet there have also been documented cases of selenium accumulation in wildlife around waters with sulfate concentrations as high as 2,000 to 100,000 mg/L (Birkner, 1978; Skorupa, 1998). The differences among these studies is likely associated with the oxidation state of selenium, with sulfate competing with most effectively with selenate (Se+6, VI) (Ogle et al., 1988). EPA was not able to quantitatively evaluate transport of sulfate ions through the subsurface or the impact on selenium uptake at various concentrations and different environmental conditions due to methodological challenges and a lack of relevant data (U.S. EPA, 2016c). Because any competition from sulfate would reduce exposure to selenium, the inability to quantify the effects of sulfate will result in an overestimation of risk. However, the magnitude of this uncertainty is unknown.

Selenium Speciation

The oxidation state of selenium can impact the mobility of this constituent in the groundwater. The most common forms of selenium found dissolved in groundwater under the standard range of environmental conditions are selenite (Se+4, IV) and selenate (Se+6, VI), with the latter as the more mobile form. Available information indicates that the dominant form of selenium expected in bulk FGD gypsum is selenate (EPRI, 2011). Therefore, all selenium applied and leached in the full-scale model was assumed to be present as selenate.

Previous modeling found the difference between high-end surface water concentrations resulting from groundwater transport for the two selenium species was over a factor of 100 due to differences in retention onto subsurface soils (U.S. EPA, 2014b). Given the magnitude of the difference relative to the modeled risks for selenate, EPA did not separately model the transport of selenite. Once released into the environment, the dominant oxidation state will be controlled by local pH and redox conditions that can be influenced by plant and microbial activity. This evaluation could not consider how these types of site-specific factors may affect the oxidation state of selenium during transport through the subsurface. However, to the extent that some fraction of the selenium is either initially present as selenite or converted to this state after application, the full-scale model has the potential to result in an overestimation of risk for this constituent. The magnitude of this uncertainty is unknown.

Chromium Speciation:

Results presented in **Section 6 (Risk Modeling)** reflect the most toxic species of each constituent. For chromium, the hexavalent (Cr+6, VI) species is both more toxic and more mobile in the environment than the trivalent (Cr+3, III) species. EPA did not identify much data on how much, if any, chromium (VI) is initially present in FGD gypsum. Torbert et al. (2018) analyzed runoff from a samples off FGD gypsum and found measurable chromium (VI). Out an abundance of caution, the Agency initially assumed that all chromium was present in the hexavalent state. To understand the extent to which effects may vary, EPA recalculated risks with the reference dose for chromium (III). **Table 7-2** compares these results of the modeled risks for different chromium species.

Table 7-2. Comparison of Model Results for Hexavalent and Trivalent Chromium

Constituent	Wash Status	90th Percentile HQ				
		Phosphorus Runoff	Infiltration	Nutrient Amendment	Sodic Soils	Aluminum Toxicity
Chromium (VI)	Combined	**1.3**	**1.1**	0.57	0.75	0.74
Chromium (III)	Combined	0.02	0.01	< 0.01	< 0.01	< 0.01

Although the current evaluation assumed all of applied chromium was hexavalent, the model used empirical leachate data to estimate losses from the soil. The speciation of chromium in the leachate is not known. However, chromium (III) is less mobile and so could result in higher estimates of accumulation in the soil. If even a minor fraction of the applied chromium is trivalent, then it would eliminate potential long-term risks because the risks for chromium (III) are two orders of magnitude lower than those for chromium (VI). In addition, reduction of chromium (VI) has been shown to be energetically favorable and unlikely to reverse in high-organic, aerobic soils (Brose and James, 2010; HHS, 2012). Even if all the chromium present in FGD gypsum is hexavalent at the time of application, no single application at the rates considered in this evaluation would pose short-term risk. Therefore, EPA concludes that long-term risks from chromium in soil will also be below levels of concern.

Background Soil Concentrations:

Background concentrations are the constituent levels found in environmental media that have not been impacted by releases from the waste. Background concentrations may originate from natural or anthropogenic sources. The current evaluation assumed that background concentrations in each medium (e.g., soil, ground water) were negligible. The modeled exposures are based solely on releases from applied FGD gypsum. This approach was selected because background can be highly variable, even over small areas, and so it is not possible to reliably characterize contributions from background without robust, local data. The following text discusses the potential sources of uncertainty associated with background.

To better understand how long-term application of FGD gypsum might add to exposures from soil, EPA compared the modeled concentration of thallium accumulated from FGD gypsum application with measured background surface soil concentrations from across the United States (U.S. DOI, 2013). The results of this comparison are presented in **Table 7-3**. Although this type of data is useful for qualitative comparisons, the data cannot be used to reliably estimate total soil concentration that might result from application of FGD gypsum. Soil concentration are highly site-specific and can vary considerably over small areas. Therefore, a nation-wide or even a state-wide dataset may not provide accurate estimates of total concentrations. In particular, agricultural fields that have been heavily engineered may have higher concentrations of some elements than other undisturbed soil. Therefore, the concentrations in this table should not be extrapolated outside of this limited context of this discussion.

Table 7-3. Comparison of FGD Gypsum and Surface Soil Concentrations

Constituent	Percentile FGD Gypsum Accumulation (mg/kg)		Percentile Background Surface Soil (mg/kg)		
	50th	90th	10th	50th	90th
Thallium	0.001	0.05	0.2	0.4	0.7

Long-term accumulation of thallium from FGD gypsum is lower than the majority of background surface soil concentrations, even at high-end concentrations. That would mean that more than 90% of existing surface soils pose higher risks than those modeled in this evaluation. The majority of thallium in background soil may not be bioavailable in some areas, but this is unlikely to be true across the entire country. Instead, it is more likely that this evaluation overestimated exposures to beef and milk through the combination of high-end data and assumptions intended to protect human health. There is greater uncertainty associated with the beef and milk pathways because both require modeling accumulation in the soil, followed by sequential uptake into plants and cattle prior to ingestion by human receptors. These multiple levels of accumulation compound the uncertainties.

It is not possible to substantially refine risk estimates with available data. However, the fact that high-end (i.e., 90th percentile) thallium accumulation in soil from FGD gypsum application is lower than the low-end (i.e., 10th percentile) of existing background concentrations indicates that contributions to existing exposures is minor. The majority of thallium was found to remain in the soil, rather than be released into infiltration or runoff, so it is not likely that the magnitude of potential thallium accumulation was underestimated. Therefore, given the low exceedance of the health-based criteria identified in the full-scale model, EPA concludes that all risks from thallium in soil are below levels of concern.

Background Water Concentrations:

Mercury and selenium are the two modeled constituents found at or near levels of concern in surface water. FGD gypsum is not the only source of these contaminants in the environment. Other point and non-point sources from either natural or anthropogenic sources can also contribute to levels in surface water. Both constituents are causes of contamination reported by states under the Clean Water Act Section 303(d). This list of impaired or threatened waterways is compiled by states, which have primary responsibility to notify the public of chemical contamination that may present a public health hazard. **Figure 7-5** presents two maps of waterways within the economic feasibility zone that were reported as impaired for selenium and mercury for any reason during the most recent round of reporting in 2016. Impaired waters are shown in red. Blank areas represent regions outside the feasibility zone or where state data were not available in a form that could be readily mapped.

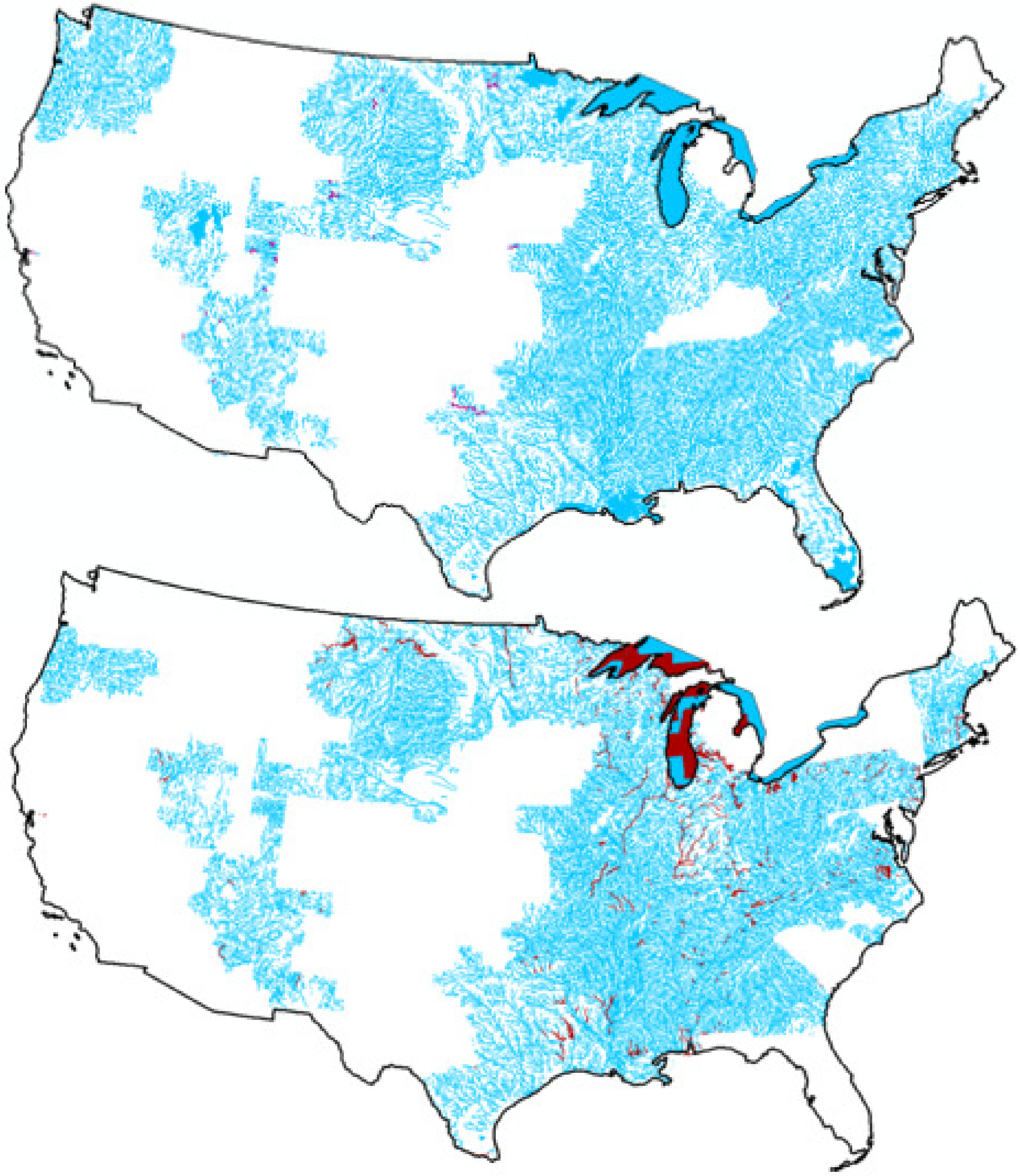


Figure 7-5: Occurrence of impaired waterways for selenium (top) and mercury (bottom)

These figures are provided for illustrative purposes only. The available data are not complete and recommendations based on this current list will gradually become outdated as sources of contamination are addressed and updated. Proximity to an impaired waterbody does not necessarily mean use of FGD gypsum is inappropriate. The majority of modeled scenarios fell far below levels of concern and are not likely to represent a substantial source of either mercury or selenium. However, existing (background) sources of contamination can still be an important consideration when determining where application of FGD gypsum may be appropriate.

7.2. Sensitivity Analyses

The purpose of these sensitivity analyses is to review the results of the full-scale model reported in **Section 6 (Risk Analysis)** and identify any sensitive model inputs that could be used to limit releases and reduce modeled risks to below levels of concern. Based on the uncertainty analyses, EPA previously determined that all risks from soil pathways fall below levels of concern and so the following discussion focuses on the remaining risks identified for releases to surface water. The model found that risks for this pathway fell below levels of concern for a majority of model runs; risks are instead driven by high-end application scenarios. Therefore, it is likely that modeled risks can be mitigated with only minor limits on applications. Such limits can inform best management practices for application of FGD gypsum; however, the limits identified in this evaluation are intended to be informative and not prescriptive. States and others knowledge of actual application practices and local environmental conditions should make the decision about the appropriateness of any limits on use.

7.2.1. Constituent Concentrations

Identified risks from FGD gypsum might be managed through limits on the concentrations allowed in FGD gypsum. However, the total mass of a constituent is not always a reliable indication of how much can readily leach out. Therefore, EPA plotted the total and leachable contents of each sample for which both were available to better understand whether limits based on total content could reliably reduce leachable content. **Figure 7-6** presents the results of this comparison. The graph contains data for washed and unwashed samples and are intended to present general relationships, rather than representative distributions. Consideration of washed and unwashed data separately did not substantially alter the relationship.

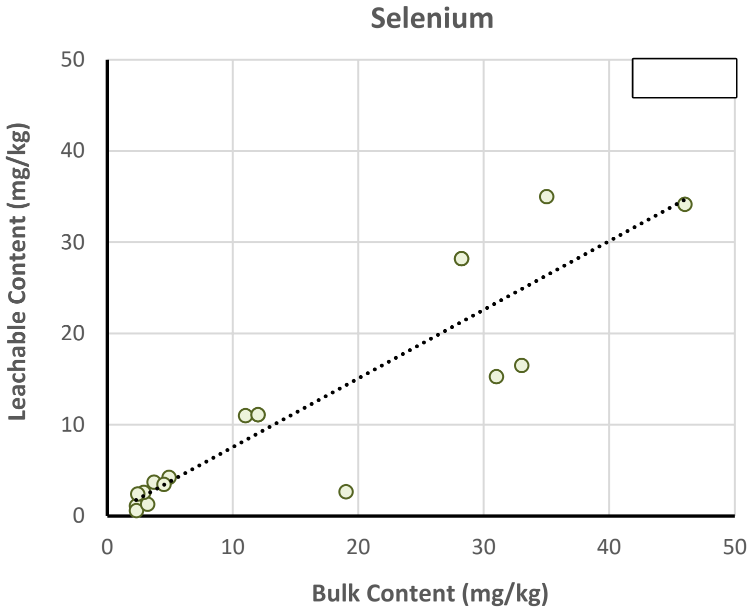


Figure 7-6: Relationship between bulk and leachable content

There is a clear relationship between the amount of selenium in FGD gypsum and the amount available to be leached out. In many cases, almost all of the mass in the FGD gypsum is leachable, though it may be released gradually over time. Based on this relationship, limits on selenium concentration may be one method to control releases. Therefore, EPA filtered the full-scale model, controlling for bulk selenium concentration in the applied FGD gypsum, to better understand the potential effect of such limits on national risks. This review indicates that application of FGD gypsum containing less than 25 mg/kg selenium would not pose any concerns to human health or the environment when applied at agronomically relevant rates. This identified concentration corresponds to the 90th percentile of all model runs. As a result, the vast majority of FGD gypsum generated in the United States would not warrant any limits when applied in agronomically relevant rates.

7.2.2. Application Rate

There is a clear and direct relationship between the mass of FGD gypsum applied and the amount of selenium that can be released from a watershed. Thus, identified risks may be managed through limits on the rate at which the FGD gypsum is applied to fields. EPA filtered the full-scale model results, controlling for application rate, to better understand the potential effect of such limits on national risks. This review indicates that an average application around 1 ton/acre would not pose any concerns to human health or the environment, even if widely applied across a watershed.

This identified rate is greater than the time-averaged, high-end rates for several uses: 1.7 tons/acre every 2 years for nutrient application, 10 tons/acre every 10 years for sodic soils, and 11 tons/acre every 10 years for aluminum toxicity. Although use for sodic soils and aluminum toxicity tend to have higher individual-year application rates, the total area of over which FGD gypsum is expected to be applied be far smaller. This is because these two uses are expected to target to specific problem areas, rather than entire fields or watersheds. Based on this sensitivity analysis, it is expected that these three uses of FGD gypsum would not warrant any limits when applied in agronomically relevant rates. Use to prevent phosphorus runoff and improve drainage might pose concerns where FGD gypsum is annually applied across a watershed. In these instances, limiting applications to an average of 1 ton/acre can ensure that these uses do not pose concern to either human health or the environment.

7.2.3. Application Area

There is a clear and direct relationship between the proportion of a watershed over which FGD gypsum is applied and the amount of selenium that can be released. Thus, identified risks may be managed through limits placed on the area of application. To better understand the potential effect of such limits on national risks, EPA filtered the full-scale model results to control for application area as a proportion of the total watershed. EPA controlled for the proportion because the size of each watershed can vary considerably. The same field area in a larger watershed would provide greater opportunity for mixing and dilution with precipitation that falls outside the fields. Thus, the proportion of the watershed with FGD gypsum applied provides a more consistent frame of reference for comparison. This review found that application on 40% or less of the drainage area for a headwater stream posed no concerns to human or ecological receptors. For comparison, the 50th percentile of modeled headwater drainage areas is about 1,300 acres, while the 90th percentile is about 5,800 acres.

The identified proportion of 40% corresponds to the 90th percentile of all model runs. This is both because the area dedicated to agricultural fields in many watersheds is already less than 40% of the total land area and because the model allowed the field area with FGD gypsum applied in each run to vary probabilistically. The maximum proportion in any individual model run was 93% of the entire watershed. Based on this sensitivity analysis, it is anticipated the use of FGD gypsum will pose no concerns in many regions of the country. Although it may ultimately not be practical to implement limits based on application area because there can be many land owners within a single watershed that would need to coordinate, the identified limit of 40% can still provide one means to understand where widespread application of FGD gypsum might warrant further review.

7.2.4. Regional Variation

Potential risks associated with the use of FGD gypsum may differ across the country as a result of local conditions (e.g., precipitation rate, amount of farmland). Thus, a single set of management standards may not be equally appropriate for each region. To understand how modeled risks vary based on geography, EPA aggregated model results for ecological exposure to selenium at a HUC4 level. As part of this analysis, EPA included all fields within the economic feasibility zone. **Figure 7-7** depicts how the 90th percentile of modeled long-term risks vary geographically. Each shaded region represents an individual HUC4, some of which extend outside the borders of the country.

Regions that are not fully shaded reflect those that fall partially outside the economic feasibility zone modeled in this evaluation.

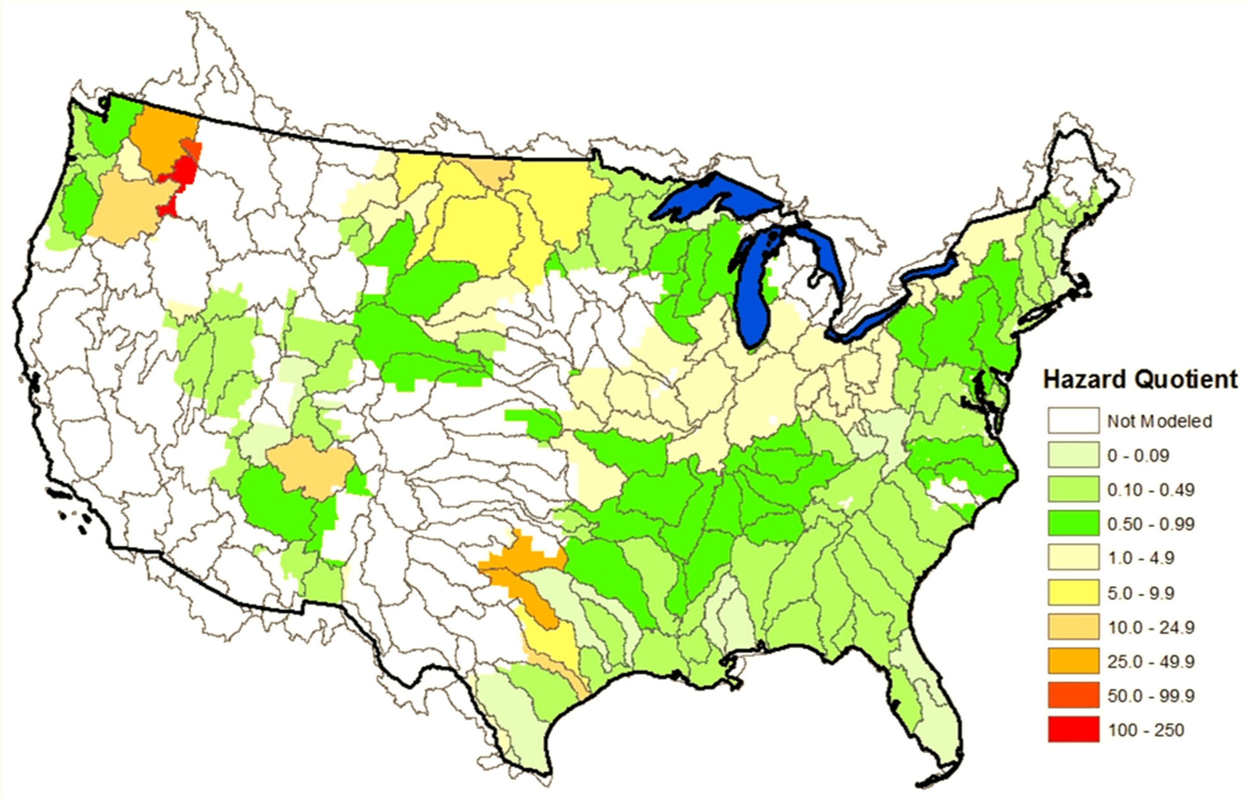


Figure 7-7: Geographic variability of modeled risks by individual HUC4.

These results show that risks vary across the country. These findings generally align with previous sensitivity analysis on application area. Risks are lowest in the South and along both coasts where there is less cumulative field area for application of FGD gypsum. Risks increase somewhat in the Midwest where there is more agricultural land. Risks are more variable further west. Pockets of lower risk are attributed to the limited number of fields in these regions, which result in few model runs for those areas, combined with an arid environment where there is effectively no modeled infiltration or runoff. Pockets of disproportionately high risk are attributed to regions of low precipitation that could result in extremely high modeled concentrations for constituents like selenium with high solubility. The magnitude of risk in these parts of the country are expected be exaggerated, but caution may still be warranted because these regions tend to be those with greater tendency toward existing issues with water bodies impaired by selenium. Although modeled risks can be disproportionately high in the west, it is clear that the small number fields and associated model runs did not inappropriately skew high-end risks on a national scale. Based on this sensitivity analysis, it is anticipated the use of FGD gypsum will pose no concerns in many regions of the country.

7.3. Summary

EPA applied the best information available to characterize the most likely management and release scenarios with an aim to minimize the influence of uncertainties on a national scale. As a result, while there is potential for management practices and associated releases at individual farm fields to diverge from those modeled, it is expected that the broader potential for releases is adequately captured within the probabilistic model. Yet sources of uncertainty inevitably remain that may individually underestimate or overestimate risks to some degree. In this section, EPA analyzed these cumulative effect of these uncertainties to better understand the magnitude of each and the potential to impact the model results reported in **Section 6 (Risk Modeling)**.

EPA identified two instances where uncertainties resulted in an overestimation of risk substantial enough to impact the conclusions drawn from the full-scale model. Both instances involved accumulation of constituent mass in soil. EPA determined that the full-scale model overestimated risks from chromium and thallium accumulation in soil:

- The full-scale model identified potential risks to ecological receptors from direct contact with and ingestion of chromium in soil. This analysis assumed that all of the applied chromium mass was present in the hexavalent oxidation state. This assumption was made because EPA did not identify any data on the speciation of chromium or any other constituent in FGD gypsum. However, based on a review of the literature, EPA concluded that hexavalent chromium applied to agricultural soils will tend to convert to the trivalent state over time. Therefore, it is unlikely that hexavalent chromium will accumulate to the concentrations modeled.
- The full-scale model identified potential risks to human health from ingestion of thallium that had accumulated in both milk and beef from livestock. However, a comparison of modeled thallium concentrations with measurements of surface soils across the country found the accumulated mass to be a small fraction of the existing mass in background soil. Were this accurate, over 90% of background soils would already pose greater risk. The modeled risks are more likely to be driven by the compounding uncertainty of successive accumulation from gypsum-amended soil into the forage or feed and then into the livestock.

Based on these considerations and fact that both constituents had low exceedances of health-based criteria after nearly 100 years of application, EPA concluded that the accumulation of chromium and thallium in soil pose no short- or long-term risks to either human or ecological receptors. With elimination of these two constituents, all potential risks from exposure to soil fall below levels of concern. Therefore, EPA concludes that accumulation of FGD gypsum in agricultural soils does not warrant further evaluation.

EPA identified a number of uncertainties for releases to surface water that have the potential to either overestimate or underestimate risk, but the data available to characterize these uncertainties are often limited. A qualitative review of these uncertainties found many are likely to be relatively minor or have no effect on conclusions (e.g., overestimation of risk for constituents found to pose no concern). The single greatest uncertainty identified how widely or frequently this material will applied in the absence of any restrictions. Therefore, EPA assumed that gypsum could be applied as frequently as every year on all agricultural fields where it might provide a benefit. As a result, modeled results will overestimate risk to the extent that FGD gypsum is not spread as widely or as frequently due to economic or other practical considerations. Yet it cannot be ruled out that such widespread applications are possible. Therefore, EPA determined that it was most appropriate and protective to draw conclusions about the full range of potential uses based on the results of the full-scale model.

EPA conducted sensitivity analyses to review the results of the full-scale model reported in **Section 6 (Risk Analysis)** and identify any sensitive model inputs that could be used to reduce the modeled risks to below levels of concern. These analyses confirm that modeled risks are driven by a small subset of model runs that reflect wide applications at high concentrations, rates, and frequencies. However, the vast majority of modeled application scenarios pose no concerns to human health or the environment. Indeed, there are a number of uses and regions of the country for there is likely no potential for concern at all. In instances where FGD gypsum might be applied at the highest rates and frequencies over a majority of a given watershed, it is possible to mitigate the potential for risk with minor limits on the either application rate and area or the concentration of selenium in the FGD gypsum. Based on these results, there is a high degree of confidence in the principal finding of the full-scale analysis that application of FGD gypsum will not pose any concerns in the majority of application scenarios.

8. Final Summary and Conclusions

The *Methodology for Evaluating Beneficial Uses of Industrial Non-Hazardous Secondary Materials* (U.S. EPA, 2016a) and the *Beneficial Use Compendium: A Collection of Resources and Tools to Support Beneficial Use Evaluations* (U.S. EPA, 2016b) provide both an analytical framework and resources that can aid states, tribes, local governments and others with evaluations of the potential environmental impacts associated with the beneficial use of industrial materials. This current evaluation provides an example of these resources applied to an unencapsulated beneficial use: FGD gypsum used as a replacement for mined gypsum on agricultural fields. This section provides a summary of the full evaluation and the conclusions that can be drawn based on a quantitative and qualitative review of all available information.

8.1. Evaluation Summary

EPA sequentially applied each step of the analytical framework, culminating is a national-scale probabilistic model of environmental fate and transport. Following completion of each step, EPA reviewed the findings and identified any individual constituents or exposure pathways that posed no concerns. These constituents and pathways were removed from further consideration before proceeding on to the next step to help streamline subsequent steps. The following text provides a summary of each step of this evaluation and the associated findings. The purpose of this summary is to highlight how each step contributed to the characterization of potential environmental impacts that might result from use of this industrial material. The first section of the document provides a general introduction to the evaluation. The subsequent sections are summarized below:

Section 2 (Planning and Scoping): Prior to any quantitative analysis, EPA reviewed all available information about FGD gypsum use and composition to define the scope of the evaluation. Every use of gypsum considered in this evaluation is either for application directly on the ground surface or mixed together with surficial soils. As a result, every use may result in the same types of releases to the environment. Thus, a single conceptual model was used to represent all uses. This conceptual model formed the basis for all subsequent data collection and modeling efforts.

Section 3 (Existing Evaluations): EPA reviewed the available literature and identified two sources with information relevant to the current evaluation. A review of the data quality in both found these sources to be an appropriate basis for conclusions about FGD gypsum used in agricultural applications. Based on the information provided by these existing evaluations, EPA concluded that potential exposures from windblown dust fall below levels of concern and that potential exposures from radiation are comparable to those from mined gypsum. Therefore, these pathways were not carried forward for further evaluation.

Section 4 (Comparison with Analogous Product): FGD gypsum was compared to mined gypsum to determine whether there is potential for greater releases from FGD gypsum than from analogous materials that would otherwise be used. Based on these comparisons, EPA concluded that many of the constituents in FGD gypsum are comparable to those in mined gypsum. This is because a major source of constituent mass in FGD gypsum is the limestone used in wet scrubbers. Limestone is a naturally occurring mineral similar to gypsum and so it is reasonable that the bulk content of many elements would be similar. The constituents found to be higher in FGD gypsum tend to be those that are most volatile during coal combustion. This allows the constituents to pass through particulate control devices and enter the wet scrubber, where they are captured along with sulfur dioxide. For releases to leachate, a comparison could not be conducted due to a lack of leachate data for mined gypsum. Therefore, all constituents were carried forward for this pathway. For releases to air, the comparison found the potential for greater volatilization of mercury from FGD gypsum. Therefore, this pathway was also retained for further evaluation.

Section 5 (Screening Analysis): EPA conducted a streamlined analysis that used a combination of high-end waste characterization data and simplifying assumptions to capture an upper bound of possible exposures. Exposures found to be below levels of concern based on this screening were eliminated from further consideration with a high degree of confidence. Based on the results of this analysis, all exposures from releases to air were found to be below levels of concern. Various constituents were retained for further evaluation of exposures from releases to soil (i.e., chromium, mercury, selenium, thallium); ground water (i.e., antimony, arsenic, chromium, thallium), surface water (i.e., arsenic, cadmium, chromium, iron, lead, mercury, manganese, selenium, thallium) and sediment (i.e., antimony, cadmium, chromium, lead, manganese, mercury, nickel, zinc).

Section 6 (Risk Modeling): EPA conducted a more refined, full-scale analysis to better incorporate the variability of constituent characteristics, environmental setting and receptor behavior that can impact constituent release, transport and exposure. The probabilistic results provide a best estimate of risks that may result from the use of FGD gypsum in agriculture. No concerns were identified from releases to ground water or sediment. High-end risks (i.e., 90th percentile) were identified for chromium for ecological receptors in soil, selenium for ecological receptors in surface water and human ingestion of fish, and thallium for human ingestion of beef and milk. Corresponding median scenarios (i.e., 50th percentile) are all one or more orders of magnitude below levels of concern for each of these exposure pathways.

Section 7 (Uncertainty and Sensitivity Analyses): EPA reviewed the major sources of uncertainty known to be associated with both the data and modeling approach used in this evaluation. The purpose of this review was to determine whether the magnitude of these uncertainties is great enough to alter the conclusions that would otherwise be drawn from the full-scale model. EPA did identify two instances where the full-scale model is known to overestimate risk to a degree that could affect the conclusions drawn from the full-scale model. The available data and assumptions used to model chromium and thallium accumulation in the soil are expected to substantially overestimate risk for these constituents. Based on this review, EPA concluded that accumulation of chromium in soil and thallium in beef/milk pose no concerns. Therefore, the only remaining risks identified in the full-scale analysis are from selenium accumulation in surface water.

EPA conducted additional sensitivity analyses to better understand what model inputs drive these remaining potential risks and might inform any limits on applications that could ensure they do not occur. These analyses show that modeled risks are driven by a relatively small subset of model runs that reflect wide applications at the highest rates and frequencies. However, the vast majority of modeled application scenarios pose no concerns to human health or the environment. In instances where applications are expected to be applied at the highest rates and frequencies over a majority of a given watershed, it is possible to mitigate the potential risks identified in this evaluation with minor limits on application practices.

8.2. Conclusions

Based on available data and modeling documented in this evaluation, EPA reached the following conclusions about the use of FGD gypsum as an agricultural amendment:

- The limestone slurry used in wet scrubbers is a major source of constituent mass to FGD gypsum. Both limestone and mined gypsum are naturally occuring and, as a result, many constituents are present at comparable levels in both FGD and mined gypsum. The constituents found at higher levels in FGD gypsum tend to be those that are most volatile during coal combustion. These volatile constituents are able pass more easily through particulate control devices and are instead captured in wet scrubbers along with sulfur dioxide. Little leachate data were identified for mined gypsum and so similar comparisons could not be made for that release pathway. It is possible the leaching behavior of some constituents may also be comparable between the two materials, particularly where the majority of the constituent mass originates from the limestone slurry. However, the constituents with the highest modeled risks were also those that are most volatile and most highly concentrated in FGD gypsum. Therefore, it is unlikely that this lack of data would alter the conclusions of this evaluation.
- Washing of FGD gypsum was found to reliably reduce the bulk content of boron, chloride and manganese. Data on the bulk content (i.e., mg/kg) for bromide was not available for comparison; however, it is expected that similar reductions would be found for this constituent. Washing was found to reliably reduce leachate concentrations (i.e., mg/L) of antimony, bromide, boron, cadmium, chloride, cobalt, lead, manganese, molybdenum, nickel, selenium and thallium. The magnitude of reduction varies among constituents. For antimony, bromide, lead, and thallium, washing reduced leachate concentrations in all samples to below detection limits. Reasons for the absence of reduction in other constituents may vary and include that a given constituent has low solubility under the conditions present during washing, a substantial fraction of the

overall constituent mass is present in a relatively recalcitrant form, and/or that a real reduction exists but the magnitude is too small to reliably distinguish from natural sample variability.

- Application of FGD gypsum did not result in accumulation of constituent mass in soil, crops, livestock, air, or ground water at levels that pose concern for human health or the environment under any of the application scenarios evaluated.
- High-end risks from selenium were identified for ecological receptors in smaller headwater streams and for human fishers who consume high quantities of fish caught from larger mainstem streams. These risks were found to be only slightly above relevant health benchmarks. This indicates that the vast majority of modeled application scenarios for FGD gypsum will pose no concerns to human health or the environment. In areas where FGD gypsum could be applied both widely and at high rates, modeled risks can be mitigated through minor limits on the application practices, such as those identified in sensitivity analyses.
- This evaluation could not consider impacts to more smaller and more static water bodies, such as farm fish ponds, due to a lack of information about the associated locations, volumes, drainage areas, or turnover rates. There is an unknown potential for greater accumulation of constituent mass in these water bodies due to the greater residence time of water in these systems. Thus, further consideration may be warranted prior to substantial applications in drainage area of such water bodies.
- This evaluation focused on potential environmental impacts unique to FGD gypsum. However, risks can also result from the mismanagement of other types of gypsum. For example, the high concentrations of sulfate in gypsum may pose risk to cattle that are allowed to graze in fields too soon after application. USDA has developed guidelines to address the risks that are shared among all types of gypsum in *Conservation Practice Standard: Amending Soil Properties with Gypsum Products* (USDA, 2015b). These guidelines identify agronomically relevant application rates intended to ensure the management of agricultural amendments remains protective of human health and the environment.

It has previously been established that there are a number of compelling benefits associated with the use of FGD gypsum in agriculture, such as a providing key nutrients to crops and limiting phosphorus runoff to nearby water bodies. These uses may also provide benefits outside of the fields, such as helping to reduce greenhouse gas emissions from mining, diverting these waste from CCR landfills, and providing cost saving to farmers. Based on these model results, application of FGD gypsum to fields at the agronomically relevant rates considered in this evaluation can provide benefits while remaining protective of human health and the environment.

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Appendix A: Constituent Data

This appendix provides a summary of the collection and management of raw data drawn from the available literature and considered in the beneficial use evaluation of flue gas desulfurization (FGD) gypsum in agricultural applications. The rationale for excluding any literature sources that were considered, but not retained, is discussed. The raw data for those literature sources that were relied upon in the current evaluation can be found in the constituent database.

- **Attachment A-1:** provides a summary of communications between EPA and authors to obtain unreported data or clarify analytical methodology.
- **Attachment A-2:** provides a summary of bulk and leachate concentration data used in the evaluation following the data quality review described in this appendix.

A.1 Data Collection

USDA and EPA reviewed the available literature and assembled those that appeared to contain information on the constituent concentrations present in or released from FGD and mined gypsum. A number of relevant literature sources, in particular grey literature, had already been obtained through previous EPA or USDA investigations. Thus, EPA began with a review of the references cited in these and all subsequently collected studies. EPA also queried Environmental Sciences and Pollution Management, EBSCO HOST, PubMed, Science Direct, Web of Science, and JSTOR for the key words: “gypsum,” “flue gas desulfurization gypsum,” “FGD gypsum,” “mined gypsum,” “natural gypsum,” and “synthetic gypsum.” Although some literature sources used other terms, such as “coal gypsum” or “FGD products,” these studies tended to be older and ambiguous about whether the materials analyzed fit the current definition of FGD gypsum. Because capturing available information on the composition and behavior of gypsum was a primary goal of the literature search, search terms related to the specific beneficial use were not used. The literature search resulted in a total of 199 unique sources, of which 75 were determined to contained potentially relevant information based on preliminary review of abstracts and tables.

A.2 Data Quality Review

EPA reviewed all of the literature sources assembled to ensure that the data drawn from each were of sufficient quality to form the basis for conclusions about the beneficial use of FGD gypsum. The following subsections detail how the Agency applied the data quality assessment factors outlined in *A Summary of General Assessment Factors for Evaluation the Quality of Scientific and Technical Information* (U.S. EPA, 2003). When it was determined that data from a particular literature source were not germane, then those data were removed from the database entirely. When individual data points or entire studies were found to introduce an unacceptable level of uncertainty into the evaluation, these data were filtered out from the dataset prior to analysis. However, these data were retained in the database for reference.

A.2.1 Clarity and Completeness

Clarity and completeness are the degree to which a literature source transparently documents all assumptions, methods, quality assurance protocols, results, and other key information. An evaluation that is both clear and complete provides enough detail that an outside party with access to the proper resources can replicate the analyses. During the review of the assembled literature, EPA found that some authors chose to present summary statistics instead of full datasets. Others did not specify information about the gypsum that, while not the focus of that particular literature source, was of importance to the current beneficial use evaluation. EPA made an effort to reach out to the authors and obtain the missing information. Those who responded did not always have answers to the questions, either because that information was never collected or because the authors no longer had access to the raw data. Although some questions remained unanswered, no data were eliminated because of insufficient clarity or completeness. Instead, the data are presented with the relevant field marked as “unknown” for transparency. A summary of the effort to contact authors and the additional information obtained are provided in **Attachment A-1**.

A.2.2 Evaluation and Review

Evaluation and review is the extent to which a literature source has undergone independent verification, validation and peer review. An independent review is one conducted by objective technical experts who were not associated with the generation of the work under review either directly through substantial contribution to its development, or indirectly through significant consultation during the development of the work. Independent review is intended to identify any errors or bias present in how data are collected, handled or interpreted and also to ensure that the findings are accurate, reliable and unbiased.

The majority of literature sources assembled were drawn from independently peer-reviewed journals; however, a number of grey literature sources were also identified. Some of these sources were data submitted directly to EPA by states and other parties. These data were made available to the public and a panel of independent peer reviewers for comment as part of the development of *Human and Ecological Risk Assessment of Coal Combustion Wastes* (U.S. EPA, 2014). Some of the sources reviewed were grey literature written by EPA and USDA. While these studies were not all submitted for an independent external review, the data were all collected according to detailed sampling and analysis plans that specify relevant QA/QC procedures.

In several cases, data from grey literature sources were later published in peer-reviewed journals. Comparison of the data reported in the two sources identified occasional differences in values between the grey and published sources. EPA reached out to the authors to determine the cause of these apparent discrepancies. Only one author responded to clarify that the grey literature in

EPRI (2008) contained preliminary data that was later updated. In most other cases, data from the grey literature sources were used because they were more complete and any discrepancies identified were minor. In a few cases, individual data points from the later peer-reviewed sources were included because they were not reported in the earlier grey literature. A summary of the studies removed from the database due to duplication of data is presented in **Table A-1**.

Table A-1: Duplicate Sources Filtered from Database

Grey Literature Citation	Journal Citation	Gypsum Type	Media Analyzed
EPRI (2013a), EPRI (2013b)	Norton (2011)	FGD, Mined	Crop, Leachate, Soil
EPRI (2011b)	DeSutter et al. (2011)	FGD	Crop, Soil
EPRI (2011c)	DeSutter et al. (2014)	FGD, Mined	Crop, Soil
EPRI (2012a)	Kost et al. (2014)	FGD, Mined	Crop, Gypsum, Soil
EPRI (2008), EPRI(2012a), EPRI (2013a), and EPRI (2013b)	Chen et al. (2014)	FGD, Mined	Gypsum

A.2.3 Soundness

Soundness is the extent to which the methods employed by a literature source are reasonable and consistent with the intended application of the data. This means that any methods used to collect and measure data have demonstrated the technical ability to reliably and repeatedly achieve desired levels of accuracy and precision, and that any methods used to analyze and interpret data, such as equations, models and simplifying assumptions, are adequately justified and rooted in accepted scientific principles.

Sample Collection Methods:

Some studies did not report the approach used to collect solid soil and gypsum samples. In cases where the samples were provided by the facilities, the author may not have this information. As a result, it is sometimes unclear whether the data represent individual grab samples or composite samples collected over an unspecified area or time interval. Both collection methods are valid and reflect the material under evaluation, but provide somewhat different information. Individual grab samples have the potential to capture isolated "hot spots," while composite samples provide a more typical estimate of the concentrations present either spatially or temporally. The available dataset is known to contain a combination of both types of samples. Combining data points obtained from different collection methods treats them as identical, which will introduce some uncertainty into the evaluation.

This evaluation focuses primarily on chronic exposures that are best captured by data that provide representative values for each source. While individual grab samples may over- or underestimate these values on a case-by-case basis, these samples still reflect actual concentrations in the gypsum. A large fraction of the available data for many constituents are grab samples. Eliminating these samples would greatly reduce the available data and might omit high-end values that ensure the evaluation remains protective. In addition, a comparison of grab samples in the database from the same sources indicates that there is not a high degree of variability among the gypsum produced at a facility at a given time. This may be attributed to the substantial mixing of the slurry that occurs during gypsum production and handling. Therefore, the uncertainty introduced into the evaluation is relatively small and no samples were filtered out due to the sample collection methods used.

Sample Preparation Methods:

EPA reviewed the methods used to prepare samples for analysis reported by each source to determine whether the resulting data were of sufficient quality to incorporate in this evaluation. The purpose of the preparation methods is to convert constituent mass into a soluble form that can be measured by standard analytical instruments. The majority of studies report digestion by a combination of heat and one or more of the following acids: hydrochloric acid, hydrofluoric acid, nitric acid and perchloric acid. Different acids are effective at breaking down different compounds, such as organic matter, iron oxides and silicates. As a result, a combination of acids is often used to maximize dissolution.

FGD gypsum typically composed of 95% or more calcium sulfate, which is a mineral that dissolves readily in water, and less than 1% silica (Henkels and Gaynor, 1995). Therefore, it is unlikely there is a substantial amount of constituent mass retained in recalcitrant minerals. Thus, EPA concluded that combining data with different acids is unlikely to introduce substantial uncertainty into the evaluation. No samples were filtered out due to the sample preparation methods used.

Detection Limits:

A detection limit is the lowest level at which an analytical instrument can reliably differentiate actual constituent concentrations from background noise. When a constituent is not detected above this limit, the analytical results are typically reported as less than the detection limit because the potential still exists for the constituent to be present at a lower level. The detection limit varies among studies because of differences in the methods used to prepare samples, the sensitivity of analytical instruments, and interference from solid media or other chemical constituents.

Non-detect (or “left-censored”) data are typically the lowest values in a dataset. Elimination of these non-detect data may bias the remaining dataset high. Therefore, EPA incorporated all of these data into the constituent database at the detection limit and flagged the values as non-detect in a separate column. However, non-detect values were not always the lowest values reported for some constituents and, in some cases, were the highest. High detection limits introduce a great amount of uncertainty into the evaluation and can bias the overall dataset high. Therefore, EPA filtered out any non-detect values that were greater than the 90th percentile of the remaining, detected data. A summary of the data filtered out from the database prior to analysis due to high detection limits is presented in **Table A-2**.

Table A-2: High Detection Limits Filtered from Database

Source	Gypsum Type	Constituent	Reported Detection Limit	Reported Units
Bryant et al. (2012)	FGD	Beryllium	3.1	mg/kg
	FGD	Cadmium	3.1	mg/kg
	FGD	Lead	3.1	mg/kg
	FGD	Thallium	2.5	mg/kg
Chen et al. (2008)	FGD	Arsenic	11	mg/kg
	FGD	Cadmium	1	mg/kg
	FGD	Lead	5	mg/kg
DeSutter and Cihacek (2009)	Mined	Arsenic	2.6	ug/g
	Mined	Selenium	1.2	ug/g
OSU-E (2005)	Mined	Cadmium	0.48	ppm
	Mined	Selenium	1.45	ppm
EERC (2007)	FGD	Cadmium	1	ug/g
	FGD	Cadmium	1	ug/g
	FGD	Cadmium	1	ug/g
	FGD	Lead	3	ug/g
	FGD	Lead	3	ug/g
	FGD	Lead	3	ug/g
	FGD	Lead	3	ug/g
EPRI (2008)	FGD	Thallium	1.44	ug/g
EPRI (2011a)	Mined	Beryllium	0.1	mg/kg
	Mined	Beryllium	0.1	mg/kg
EPRI (2012b)	Mined	Arsenic	4.21	mg/kg
	Mined	Beryllium	0.18	mg/kg
	FGD	Beryllium	0.18	mg/kg
	Mined	Selenium	4.86	mg/kg
EPRI (2013b)	FGD	Thallium	1.44	mg/kg
	FGD	Thallium	1.44	mg/kg
Schomberg et al. (2018)	FGD	Beryllium	0.16	mg/kg
	FGD	Beryllium	0.13	mg/kg
	FGD	Thallium	1.3	mg/kg
	FGD	Thallium	1.6	mg/kg
Yost et al. (2010)	Mined	Cadmium	0.9	mg/kg
	FGD	Lead	3.4	mg/kg
	FGD	Lead	3.5	mg/kg

A.2.4 Applicability and Utility

Applicability and utility is the extent to which the findings of a literature source are relevant for the intended use. This means that the purpose, design and findings of the data can support a similar set of conclusions when applied to the conceptual model for the beneficial use. EPA reviewed each of the studies collected to ensure that the data contained were representative of the materials used and environmental conditions relevant to the current evaluation.

Country of Origin:

When reviewing the available literature, EPA did not initially screen based on the country in which the gypsum was generated, as this data might be useful later for comparisons. Thus, some data were collected from countries outside the United States. Differences in the composition of the coal burnt and the pollution control technologies used in these countries may result in a trace element composition that does not reflect gypsum generated in the United States. Thus, to ensure data relied upon was representative, EPA filtered out data on gypsums generated outside North America from the constituent database. An exception was made where measurements were not directly related to composition of the gypsum, such as crop uptake from gypsum amended soils. Even if the gypsum used in the literature source had higher levels of trace elements than may be found in the North America, the literature source can still provide useful information about the tendency of plants to accumulate constituents mass based on the concentrations present in surrounding soils. A summary of the data removed from the database due to the country of origin is presented in **Table A-3**.

Table A-3: Foreign Gypsum Filtered from Database

Source	Countries	Gypsum Type	Media
Alvarez-Ayuso et al. (2006)	Spain	FGD	Bulk Concentration
Alvarez-Ayuso and Querol (2007)	Spain	FGD	Bulk Concentration
Alvarez-Ayuso et al. (2011)	Spain	FGD	Bulk Concentration, Leachate
Amezketa et al. (2005)	Spain	FGD, Mined	Bulk Concentration
Berland et al. (2003)	Germany, Japan, United Kingdom	FGD	Bulk Concentration
Rallo et al. (2010)	Spain	FGD	Bulk Concentration
Stergarsek et al. (2008)	Slovenia	FGD	Bulk Concentration
Yodthongdee et al. (2013)	Thailand	FGD	Bulk Concentration

Pelletized Samples:

Some studies measured constituent concentrations present in and released from mined gypsum that had been pelletized. Pelletization is a process that coats the gypsum with a binding solution, such as a cellulose-based polymer (U.S. EPA, 2012). The resulting product is marketed as "easier to transport and apply" and "providing more efficient delivery of nutrients." There was not enough information to determine whether pelletization may contribute additional constituent mass or alter the leaching behavior of mined gypsum. Therefore, EPA filtered out data for the mined gypsum samples that had been pelletized. Samples of FGD gypsum were typically collected directly from utilities and so none had undergone additional processing. A summary of the data filtered out prior to quantitative analysis presented in **Table A-4**.

Table A-4: Pelletized Gypsum Filtered from Database

Source	Gypsum Type	Media
U.S. EPA (2012)	Mined	Bulk Concentration, Leachate
Chen et al. (2014)	Mined	Bulk Concentration
EPRI (2008)	Mined	Bulk Concentration
EPRI (2012a)	Mined	Bulk Concentration, Leachate
EPRI (2013a)	Mined	Bulk Concentration, Leachate
EPRI (2013b)	Mined	Bulk Concentration, Leachate
Kost et al. (2014)	Mined	Bulk Concentration

Materials Other Than Gypsum:

Not all of the studies identified from the literature focused exclusively on gypsum. Several also contained information on other FGD materials or other coal combustion residuals. Because these non-gypsum materials are not the focus of this evaluation, these data were not even considered for the database. However, other studies appeared to measure gypsum, but further inspection revealed that the studies erroneously labeled the material as gypsum or had mixed gypsum with other materials (e.g., chicken litter) prior to analysis. Because these measurements cannot be used to characterize raw gypsum, these data were also removed from the constituent database. A summary of the non-gypsum materials removed from the database is presented in **Table A-4.**

Table A-4: Non-Gypsum Material Filtered from Database

Source	Material Analyzed
Alvarez-Ayuso and Querol (2007)	FGD gypsum treated with aluminum oxide
Chen et al. (2008)	"FGD Product"
Chen et al. (2009)	FGD-$CaSO_3$
EPRI (2008)	FGD containing fly ash
U.S. EPA (2009)	FGD containing fly ash

Analytes and Reported Units:

Environmental contamination was not the primary focus of every literature source and, as a result, many studies reported data on some analytes that were not germane to the current evaluation. Other studies reported data on relevant analytes, but in units that could not be reliably converted into a useable form with the information provided (e.g., mass percent). These data could not be incorporated into a quantitative evaluation and were removed from the constituent database. **Table A-5** presents a summary of the data removed from the database because of unusable analytes or units.

Table A-5: Unsuitable Analytes and Units Filtered from Database

Source	Gypsum Type	Analyte	Units
EERC (2007)	FGD	Calcium oxide, Magnesium oxide, Phosphorus pentoxide, Potassium oxide, Sodium oxide	%
EPRI (2008)	FGD, Mined	Lime test index, Total Neutralizing Potential	various
EPRI (2012a)	FGD, Mined	Lime test index, Total Neutralizing Potential	various
EPRI (2012b)	FGD, Mined	Total Neutralizing Potential	various
EPRI (2013a)	FGD, Mined	Total Neutralizing Potential	various
EPRI (2013b)	FGD, Mined	Calcium carbonate, Lime test index, Nitrogen, Phosphorus	various
Kost et al. (2014)	FGD, Mined	Total Neutralizing Potential	various
Stergarsek et al. (2008)	FGD	Arsenic, Bromine, Mercury, Selenium	mg/ton coal
Stout et al. (1999)	FGD	$CaCO_3$ equivalent	g/kg

Wallboard Samples:

Some data included in the evaluation for total content were gypsum that had been processed into wallboard. Certain steps in the wallboard production process, specifically washing to remove impurities and heated drying, have the potential to alter the constituent composition of the gypsum. Other steps in the production process are unlikely to alter constituent concentrations within the bulk gypsum (EERC, 2003). While additives, such as starch and vermiculite, may be mixed in during these processes, these typically account for less than a half percent of the total product mass and are not anticipated to appreciably alter overall constituent concentrations.

Heated drying and calcination processes expose FGD gypsum to temperatures above 128°C (262°F). These elevated temperatures accelerate the release of mercury, but the fraction that will volatilize depends on a number of factors that include the specific temperature and drying time (NETL, 2008). Because less energy-intensive dewatering processes are available that do not result in elevated releases to the surrounding air; such as hydrocyclones, centrifuges and belt presses (Genck et al., 2008); EPA assumes that gypsum intended for agricultural applications will not be exposed such high temperatures after generation. To determine if inclusion of these wallboard data in the dataset may have biased constituent distributions lower, EPA compared the bulk content distributions with and without these data. EPA found that both the median and high-end concentrations were unchanged after removal of these data. Therefore, the inclusion of these data are unlikely skew model results in this instance.

Mercury Emission Sample Collection:

During the review of the assembled data, EPA identified differences among the mercury emission data that could not be resolved with the information reported in the literature. These discrepancies arose primarily from the disparate methods used to sample for mercury. Some studies measured the total mercury captured by filters over a specified timeframe from which a time-averaged emission rate was calculated, while others used a real-time analyzer to measure the actual emission rate at specified time intervals. Even when samples were collected with similar methods, each literature source reported values averaged over different timeframes. It was not possible to transform and aggregate all of the available data in a meaningful way and, because mercury emissions can be highly variable over time, presenting the data without these caveats in the database runs the risk of being misleading. Therefore, because these data were not relied upon in the evaluation and to avoid confusion from the presentation of these data without proper context, EPA chose to remove these data from the database. A summary of the mercury emission data removed from the database is presented in **Table A-6**.

Table A-6: Mercury Emission Data Filtered from Database

Source	Media
Briggs et al. (2014)	Unamended Soil, Mixed Soil and FGD/Mined Gypsum
Cheng et al. (2012)	FGD Gypsum
Gustin and Ladwig (2010)	FGD Gypsum
Pekney et al. (2009)	FGD Gypsum
Shock et al. (2009)	FGD Gypsum, Mined Gypsum
Wang et al. (2013)	Mixed Soil and FGD Gypsum
Xin et al. (2006)	FGD Gypsum

Sampling Depth:

When reviewing the soil data used to calculate BCFs, EPA identified some soil samples that were collected from depths outside of the typical crop root zone. Crops are unlikely to be exposed to concentrations this deep and so use of these data to calculate BCFs was considered inappropriate. Samples that included depths much greater than 20 cm were excluded. A summary of the data filtered out prior to quantitative analysis presented in **Table A-7**.

Table A-7: Soil Depth Filtered from Database

Source	Media	Depth
DeSutter et al. (2014)	Unamended Soil, FGD Amended Soil, and Mined Gypsum Amended Soil	15 - 30 cm
EPRI (2011c)	Unamended Soil, FGD Amended Soil, and Mined Gypsum Amended Soil	15 - 30 cm
EPRI (2012a)	Unamended Soil, FGD Amended Soil, and Mined Gypsum Amended Soil	15 - 30 cm

Lysimeter Data:

Lysimeters are sampling devices placed under the soil to collect leachate samples that reflect the mixture of gypsum and soil. In theory, such samples can empirically demonstrate how interactions with the soil affect the leaching behavior of gypsum to provide a more accurate estimate of field leaching. However, in practice, available samples introduce too much uncertainty to incorporate into this evaluation:

- The sources identified did not separately test the leaching behavior of the gypsum applied to the soil. Therefore, it is not possible to determine how leachate concentrations changed as a result of mixing.
- Each of the sources reflect only a few soils amended with single type of gypsum. It is not possible to make statements about the representativeness of these samples without additional information.
- The studies only measured leaching over the course of about a year. This could underestimate long-term leaching because constituent mass has not had enough time to fully migrate through the soil column.
- For multiple constituents, all or nearly all of the lysimeter samples collected among these literature sources were non-detect (i.e., aluminum, antimony, arsenic, beryllium, cadmium, cobalt, copper, iron, lead, selenium, thallium). Non-detect data has the potential to yield useful information, but the detection limits were often near or above the highest leachate concentration measured from unmixed FGD gypsum (i.e., antimony, arsenic, lead, thallium).
- The available leachate data from unmixed FGD gypsum demonstrates there is a high potential for depletion of available constituent mass within a year. Because the studies do not specify the amount of precipitation and the infiltration captured in the lysimeter, the magnitude of this dilution cannot be estimated.

Based on these considerations, any conclusions drawn with this data would require substantial caveats. Because these data could not be incorporated into the evaluation, they were filtered out of the constituent database when characterizing leaching behavior. A summary of the data filtered out prior to quantitative analysis presented in **Table A-8**.

Table A-8: Lysimeter Data Filtered from Database

Source	Media
Briggs et al. (2014)	Unamended Soil, Mixed Soil and FGD/Mined Gypsum
EPRI (2012a)	Mixed Soil and FGD/Mined Gypsum
EPRI (2013a)	Unamended Soil, Mixed Soil and FGD/Mined Gypsum
EPRI (2013b)	Unamended Soil, Mixed Soil and FGD/Mined Gypsum
Wang (2012)	Unamended Soil, Mixed Soil and FGD Gypsum

Environmental Factors:

EPA reviewed the assembled literature to determine whether each source reported sufficient information about the environmental factors that control releases into each media. Information on these factors allows EPA to appropriately apply the data to corresponding field conditions. EPA identified the key properties through a review of the literature. Some have been well established in the literature, while others are suspected and are the topic of continued investigation.

The soil pH and composition (e.g., clay), as well as the type of plant grown, were identified as the key factors most likely to influence plant uptake of inorganics (ORNL, 1984; U.S. EPA, 1999). When available, this information was assembled in the database along with the reported soil and plant concentrations. All studies reported information on the type of plant grown, but some did not report soil pH and/or composition. EPA did not remove any samples from the database when this information was not available because it was still known that the samples represent conditions under which the crops can be grown. There was no indication that any of the studies took steps to alter the soil in ways that would result in unrealistic conditions. Therefore, no data were removed from the database due to missing information about soil pH or composition.

The liquid to solid (L/S) ratio and the equilibrium pH of leachate samples were identified as two key factors most likely to influence the leaching behavior of inorganics (U.S EPA, 2010). When available, this information was assembled in the database along with the leachate concentrations. The L/S ratio is specified by most leaching tests. Therefore, even when this information was not explicitly reported, it could be easily inferred from the test used. Although many tests also specify the initial pH of the eluent, some do not specify a final, equilibrium pH of the solution. This is because these extraction tests are designed to assess the behavior of materials exposed to specific inputs (e.g., acid rain). The final equilibrium pH can vary dramatically from the initial pH based on the chemistry of the material tested. Leachate concentrations can vary by orders of magnitude with incremental changes in pH. Thus, if the final pH of a sample was unknown, the samples introduce too much uncertainty and were entirely removed from the database. If the final pH was recorded but fell outside the relevant pH range of between 5 and 8, then the data were retained in the database but filtered out prior to quantitative analysis. **Table A-9** presents a summary of the data removed because of unknown or unsuitable environmental conditions.

Table A-9: Incomplete Environmental Data Filtered from Database

Source	Gypsum Type	Media	Missing Information
Bryant et al. (2012)	FGD	Leachate	No final pH
Pasini and Walker (2012)	FGD	Leachate	No final pH
Xin et al. (2006)	FGD	Leachate	No final pH

A.2.5 Variability and Uncertainty

Variability and uncertainty are the extent to which a literature source effectively characterizes, either quantitatively or qualitatively, these two factors in the information relied upon and in the procedures, measures, methods or models utilized. Proper characterization of the major sources of variability and uncertainty provides greater confidence that the data can form the basis for sound conclusions in the beneficial use evaluation.

Each individual literature source provides raw data on only a few samples and so is unlikely to fully capture the variability of constituent concentrations present in and released from gypsum. To address this fact, this beneficial use evaluation aggregated data from all of the available sources found to otherwise be of sufficient quality. Because more data ensures better characterization of the gypsum, there is no reason to exclude any individual sample because it alone does not fully capture variability.

Each literature source provides raw data on constituent concentrations present in or released from gypsum. The methods used to measure these concentrations were found to be sound, so there is minimal uncertainty associated with the detected values reported in any of the literature sources. Therefore, no data were eliminated as a result of uncertainties about the specific values reported.

A.3 Data Management

The data found to be of sufficient quality were incorporated into the database as reported in each literature source. Once all of the data were assembled, additional management steps were taken prior to use of the data in any quantitative analyses to ensure the dataset was not biased towards a single samples or source. To do this, EPA identified and combined replicate and duplicate samples.

Replicate samples are two or more measurements of the same sample, often collected for QA/QC purposes. These replicate samples are typically averaged into a single data point to obtain a representative value for the sample. Some literature sources only report the mean of the replicates and the corresponding standard deviation. EPA did not make an attempt to reach out and obtain values for each individual replicate from these sources because the same average value would have been calculated for this evaluation prior to use in any quantitative analyses.

Duplicate samples are two or more field samples intended to represent the same source, which are collected and analyzed in an identical manner. For a number of reasons, such as heterogeneity of the source material and sensitivity of the analytical equipment, values measured for these samples may not be identical. This evaluation averaged duplicate samples to obtain a more representative value for that source. EPA chose to treat any samples collected from the same facility as duplicates, regardless of whether the samples were collected as part of separate studies or at different times. This was done to avoid biasing the overall data set towards facilities that had been more heavily sampled. Each literature source had a unique approach to labeling samples that sometimes made it difficult to identify the facility associated with a sample. There was often enough other information presented, such as the geographic location of the source, to determine whether samples were duplicates. However, it is possible some that some duplicate sample remain.

To prepare duplicate and replicate samples for quantitative analysis, EPA first averaged all replicates for a single sample and then all duplicates from a single source. Where duplicates and replicates were a mixture of detect and non-detect values, the non-detect values were set to the detection limit. Because the constituent was detected in other duplicated or replicates, it was assumed that the detection limit would provide the best, conservative estimate of the true value. The resulting averaged value was flagged as detected for the quantitative analysis.

A.4 Summary of Available Data

Table A-10 presents a summary of the available data for FGD gypsum bulk concentration and leachate available to characterize each of the release pathways, after filtering for data quality and managing replicate and duplicate samples. Both bulk concentration and leachate data were necessary to provide realistic estimates of potential exposures. Constituents with insufficient data to characterize bulk concentration (e.g., bromide, tin), leachate (e.g., fluoride, lithium, titanium), or both (e.g., silver) were not retained for full evaluation. The uncertainties introduced due to this lack of data are discussed as further in **Section 7 (Risk Characterization)**. Constituents known to be macronutrients for plants and animals (i.e., calcium, carbon, magnesium, nitrogen, potassium, sulfur/sulfate) were not retained for evaluation and are not discussed further in this document. Contributions of these nutrients from applied FGD gypsum should be factored into relevant nutrient management plans. A summary of the concentration data used in the evaluation is presented in **Attachment A-2**.

Table A-10: Summary of Filtered Constituent Data

Constituent	CASRN	Bulk Concentration Detected / Total Samples			Leachate Detected / Total Samples		
		Unwashed	Washed	Unknown	Unwashed	Washed	Unknown
Aluminum	7429-90-5	21 / 21	29 / 29	11 / 11	37 / 39	37 / 38	4 / 7
Antimony	7440-36-0	21 / 21	29 / 31	8 / 10	8 / 39	11 / 39	7 / 7
Arsenic	7440-38-2	27 / 29	31 / 33	6 / 16	11 / 46	5 / 53	0 / 10
Barium	7440-39-3	21 / 21	31 / 31	11 / 11	46 / 46	53 / 53	10 / 10
Beryllium	7440-41-7	7 / 10	13 / 21	3 / 10	1 / 39	5 / 39	2 / 7
Bismuth	7440-69-9	0 / 0	0 / 2	0 / 0	0 / 0	0 / 0	0 / 0
Bromide	7726-95-6	0 / 0	0 / 0	0 / 0	5 / 19	0 / 23	0 / 0
Boron	7440-42-8	11 / 11	19 / 20	10 / 12	39 / 39	31 / 39	4 / 7
Cadmium	7440-43-9	21 / 21	29 / 31	8 / 16	24 / 45	18 / 53	3 / 10
Calcium	7440-70-2	20 / 20	29 / 29	12 / 12	39 / 39	39 /39	7 / 7
Carbon	7440-44-0	11 / 11	9 / 9	0 / 0	0 / 0	0 / 0	0 / 0
Chloride	16887-00-6	18 / 18	14 / 14	0 / 1	22 / 22	10 / 23	0 / 0
Chromium	7440-47-3	21 / 21	31 / 31	13 / 16	30 / 45	32 / 53	4 / 10
Cobalt	7440-48-4	20 / 21	24 / 31	4 / 11	25 / 39	18 / 39	4 / 7
Copper	7440-50-8	14 / 16	18 / 22	7 / 13	24 / 39	25 / 39	4 / 7
Fluoride	16984-48-8	5 / 11	4 / 9	0 / 0	0 / 0	0 / 0	0 / 0
Iron	7439-89-6	28 / 28	29 / 29	16 / 16	27 / 39	24 / 39	6 / 7
Lead	7439-92-1	20 / 21	25 / 32	8 / 16	7 / 46	13 / 53	4 / 10
Lithium	7439-93-2	10 / 10	19 / 19	7 / 9	0 / 0	0 / 0	3 / 3
Magnesium	7439-95-4	21 / 21	29 / 29	13 / 13	39 /39	39 / 39	7 / 7
Manganese	7439-96-5	10 / 10	20 / 20	12 / 13	37 / 38	37 / 39	7 / 7
Mercury	7439-97-6	35 / 35	51 / 51	20 / 22	30 / 52	41 / 57	7 / 8
Molybdenum	7439-98-7	21 / 21	30 / 31	9 / 12	24 / 39	26 / 39	7 / 7
Nickel	7440-02-0	10 / 10	21 / 22	11 / 16	37 / 39	34 / 39	6 / 7

Table A-10: Summary of Filtered Constituent Data

Constituent	CASRN	Bulk Concentration Detected / Total Samples			Leachate Detected / Total Samples		
		Unwashed	Washed	Unknown	Unwashed	Washed	Unknown
Potassium	7440-09-7	20 / 21	27 / 29	11 / 12	2 / 2	10 / 10	4 / 7
Selenium	7782-49-2	29 / 29	32 / 32	11 / 16	43 / 46	41 / 53	4 / 10
Silicon	7440-21-3	21 / 21	29 / 29	8 / 8	0 / 0	0 / 0	3 / 3
Silver	7440-22-4	0 / 1	0 / 2	0 / 2	0 / 7	0 / 14	0 / 3
Sulfate	14808-79-8	0 / 0	0 / 0	1 / 1	22 / 22	23 /23	0 / 0
Sodium	7440-23-5	20 / 21	22/ 29	6 / 10	2 / 2	10 / 10	7 / 7
Strontium	7440-24-6	21 / 21	29 / 29	8 / 8	39 / 39	39 / 39	7 / 7
Tin	7440-31-5	0 / 0	0 / 0	0 / 0	0 / 36	1 / 29	0 / 0
Titanium	7440-32-6	3 / 11	3 / 9	0 / 0	0 / 0	0 / 0	0 / 0
Thallium	7440-28-0	19 / 19	28 / 30	4 / 8	18 / 39	7 / 39	6 / 7
Uranium	7440-61-1	0 / 0	2 / 2	0 / 0	0 / 0	0 / 0	0 / 0
Vanadium	7440-62-2	10 / 10	21 / 22	8 / 10	31 / 39	26 / 39	3 / 7
Zinc	7440-66-6	10 / 10	22 / 22	12 / 13	39 / 39	39 / 39	6 / 7

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EPRI. 2011c. “Flue Gas Desulfurization Gypsum Agricultural Network: North Dakota Sites 1 and 2 (Wheat).” Prepared by W. Dick and D. Kost of Ohio State University and T. DeSutter and L. Cihacek of North Dakota State University. Palo Alto, CA. December.

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EPRI 2012b. “Flue Gas Desulfurization Gypsum Agricultural Network: Ohio Sites 1 (Mixed Hay) and 2 (Corn).” Prepared by W. Dick and D. Kost of Ohio State University and D. Smeal of New Mexico State University. Palo Alto, CA. October.

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Attachment A-1. Summary of Communications

<table>
<tr><th>Date</th><th>Contact Name</th><th>Affiliation and Title</th></tr>
<tr><td>4/10/15</td><td>Ardeshir Adeli</td><td>USDA, Soil Scientist</td></tr>
<tr><td colspan="3">Question/Request: Were the gypsum samples in the Sheng et al. (2014) literature source washed or unwashed?
Summary of Response: A full load truck of FGD gypsum was delivered from the power plant either from Georgia or Alabama; contact was unsure of the process related to the FGD gypsum.</td></tr>
<tr><td>4/10/15</td><td>Dr. Souhail Al-Abed</td><td>U.S. EPA, Research Chemist</td></tr>
<tr><td colspan="3">Question/Request: Were the gypsum samples in the Al-Abed et al. (2008) literature source washed or unwashed?
Summary of Response: The gypsum was not washed.</td></tr>
<tr><td>4/10/15</td><td>Ashok Alva</td><td>U.S. EPA, Research Soil Scientist</td></tr>
<tr><td colspan="3">Question/Request: Were the gypsum samples in the Alva et al. (1998a,b) literature source washed or unwashed?
Summary of Response: The FGD gypsum used in the literature source was a byproduct from scrubber from a power plant near Tampa, FL. The sample was collected from the bulk storage landfill near the plant and there were no notes on what was done to the FGD gypsum before being sent to storage. The samples underwent no pretreatment before being used in the experiment.</td></tr>
<tr><td>4/10/15</td><td>Esther Alvarez-Ayuso</td><td>Instituto de Recursos Naturales y Agrobiología de Salamanca, Research Associate</td></tr>
<tr><td colspan="3">Question/Request: Were the gypsum samples in the following studies washed or unwashed?
▪ Alvarez-Ayuso et al. (2006)
▪ Alvarez-Ayuso and Querol (2007)
▪ Alvarez-Ayuso et al. (2008a)
▪ Alvarez-Ayuso et al. (2008b)
▪ Alvarez-Ayuso et al. (2011)
Summary of Response: No response received.</td></tr>
<tr><td>4/10/15</td><td>M. Antonia Lopez-Anton</td><td>Instituto Nacional Del Carbón, Professor Chemical Processes in Energy and Environment</td></tr>
<tr><td colspan="3">Question/Request: Were the gypsum samples in the Rallo et al. (2010) literature source washed or unwashed?
Summary of Response: The gypsums used in the literature source were taken directly from the combustion plant as obtained from the wet desulfurization system, unwashed.</td></tr>
<tr><td>4/10/15</td><td>Virupax Baligar</td><td>USDA, Research Soil Scientist</td></tr>
<tr><td colspan="3">Question/Request: Were the gypsum samples in the Clark et al. (2001) literature source washed or unwashed?
Summary of Response: The FGD was unwashed, as the materials used had been collected at the laboratory. They did not go to the bottled gypsum one gets from a chemical distributor, they just used FGD materials.</td></tr>
<tr><td>4/10/15</td><td>Dr. Candace Kairies-Beatty</td><td>Winona State University, Assistant Professor</td></tr>
<tr><td colspan="3">Question/Request: Were the gypsum samples in the Kairies et al. (2006) literature source washed or unwashed?
Summary of Response: No response received.</td></tr>
</table>

<table>
<tr><th>Date</th><th>Contact Name</th><th>Affiliation and Title</th></tr>
<tr><td>4/9/15</td><td>Dr. Ray Bryant</td><td>USDA, Research Soil Scientist</td></tr>
<tr><td colspan="3">Question/Request: Were the gypsum samples in the Bryant et al. (2012) literature source washed or unwashed?
Summary of Response: The partner for the experiment was Constellation Energy (now Raven Power) plant in Brandon Shores, MD. The plant came on line in 2010 with modern technology. They use a forced oxidation wet scrubber system after the removal of fly ash. The ditch filter was constructed in 2007 prior to their plant coming on line. Constellation Energy provided the FGD gypsum from a plant that used a scrubbing process similar to what they were building at the time, so it should fit this same description. The samples were used as it was delivered and were not washed.</td></tr>
<tr><td>4/10/15</td><td>Liming Chen</td><td>The Ohio State University, Research Associate</td></tr>
<tr><td colspan="3">Question/Request: In Chen et al. (2008) data in Table 2 overlap with data in Chen et al. (2009), but the sulfur values differ between the two sources, is there an explanation?
In Chen et al. (2014) data in Table 3 overlap with data in EPRI (2013), but some values for barium and molybdenum differ between the two sources, is there an explanation?
Were the gypsum samples in the following studies washed or unwashed?
▪ Chen et al. (2008)
▪ Chen et al. (2009)
▪ Chen et al. (2014)
▪ OSU-E (2011)
▪ Tubail et al. (2008)
Summary of Response: No response received.</td></tr>
<tr><td>4/10/15</td><td>Chin-Min Cheng</td><td>The Ohio State University, Senior Research Associate</td></tr>
<tr><td colspan="3">Question/Request: Requested measurement data behind Figures 2-5 and Table 3 from Cheng et al. (2012). Additional requests:
1) Table 1 in the 2012 article indicates that the data for the AFO-gypsum is from your 2009 article (Table 8, I assume) but the data do not appear to match up between the two sources. Could you please explain and indicate which source would be the best to use?
2) Figures 2 through 6 and Table 3 in the 2012 article appear to be based on measurements of air, water, soil, and crops that would be of great interest to our EPA client for incorporation in their database. Would you be able to supply the actual individual measurements behind these exhibits in a spreadsheet or other tabular format?
3) Were the gypsum samples in the literature source washed or unwashed?
Summary of Response: Attached the data for the Figures 2-6 and provided the data for Table 3 later. In response to the first question, the data shown in the 2012 paper were from a separate analysis. The statement is not accurate. They actually took another aliquot of AFO-Gypsum sample and analyzed the chemical composition with other two FGD samples before the experiment, Cheng attached the data in the spreadsheet. The plant did not use centrifuge in its process when the literature source was carried out. The gravity belt thickener received gypsum slurry directly from the underflow of hydroclone. The sample was collected from the disposal end. So, the sample was unwashed.</td></tr>
<tr><td>4/10/15</td><td>Dr. Thomas DeSutter</td><td>North Dakota State University, Associate Professor</td></tr>
<tr><td colspan="3">Question/Request: DeSutter et al. (2014) and EPRI (2011d) overlap, but the Table 3 data in DeSutter et al. (2014) do not seem to match any in EPRI (2011d), would like to verify that they are truly different.
Were the gypsum samples in the DeSutter and Cihacek (2009) literature source washed or unwashed?
Summary of Response: Dr. DeSutter attached two documents. On page 305 of the article you will find: “Seed mass was corrected to 13.5% moisture before reporting.” The EPRI article was not corrected for this. They did not clean the samples with water or any other solvent, the FGD was received from Muscatine in 55 gal steel drums.</td></tr>
</table>

Date	Contact Name	Affiliation and Title
2/05/15	**Dr. Warren A. Dick**	**Ohio State University, Professor of Soil Science**

Question/Request: There are discrepancies in the pre-treatment data between the 2008 progress report (Tables 4-3 and 4-4) and the 2011 North Dakota Sites 1 and 2 (Wheat) report (Table 2-1). Specifically, while the Mehlich 3 data match, the 2011 pre-treatment data for EPA method 3051a do not exactly match those of the 2008 tables, for both the Gary and Wayne sites. Another concern with the EPA3051a data is that the second row (constituents As-V) appears to have the exact same data between the Gary and Wayne sites for all scenarios.

Summary of Response: Two Excel files were provided that contained the corrected data. Table 4.3 was correct Table 4.4 was provided with corrected data. Data were transferred properly to Table 4.4 from computer files.

Concerning Table 2-1 (also sent as an attachment), as far as OSU could ascertain the response is as follows. The EPA 3051a numbers in the table, not in parentheses, are from Tom DeSutter analyses except for Hg. The EPA 3051a numbers in parentheses were from the same sites and the same sample, but were analyzed by STAR Laboratory at OSU. For the most part, the values compare very well. The OSU data (in parentheses) are also the same data as in Table 4.3 and in corrected Table 4.4. The Hg values from STAR Laboratory for Sites 1 and 2 were reversed and this has been corrected in the revised Table 2-1.

Date	Contact Name	Affiliation and Title
2/17/15	**Dr. Warren A. Dick**	**Ohio State University, Professor of Soil Science**

Question/Request:

Issue 1. What are the site names associated with the “Site 1” and “Site 2” designations in EPRI 2011 Table 2-1? Comparing the data values in EPRI 2011 Table 2-1 to those in EPRI 2008 Tables 4-3 and 4-4, as well as EPRI 2011 Tables 3-4 and 3-5, the site names for the designations are not consistent, as follows:

- EPRI 2008 site 1 = Wayne
- EPRI 2011 site 1= Gary
- EPRI 2008 site 2 = Gary
- EPRI 2011 site 2 = Wayne
- EPRI 2008 Table 4-3 (Gary) replicates data found in EPRI 2011 Tables 3-4 and 2-1, but matches Site 1 data in Table 3-4 and Site 2 data in Table 2-1
- EPRI 2008 Table 4-4 (Wayne) replicates data found in EPRI 2011 tables 3-5 and 2-1, but matches site 2 data in Table 3-5 and site 1 data in Table 2-1

Issue 2. As shown below, there are a few data discrepancies between the 2008 and 2011 reports; which is the correct value in each case?

- EPRI 2008 Table 4-3: EPA3051a value for Check 0 Sr is 26.98, but the value from 2011 Table 3-4 EPA 3051a is 30.0. Is this rounding?
- EPRI 2008 Table 4-4: EPA3051a value for Check 0 P is 311.5, but the value from 2011 Table 3-5 EPA 3051a is 31.5. 311.5 is more in line with the other measurements – is this the correct value?”
- EPRI 2008 Table 4-4: EPA3051a value for Check 0 S is 269, but the value from 2011 Table 3-5 EPA 3051a is 6269. We suspect that 269 is correct for both because of the zero application rate. Is this correct?

Summary of Response: A file was provided with responses. It was stated that Gary and Wayne were used as site descriptors. The researchers did not recall how the descriptors were changed to Site 1 and Site 2 and then there was a mix-up.

<table>
<tr><th>Date</th><th>Contact Name</th><th>Affiliation and Title</th></tr>
<tr><td>2/20/15</td><td>Dr. Warren A. Dick</td><td>Ohio State University, Professor of Soil Science</td></tr>
<tr><td colspan="3">Question/Request: Tables 2-5 and 2-6 in EPRI (2012c) - FGD gypsum Ag Network NMexico 1 Alfalfa and 2 Sodic Soils [1025355].pdf are missing headers for the second set of results (see attached Word file). Could you supply [headers] or let us know if they should be the same as in Table 2-7.
Also, there are all zeros in a few tables here and there for non-detects and apparent detections that appear to be truncations (e.g., "<0.000" or "0.000").
What is the best way to deal with these? (Some may be other authors – we can contact them)
Summary of Response: Dr. Warren provided two tables submitted to EPRI and that included all of the column headings for Table 2-5 and Table 2-6. The numbering of the tables, themselves, is slightly different due to changes made by EPRI when they reformatted the report for publication. The last page of the attached file showed the detection limits that were mostly used in the reports. These detection limits were followed as much as possible in their reports to give some consistency among the reports as the day-to-day detection limits did vary in a very slight way. Trying to come up with values of samples being analyzed on different days near these detection limits causes problems in reporting and so the Table A-2 (or in some reports Table B-2) was used. If there are any questions about detection limits, refer to the Table A-2. This would include the question about the Mn value in Table A-10 from the 2013 EPRI Report where Mn should be reported as <0.001 mg/L.
The EPRI Table 2011b is not the work of Ohio State University. Dr. Warren believes that the data are from leaching measurements done for Ken Ladwig. Dr. Warren also reported that Soils Data 2009 2011 FGD Gypsum Watkinsville Data for EPA.xlsx, worksheet RSP09_11 Soil Rufus gkg, is not their work. He believes that these data are from USDA-ARS in Watkinsville, GA. The other detection limits or values listed are also from other reports that were not from Ohio State as they did not do any analyses of samples from Watkinsville.</td></tr>
<tr><td>4/13/15</td><td>Dr. Warren A. Dick</td><td>Ohio State University, Professor of Soil Science</td></tr>
<tr><td colspan="3">Question/Request: Kost et al. (2014) and EPRI (2012b) overlap, but Table 1 data in Kost et al. (2014) do not all match those in EPRI (2012b) Table 3-1, is there an explanation? Were the gypsum samples in the Kost et al. (2014) literature source washed or unwashed?
Summary of Response: No response received.</td></tr>
<tr><td>5/28/15</td><td>Dr. Warren A. Dick</td><td>Ohio State University, Professor of Soil Science</td></tr>
<tr><td colspan="3">Question/Request: Specifically, in Ohio State Extension Bulletin 945 (Table 1-3), Arsenic in mined gypsum is reported as 462 ppm. However, the primary data source cited (Dontsova et al 2005, Table 4) reports Arsenic in natural gypsum as < 0.52 ppm. Was there some type of conversion performed that was not documented or was it recorded incorrectly?
Summary of Response: Table 1-3 in the OSU Extension Bulletin is not properly created. The data in Table 1-3 are not from Dontsova et al, 2005. The FGD gypsum data came from a 2005 Agronomy Journal publication (attached publication). The source of this gypsum is given in the publication. The arsenic number seems extremely high but unfortunately there are no samples archived from that work. Dr. Dick assumed that all of the As numbers from the Agronomy Journal article were originally in ug/kg and somehow they got converted to mg/kg without moving the decimal. If that is the case, the values would be 0.119, 0.363 and 0.462 mg/kg. These values are much more in line with anything they have ever analyzed. The FGD gypsums values in Table 1-3 seem to be a composite of values from various samples, but he only had values associated with this particular table (i.e., Table 1-3).</td></tr>
<tr><td>4/10/15</td><td>Dinku Endale</td><td>USDA, Agricultural Engineer</td></tr>
<tr><td colspan="3">Question/Request: Were the gypsum samples in the literature source washed or unwashed?
Summary of Response: The FGD gypsum used came directly from Duke Energy's Marshall Steam Station, in Terrell, North Carolina, mid-March 2009, and stock piled under cover on site. The samples were applied directly to the plots each year from this pile. Table 1 of the paper gives some nutrient content values. They were unsure of washing during the manufacturing process and suggested contacting Duke Energy to try and find out. Their understanding was that the FGD gypsum was high grade.</td></tr>
</table>

Date	Contact Name	Affiliation and Title
4/10/15	**James Grichar**	**Texas A&M University, Senior Research Scientist**
Question/Request: Table 3 units for yield are given as “%”; are these the correct units? Were the gypsum samples in the Grichar et al. (2002) literature source washed or unwashed? **Summary of Response:** No, should be lbs/acre. The gypsum was unwashed.		
4/10/15	**Dr. Mae Gustin**	**University of Nevada, Reno, Prof., Environmental and Resource Sciences**
Multiple questions posed via a single email. **Question/Request 1:** Requested the data behind the figures in Gustin and Ladwig (2010). Were the gypsum samples in the Gustin and Ladwig (2010) literature source washed or unwashed? **Summary of Response 1:** Provided an Excel spreadsheet with the mercury data for Figure 1 but not for other figures. Gustin stated that she has all the raw data files, but cannot just turn them over, because they were paid for by EPRI and she will need to get verification from them before she can provide them. The raw data files also do not have the same labels as in the paper. Thus, this will take time for her to organize. If RTI needs to use the graphs in the paper then they can get copyright permission from EST. She also stated that it would also take her several hours to get the raw data in a format that RTI could interpret given the plant labels in the paper are different than those in the raw data files, and she did not have time to do this. Table 1 of Gustin and Ladwig (2010) states specifically whether the gypsum was washed or not. **Question/Request 2:** For Briggs et al. (2014), are the zero values in Table 1 for Methylmercury in Alabama FGD gypsum and Ohio FGD gypsum correct? What was the detection limit that was used? Were the gypsum samples in the Briggs et al. (2014) literature source washed or unwashed? **Summary of Response 2:** The values were 0 based on our detection limit. “Methylmercury concentrations in FGD gypsum materials were 1.1 ± 0.1 ng/g for Indiana FGD gypsum and below detection for the OH and AL FGD gypsum and for the GYP. Clarification email received on 4/13/2015 stating that the Indiana and Ohio FGD gypsums were washed, and the Alabama gypsum was unwashed. **Question/Request 3:** Were the gypsum samples in the Xin et al. (2006) literature source washed or unwashed? **Summary of Response 3:** No response received.		
4/10/15	**Loreal Heebink**	**Energy and Environmental Research Center, Research Chemist**
Question/Request: Were the gypsum samples in the EERC (2003) literature source washed or unwashed? Were the gypsum samples in the EERC (2007) literature source washed or unwashed? **Summary of Response:** In regards to EERC (2003), the results shown in “Review of handling and use of FGD material” were taken from literature and the author does not remember any of the references specifying whether the material was washed. No response received for the EERC (2007) question.		
4/10/15	**Dr. Milena Horvat**	**Jožef Stefan Institute, Department Head, Environmental Sciences**
Question/Request: Were the gypsum samples in the Stergarsek et al. (2008) literature source washed or unwashed? **Summary of Response:** No response received.		

<table>
<tr><th>Date</th><th>Contact Name</th><th>Affiliation and Title</th></tr>
<tr><td>2/05/15</td><td>Ken Ladwig</td><td>EPRI</td></tr>
<tr><td colspan="3">Question/Request: When we have questions about data in some of the EPRI FGD/Ag Network reports should I contact you, or are you comfortable with us going to the report authors?
Were the gypsum samples in the EPRI studies washed or unwashed?
Summary of Response: Permission provided to go directly to Warren Dick of Ohio State University.
EPRI (2008): This was preliminary data for the sites which eventually had full reports. You should refer to the full reports rather than these data.
▪ EPRI (2011c): Unwashed
▪ EPRI (2011d): Unwashed
▪ EPRI (2012b): Washed
▪ EPRI (2012c): Unwashed
▪ EPRI (2013): Washed
▪ EPRI (2013f): Washed</td></tr>
<tr><td>6/30/15</td><td>Ken Ladwig</td><td>EPRI</td></tr>
<tr><td colspan="3">Question/Request: Requested raw data behind Table 2 and Figure 3 in Gustin and Ladwig (2010) for Plants B, C, and G.
Summary of Response: Attached six Excel spreadsheets with the raw data for Plants B, C, and G.</td></tr>
<tr><td>4/10/15</td><td>Somchai Lapanantnoppakhun</td><td>Chiang Mai University, Prof., Department of Chemistry</td></tr>
<tr><td colspan="3">Question/Request: Were the gypsum samples in the Yodthongdee et al. (2013) literature source washed or unwashed?
Summary of Response: Used unwashed gypsum for the literature source.</td></tr>
<tr><td>4/10/15</td><td>Dr. Joo-Youp Lee</td><td>University of Cincinnati, Chemical Engineering Asst. Professor</td></tr>
<tr><td colspan="3">Question/Request: In Lee et al. (2009), what were the detection limits for the BDL values in Table 2?
Summary of Response: No response received.</td></tr>
<tr><td>4/10/15</td><td>Daniel McChesney</td><td>USDA, Soil Scientist</td></tr>
<tr><td colspan="3">Question/Request: Were the gypsum samples in the Rhoton et al. (2011) literature source washed or unwashed?
Summary of Response: No response received.</td></tr>
<tr><td>4/10/15</td><td>Charles Miller</td><td>U.S. DOE, National Energy Technology Laboratory</td></tr>
<tr><td colspan="3">Question/Request: Were the gypsum samples in the NETL (2005) literature source washed or unwashed?
Summary of Response: Could not answer the question with certainty. The gypsum used for the testing was taken out of USG’s inventory of feedstock at the wallboard plant, and that the wallboard met USG’s acceptance criteria. However, those acceptance criteria include a specification on chloride concentrations in the feedstock, requires efficient washing of the gypsum to meet wallboard feedstock specifications.
For example, typical specs are around 100 ppm chloride max and 10% free moisture. If the cake is unwashed, this 10% moisture would be FGD liquor, so the FGD liquor would be limited to only 1000 ppm chloride to meet that spec. Mr. Miller felt certain that the FGD system that provided the gypsum for Wallboard Test 1 operates with a considerably higher chloride concentration than 1000 ppm. Therefore, he was relatively certain this gypsum was washed as it was dewatered.</td></tr>
<tr><td>4/10/15</td><td>Dr. Darrell Norton</td><td>Purdue University, Professor (Emeritus)</td></tr>
<tr><td colspan="3">Question/Request: Norton (2011) and EPRI (2013f) overlap, but there are data discrepancies between the two data sources, is there an explanation?
Summary of Response: No response received.</td></tr>
</table>

<table>
<tr><th>Date</th><th>Contact Name</th><th>Affiliation and Title</th></tr>
<tr><td>4/10/15</td><td>Natalie Pekney</td><td>U.S. DOE National Energy Technology Laboratory</td></tr>
<tr><td colspan="3">Question/Request: Were the gypsum samples in the Pekney et al. (2009) literature source washed or unwashed?
Summary of Response: Pekney was unable to obtain the information. To avoid bias she was not informed of the source, or specific power plant, from which the samples came.</td></tr>
<tr><td>4/10/15</td><td>Dr. Andrew Sharpley</td><td>University of Arkansas, Professor</td></tr>
<tr><td colspan="3">Question/Request: Were the gypsum samples in the Stout et al. (1999) literature source washed or unwashed?
Summary of Response: Did not know the answer to the question, other than they did not wash it and it would have had to have been at the point of generation. Unable to find out any more information because the senior author of this research and person who designed the literature source and got the material, passed away in 2001, and the last two authors retired about 14 years ago and have had no ongoing contact with the research community.</td></tr>
<tr><td>4/10/15</td><td>Allen Torbert</td><td>USDA, Supervisory Soil Scientist</td></tr>
<tr><td colspan="3">Question/Request: Verify the detection limits for the “ND” results in Table 2 for Gypsum, B, Cu, Mn of the Torbert and Watts (2014) literature source.
Were the gypsum samples in the Torbert and Watts (2014) literature source washed or unwashed?
Summary of Response: The detection limits used were B <50 mg/kg; Cu < 0.8 mg/kg; and Mn < 2 mg/kg. The gypsum used in this literature source was washed.</td></tr>
<tr><td>4/10/15</td><td>Kelin Wang</td><td>Louisiana State University, Graduate Research Assistant</td></tr>
<tr><td colspan="3">Question/Request: For Wang (2012), verify that for Table 27 control value given as 0 ppb; Table 28 control value given as 0 ug. For Wang et al. (2013), provide the detection limits for Table 6.
Summary of Response: They are correct. She did not detect the mercury in control chamber. They are below the detection limit. The detection limit is 0.1 ppb. The detection limit of table 27 and 28 is 0.1 ppb.</td></tr>
<tr><td>4/10/15</td><td>Dr. Harold Walker</td><td>Stony Brook University, Director, Civil Engineering Program</td></tr>
<tr><td colspan="3">Question/Request: Were the gypsum samples in the Pasini and Walker (2012) literature source washed or unwashed?
Summary of Response: They did not perform any washing procedure prior to use in leaching experiments and are not aware of the utility washing the material either. It was their understanding that the material they received was consistent with the material being disposed of in the landfill.</td></tr>
<tr><td>2/12/15</td><td>Dr. Lisa Yost</td><td>Environ (formerly at Exponent)</td></tr>
<tr><td colspan="3">Question/Request: We are collecting compositional data on natural and FGD (synthetic) gypsum for the U.S. EPA. One of our data sources is your 2010 article published in Human and Ecological Risk Assessment, “Lack of complete exposure pathways for metals in natural and FGD gypsum”. In Tables 1 and 2 you provide compositional data for natural and synthetic gypsum, respectively. However the results are presented as statistics (e.g., means, min, max) of multiple samples. Would it be possible to obtain the individual sample results for the data in Tables 1 and 2? If so, a spreadsheet format would be ideal, although we can use any tabular format. Along with the constituent concentrations, we also are interested in the detection limits for any non-detects and any sample identification information (e.g., location, site name, sample ID, plant, process, dates) you can provide so we can properly distinguish the samples in the database.
We also had a question on the analytical methods. What procedure (e.g., U.S. EPA Method 3051, or 3051A) was used to digest/solubilize the sample prior to sample analysis?
Summary of Response: Dr. Yost replied that she was the author and would like to help but she needs her client’s permission which is complicated by the fact that the client is no longer at that the previous job. However, she stated that she would give it a try. (On 3/19/2015, Dr. Yost suggested a call. RTI and EPA suggested dates and times for a call but received no response).</td></tr>
</table>

Attachment A-2. Concentration Summary

The tables below provide summary statistics on the bulk and leachate concentrations in FGD gypsum. The summary statistics presented below reflect the raw data after the filtering discussed in this appendix. All of the data is weighted equally and does not account for the prevalence of different environmental conditions captured in the full-scale model, such as variations in soil pH.

Summary of Filtered Bulk Concentration Data

Constituent	Bulk Concentration (mg/kg)					
	Min	Median	Mean	St. Dev	Geo Mean	Max
All Samples						
Aluminum	61	380	1,232	2,287	502	12,700
Antimony	0.06	0.60	2.6	4.2	0.76	24
Arsenic	0.19	2.8	3.2	2.4	2.2	11
Barium	0.74	12	22	21	13	82
Beryllium	0.01	0.03	0.05	0.10	0.03	0.60
Boron	0.76	10	33	67	13	387
Cadmium	0.01	0.13	0.24	0.31	0.13	1.9
Chloride	34	480	687	621	427	2,616
Chromium	0.10	2.9	4.4	3.8	3.0	15
Cobalt	0.04	0.31	0.80	1.0	0.40	4.3
Copper	0.001	1.3	1.5	1.0	0.95	4.1
Iron	222	1,000	1,161	852	952	5,881
Lead	0.002	1.0	1.5	1.5	1.0	8.3
Magnesium	70	1,322	1612	1,566	1,001	7,430
Manganese	0.03	8.75	22	33	8.9	161
Mercury	0.01	0.34	0.46	0.50	0.30	3.1
Molybdenum	0.11	0.95	1.6	2.0	1.0	12
Nickel	0.09	1.3	1.6	1. 4	1.3	6.8
Selenium	0.73	5.4	9.4	9.5	6.1	46
Strontium	71	161	197	104	174	531
Thallium	0.002	0.02	0.30	0.58	0.05	2.8
Vanadium	0.15	1.9	3.4	4.8	2.2	30
Zinc	1.6	6.2	9.1	6.8	7.0	29
Unwashed Only						
Aluminum	142	959	2,080	3,028	1,035	12,700
Antimony	0.08	0.66	2.4	3.0	0.95	9.1
Arsenic	0.95	3.0	3.6	2.3	3.1	11
Barium	2.4	27	33	25	22	82
Beryllium	0.01	0.04	0.06	0.05	0.03	0.13
Boron	9.4	51	100	108	64	387
Cadmium	0.02	0.22	0.31	0.27	0.21	1.2
Chloride	81	833	1,209	1,177	790	4,816

Summary of Filtered Bulk Concentration Data

Constituent	Bulk Concentration (mg/kg)					
	Min	Median	Mean	St. Dev	Geo Mean	Max
Chromium	1.0	5.4	6.4	4.5	4.9	17
Cobalt	0.22	1.1	1.4	1.2	0.98	4.4
Copper	0.38	1.93	1.9	0.96	1.6	3.5
Iron	635	1,512	1,711	1,006	1,540	5,881
Lead	0.63	1.2	1.6	0.84	1.4	3.3
Magnesium	201	1,927	2,677	2,227	1,777	7,430
Manganese	8.7	27	49	54	28	161
Mercury	0.01	0.41	0.58	0.53	0.38	2.3
Molybdenum	0.54	1.8	2.6	2.7	1.8	12
Nickel	0.84	2.2	2.1	1.2	1.9	5.0
Selenium	0.73	6.6	11	9.1	7.4	33
Strontium	97	172	215	117	192	534
Thallium	0.01	0.28	0.48	0.60	0.14	2.3
Vanadium	1.6	3.2	6.3	8.5	3.9	30
Zinc	1.8	9.0	10	7.1	7.5	23
Washed Only						
Aluminum	61	256	1,289	2.777	397	11,600
Antimony	0.06	0.5	2.3	4.8	0.58	24
Arsenic	0.27	3.1	3.5	2.4	2.7	10
Barium	0.74	9.3	15	16	9.7	53
Beryllium	0.01	0.03	0.05	0.13	0.02	0.6
Boron	2.2	8.6	11	9.0	8.8	44
Cadmium	0.01	0.12	0.17	0.15	0.10	0.47
Chloride	15	219	271	328	127	1,255
Chromium	0.60	2.5	4.3	4.6	2.9	20
Cobalt	0.07	0.23	0.91	1.3	0.39	4.2
Copper	0.001	1.0	1.2	0.94	0.71	4.1
Iron	277	808	979	510	853	2,114
Lead	0.002	1.0	1.9	2.6	0.91	12
Magnesium	58	989	1,143	909	800	4,134
Manganese	0.97	7.5	14.8	19	7.9	79
Mercury	0.007	0.36	0.50	0.52	0.28	3.1
Molybdenum	0.15	0.97	1.7	19	0.93	8.1
Nickel	0.35	1.2	1.2	0.44	1.1	2.2
Selenium	2.2	5.03	10	11	6.6	46
Strontium	88	173	212	108	189	527
Thallium	0.002	0.02	0.30	0.57	0.05	2.8
Vanadium	0.67	1.6	2.4	2.3	1.8	10
Zinc	2.1	7.0	9.0	6.0	7.3	27

Summary of Filtered Leachate Concentration Data

Constituent	Leachate Concentration (µg/L)					
	Min	Median	Mean	St. Dev	Geo Mean	Max
All Samples						
Aluminum	2.0	220	723	1,352	237	10,282
Antimony	0.02	2.8	3.7	5.3	1.9	30
Arsenic	2.5	3.2	8.1	23	3.9	197
Barium	1.9	76	88	81	65	565
Beryllium	0.23	3.2	2.6	1.3	2.0	7.0
Boron	3.9	417	2,826	5,222	447	22,396
Cadmium	0.01	0.85	3.9	8.2	1.1	55
Chloride	2065	20,229	91,515	110,872	24,766	344,368
Chromium	0.05	9.00	10.9	16	5.1	158
Cobalt	0.37	7.00	9.5	13	5.1	69
Copper	0.50	3.50	11	17	5.5	95
Iron	0.43	4.80	2,268	5,117	36	27,320
Lead	0.07	1.2	7.9	12	1.9	31
Magnesium	290	13,890	62,672	102,178	13,405	525,800
Manganese	1.7	273	1,710	3615	253	23,659
Mercury	0.0004	0.01	0.10	0.34	0.01	3.3
Molybdenum	0.36	6.2	20	36	8.1	170
Nickel	0.05	45	81	97	20	434
Selenium	2.9	67	213	418	77	2,478
Strontium	114	539	914	764	668	3,100
Thallium	0.01	2.6	4.5	5.8	1.7	34
Vanadium	0.04	8.0	31	94	5.5	657
Zinc	1.0	165	222	275	96	1,641
Unwashed Only						
Aluminum	5.0	385	716	926	293	3,790
Antimony	0.16	2.8	3.6	2.8	2.9	17
Arsenic	2.5	3.2	14	35	5.2	197
Barium	22	88	109	82	91	445
Beryllium	0.31	3.2	3.1	0.60	2.9	3.2
Boron	58	2,214	5,282	6,691	1,922	22,396
Cadmium	0.05	1.9	7.0	12	2.6	55
Chloride	10,822	143,264	165,538	114,436	105,966	344,368
Chromium	0.16	9.0	13	23	7.3	158
Cobalt	2.1	9.0	13	15	7.3	58
Copper	1.6	8.0	15	18	8.7	92
Iron	1.5	443	2,946	4,929	120	23,590
Lead	0.08	1.2	6.4	11	2.1	31
Magnesium	1,148	5,785	104,169	121,151	44,344	525,800
Manganese	1.7	1,206	3,254	4,899	920	23,659

Summary of Filtered Leachate Concentration Data

Constituent	Leachate Concentration (µg/L)					
	Min	Median	Mean	St. Dev	Geo Mean	Max
Mercury	0.001	0.01	0.14	0.48	0.01	3.3
Molybdenum	3.8	12	28	45	12	170
Nickel	0.05	98	124	111	66	434
Selenium	13	189	291	430	159	2,478
Strontium	114	596	924	710	684	2,772
Thallium	0.01	2.6	6.9	7.4	3.9	34
Vanadium	1.4	13	53	135	13	657
Zinc	3.4	215	329	355	205	1,641
Washed Only						
Aluminum	2.0	171	847	1,769	251	10,282
Antimony	0.02	2.8	2.2	1.4	1.3	7.0
Arsenic	2.5	3.2	4.1	5.8	3.2	44
Barium	8.9	60	73	79	54	565
Beryllium	0.31	3.2	2.6	1.4	1.9	7.0
Boron	3.9	209	840	2,002	156	7,900
Cadmium	0.01	0.85	1.7	1.8	0.77	7.0
Chloride	2,065	2,065	20,711	36,941	6,166	116,409
Chromium	0.05	9.0	11	8.9	5.0	34
Cobalt	2.1	2.1	6.7	11	4.2	69
Copper	0.67	3.5	9.7	16	5.0	95
Iron	1.4	1.6	1,991	5,667	16	27,320
Lead	0.07	1.2	9.0	13	1.8	31
Magnesium	300	4,397	31,396	70,825	5,565	353,800
Manganese	1.7	81	493	900	89	4,902
Mercury	0.0004	0.02	0.06	0.15	0.01	0.83
Molybdenum	0.36	3.8	15	25	6.5	92
Nickel	0.05	37	52	66	11	310
Selenium	2.9	45	181	435	56	2,064
Strontium	290	516	945	851	680	3,100
Thallium	0.01	2.6	2.6	2.8	0.93	13
Vanadium	0.04	7.0	14	21	3.1	69
Zinc	3.8	150	154	123	77	492

Appendix B. Benchmarks

This appendix describes the approach used to select the chronic benchmarks used in this beneficial use evaluation to estimate the potential for adverse impacts to human and ecological receptors. An adverse effect is any abnormal, harmful, or undesirable change that results from exposure to a chemical constituent or other stressor. It is important be aware that the benchmarks considered relevant and appropriate for this evaluation may not be the same as those other beneficial use evaluations. In some cases, other appropriate benchmarks may be available or have already been defined by state or federal regulatory bodies based on the intended use.

B.1 Human Health Benchmarks

Adverse health effects for human receptors are divided into two main categories: carcinogenic and noncarcinogenic. Carcinogenic effects are those that result in the development of cancer somewhere in the organism. Noncarcinogenic effects are those that result in any adverse health effect other than cancer. Some stressors may result in both carcinogenic and noncarcinogenic effects, depending on the route through which the receptor is exposed and the magnitude of the exposure. EPA used **Equation B.1** and **Equation B.2** to calculate noncancer hazard quotient and cancer risk, respectively. These equations do not include any additional factors required to harmonize units.

$$(\mathbf{B.1}) \qquad \mathrm{HQ} \quad \frac{C_{ia} \cdot \mathrm{EF} \cdot \mathrm{IR} \cdot \mathrm{ABS}}{\mathrm{BW} \cdot \mathrm{RfD}}$$

$$(\mathbf{B.2}) \qquad \mathrm{Risk} \quad \frac{C_{ia} \cdot \mathrm{CSF} \cdot \mathrm{EF} \cdot \mathrm{ED} \cdot \mathrm{IR} \cdot \mathrm{ABS}}{\mathrm{AT} \cdot \mathrm{BW}}$$

Where:

ABS – Absorption factor (%)
AT – Averaging time (yr)
BW – Body weight (kg)
CSF – Cancer slope factor $(\text{mg/kg-day})^{-1}$
C_{ia} – Concentration in a given media (mg/kg or mg/L)
ED – Exposure duration (yr)
EF – Exposure frequency (days/yr)
IR – Intake rate (mg/day or L/day)
RfD – Reference dose (mg/kg-day)

The equations presented above are generalized, and can be further refined to address indirect exposures to contaminated media by substituting in **Equations B.3** through **B.7** for specific variables in **Equations B.1** and **B.2**. These equations can be applied as presented to calculate risks from a specified constituent concentration (**Section 6: Risk Modeling**), or rearranged to calculate the concentration that corresponds to a specified risk (**Section 5: Screening Analysis**). The following subsections describe the values used in this evaluation for each of the variables listed in these equations.

$$(\mathbf{B.3})\quad C_{Fish}\quad C_{Water}[(F_{T3}\cdot BCF_{T3})+(F_{T4}\cdot BCF_{T4})]$$

$$(\mathbf{B.4})\quad C_{Produce}\quad BCF_{Produce}\cdot C_{Soil}(1-MAF)$$

$$(\mathbf{B.5})\quad C_{Beef}\quad BCF_{Beef}\left[(C\cdot Q\cdot f)_{Soil}+\ (C_{Soil})\left(\sum BCF_i\cdot Q_i\cdot f_i\right)_{i=\text{Grain,Forage,Silage}}\right]$$

$$(\mathbf{B.6})\quad C_{Milk}\quad BCF_{Milk}\left[(C\cdot Q\cdot f)_{Soil}+\ (C_{Soil})\left(\sum BCF_i\cdot Q_i\cdot f_i\right)_{i=\text{Grain,Forage,Silage}}\right]$$

$$(\mathbf{B.7})\quad C_{Skin}\cdot IR\quad C_{Water}(K_p)(EV)(t_{Event})(SA)$$

Where:

BCF – Bioconcentration factor (unitless)
C_{Media} – Concentration in media (mg/kg for fish, produce, beef) (mg/L for water, milk)
$C_{Skin}\cdot IR$ – Uptake rate through skin (mg/day)
EV – Event Frequency (events/day)
f – Fraction of Media Contaminated (unitless)
F_{T3} – Fraction of Fish Ingested from Trophic Level 3 (unitless)
F_{T4} – Fraction of Fish Ingested from Trophic Level 4 (unitless)
K_p – Dermal Permeability Coefficient (cm/hr)
MAF – Moisture Adjustment Factor (%)
Q – Ingestion Rate by Cow (kg/day)
SA – Skin Area (cm^2)
t_{Event} – Duration of Individual Exposure Event (hr/events)

B.1.2 Target Hazard Quotient and Risk

Target hazard quotient and target risk are unitless numbers that represent the estimated likelihood that a non-carcinogenic or carcinogenic adverse effect will occur. Target hazard quotients, calculated for non-carcinogenic constituents, are the ratio of the constituent concentration to which a receptor may be exposed and the concentration below which no adverse effects are known or anticipated to occur. For the screening analysis, the target hazard quotient was set to 1.0 based on the recommendations of *Risk Assessment Guidance for Superfund* (US EPA, 1989). Target risks are established for carcinogenic constituents. Unlike approaches for assessing some non-carcinogenic constituents, this approach assumes that there is some risk of cancer at any level of exposure. Any increase in exposure to a carcinogen translates to some increased probability of developing cancer. The current evaluation considered cancer risks within the 1×10^{-4} and 1×10^{-6} risk range. From this range, the specific target risk of 1×10^{-5} was selected based on the US EPA Office of Resource Conservation and Recovery's presumptive listing benchmark (59 FR 66075). This level is equivalent to one additional incidence of cancer for every 100,000 individuals exposed to a given carcinogen.

B.1.2 Toxicity Values

Human health benchmarks are based on specific adverse effects that may occur. Reference doses (RfDs) and reference concentrations (RfCs) are used to evaluate noncancer effects from oral and inhalation exposures, respectively. RfDs and RfCs are estimates of a daily exposure to the human population (including sensitive subgroups) that is likely to be without appreciable risk of deleterious noncancer effects. However, an average lifetime exposure above the RfD (or RfC) does not imply that an adverse health effect will necessarily occur. Oral cancer slope factors (CSFs) are used to evaluate carcinogenic effects from oral exposures. The CSF is an upperbound estimate (approximating a 95% confidence limit) of the increased human cancer risk from a lifetime of exposure.

EPA identified toxicity values according to the hierarchy established in the *2003 Office of Solid Waste and Emergency Response Directive 9285.7-53*, which encourages prioritization of toxicity values from sources that are current, transparent and publicly available, and that have been peer reviewed (U.S. EPA, 2003). Accordingly, a three-tiered approach was followed to use higher priority data sources based on availability. Values in lower tiers may not be calculated in the same way as RfDs, RfCs and CSFs, but are treated as equivalent for the purposes of this evaluation.

Tier I

Integrated Risk Information System (IRIS) contains RfDs and RfCs for chronic noncarcinogenic health effects, and oral CSFs for carcinogenic effects. IRIS is considered the highest quality science-based, developed to support EPA regulatory activities. IRIS assessments have been peer-reviewed and represent Agency-wide consensus.

Tier II

Provisional Peer-Reviewed Toxicity Values (PPRTVs) are derived by the Superfund Program after a review of the relevant scientific literature using the methods, sources of data and guidance for value derivation used by the EPA IRIS Program. All provisional peer-reviewed toxicity values receive internal review by EPA scientists and external peer review by independent scientific experts. However, PPRTVs do not reflect Agency-wide consensus, because PPRTVs are developed specifically for the Superfund Program. PPTRVs include cancer and noncancer values for both oral and inhalation exposure and are treated as equivalent to RfDs, RfCs and CSFs.

Tier III

Agency for Toxic Substances and Disease Registry (ATSDR) Chronic Minimal Risk Levels (MRLs) are substance-specific health guidance levels for noncarcinogenic effects. An MRL is intended to be an estimate of the daily human exposure to a hazardous substance that is likely to be without appreciable risk of adverse noncancer health effects over a specified duration of exposure. MRLs are derived for oral and inhalation routes of exposure in a manner similar to RfDs and RfCs, respectively. MRLs have undergone both internal and external peer review.

New Jersey Department of Environmental Protection (NJDEP, 2009) provides an oral cancer benchmark for chromium (VI) for their soil cleanup program based on information from a study by the National Toxicity Program (NTP, 2008).

For lead, EPA currently has no consensus on the development of an RfD or CSF because of the difficulty associated with identifying an effect threshold needed to develop these benchmarks. Therefore, the maximum contaminant level (MCL) for drinking water was used as an alternative method of approximating human health risk.

Under this evaluation, only oral benchmarks were collected for all constituents except for elemental mercury which is volatile and thus was evaluated for vapor inhalation as described in **Appendix D (Screening Analysis)**. The chronic oral human health toxicity values used in this evaluation are summarized in **Table B-1**. Values were last reviewed in January 2019.

Table B-1. Chronic Oral Human Health Toxicity Values

Constituent	CASRN	Value	Target Organ	Type	Citation
Cancer (mg/kg-day)$^{-1}$					
Arsenic	7440-38-2	1.50E+00	Cancer	IRIS	U.S. EPA (1995a)
Chromium (VI)	18540-29-9	5.00E-01	Cancer	NJDEP	NJDEP (2009)
Noncancer (mg/kg-day)					
Aluminum	7429-90-5	1E+00	Neurological	PPRTV	U.S. EPA (2006a)
Antimony	7440-36-0	4E-04	Hematological	IRIS	U.S. EPA (1987)
Arsenic	7440-38-2	3E-04	Dermal, Cardiovascular	IRIS	U.S. EPA (1991a)
Barium	7440-39-3	2E-01	Kidney	IRIS	U.S. EPA (2005a)
Beryllium	7440-41-7	2E-03	Gastrointestinal	IRIS	U.S. EPA (1998a)
Boron	7440-42-8	2E-01	Developmental	IRIS	U.S. EPA (2004)
Cadmium	7440-43-9	1E-03 (Food) 5E-04 (Water)	Kidney	IRIS	U.S. EPA (1989)
Chromium (III)	16065-83-1	1.5E+00	No system effect in lab study. Respiratory, Immunological	IRIS	U.S. EPA (1998b)
Chromium (VI)	18540-29-9	3E-03	No system effect in lab study. Respiratory, Gastrointestinal, Immunological, Hematological, Reproductive, Developmental	IRIS	U.S. EPA (1998c)
Cobalt	7440-48-4	3E-04	Thyroid	PPRTV	U.S. EPA (2008)
Copper	7440-50-8	1E-02	Gastrointestinal	ATSDR	ATSDR (2004)
Iron	7439-89-6	7E-01	Gastrointestinal	PPRTV	U.S. EPA (2006)
Manganese	7439-96-5	1.4E-01	Nervous System / CNS Effects	IRIS	U.S. EPA (1995b)
Mercury (II)	7487-94-7	3E-04	Immunological, Urinary	IRIS	U.S. EPA (1995c)
Mercury (Methyl)	22967-92-6	1E-04	Nervous System, Developmental	IRIS	U.S. EPA (2001)
Molybdenum	7439-98-7	5E-03	Urinary	IRIS	U.S. EPA (1992a)
Nickel	7440-02-0	2E-02	Body Weight, Cardiovascular, Liver	IRIS	U.S. EPA (1991b)
Selenium	7782-49-2	5E-03	Dermal, Hematological, Nervous System	IRIS	U.S. EPA (1991c)
Strontium	7440-24-6	6E-01	Bone/Teeth, Musculoskeletal	IRIS	U.S. EPA (1992b)
Thallium	7440-28-0	1E-05	Hair follicular atrophy	PPRTV	U.S. EPA (2012)
Vanadium	7440-62-2	5E-03	Dermal	IRIS	U.S. EPA (1988)
Zinc	7440-66-6	3E-01	Immunological, Hematological	IRIS	U.S. EPA (2005b)

B.1.2 Exposure Factors

Exposure factors are data that quantify human behavior patterns (e.g., ingestion rates of drinking water and fish) and physiological characteristics (e.g., body weight) that affect an individual's exposure to environmental contaminants. These data can be used to construct realistic assumptions concerning the magnitude of exposure to and subsequent intake of a contaminant in the environment. The exposure factors data also enables EPA to differentiate the exposures of individuals of different ages. The derivation of human exposure factors used in both the screening and probabilistic analyses are described below.

The Agency relied primarily on the *Exposure Factors Handbook* (*EFH*) (U.S. EPA, 2011) and the *Child-Specific Exposure Factors Handbook* (*CSEFH*) (U.S. EPA, 2008). Where sufficient data were available, the percentiles and corresponding data points obtained from these two sources were used to develop a cumulative distribution in order to capture variability within the U.S. population. Otherwise, EPA relied on point values selected based on the recommendations of the EFH, CSEFH or established Agency guidance (U.S. EPA, 1991; 2014).

There has been considerable effort across the Agency to improve the accuracy and consistency of childhood exposure assessments. In the *Guidance on Selecting Age Groups for Monitoring and Assessing Childhood Exposures to Environmental Contaminants* (U.S. EPA, 2005c), EPA recommended specific age cohorts (i.e., groups) intended to better capture the large variability in physiological and behavioral characteristics of child receptors during different stages of development. Narrower age cohorts were identified where rapid developmental changes occur, while broader age groups were identified where the rate of development decreases. These age groupings and the supporting rationale for their selection have been subjected to internal and external scientific peer review. In total, receptors were divided into the eight distinct age cohort groupings recommended by U.S. EPA (2005c). The general methodology for collecting human exposure data for the probabilistic analysis relied on the EPA data from the 2011 EFH or 2008 CSEFH in one of three ways:

1. When the available data were adequate (as for most input variables), nonparametric approaches were used to fit distributions to the cumulative distribution (percentiles) of the data using @Risk software (available at www.palisade.com/risk/). Fitting nonparametric distributions removed parameter uncertainty associated with the fitting of specific parametric distributions (U.S. EPA, 2000).

2. When the available data were not adequate to support the statistical fitting for a specific age cohort, the data fit to the closest age cohort available was used instead.

3. When available data were not adequate for either of the above methods, variables were fixed at values recommended in the EFH or CSEFH or according to established EPA policy.

This section describes how the various distributions of exposure factor data were collected and processed for use in the probabilistic analysis. Probabilistic exposure analyses involve sampling values from a distribution with the same characteristics as the data using the values to estimate risk. For most variables for which distributions were developed, EPA exposure factor data were analyzed to fit nonparametric models. Steps in the development of distributions included preparing data, fitting models, assessing fit, and preparing parameters to characterize distributional uncertainty in the model inputs.

For many exposure factors, the data from the EFH and CSEFH include sample sizes and estimates of the following parameters for specific receptor types and age groups: mean, standard deviation, standard error, and percentiles corresponding to some subset of the following probabilities: 0.01, 0.02, 0.05, 0.10, 0.15, 0.25, 0.50, 0.75, 0.85, 0.90, 0.95, 0.98, and 0.99. These percentile data, where available, were used as a basis for fitting distributions. Although in no case are all of these percentiles provided for a single factor, seven or more are typically available. Therefore, using the percentiles provided a better representation of the available information than fitting distributions simply based on the method of moments (i.e., selecting models that agree with the data mean and standard deviation).

Neither the EFH nor the CSEFH makes use of the standardized age cohorts recommended in U.S. EPA (2005c). Different exposure factors are reported for different age categories based on the information available in the scientific literature. Therefore, to obtain the percentiles for fitting the eight standardized age cohorts used for the revised risk assessment, each cohort-specific value for a given exposure factor was assigned to one of the cohorts. When multiple cohorts were fit into a single cohort, the percentiles from the EFH or CSEFH were averaged within each cohort (e.g., data on 6 year olds and 9 year olds were averaged for the 6 to <11 age cohort). If sample sizes were available, weighted averages were used, with weights proportional to sample sizes. If sample sizes were not available, equal weights were assumed (i.e., the percentiles were simply averaged).

Nonparametric distributions were used to characterize the data. The nonparametric approach fits an optimal smooth curve to the cumulative distribution (percentiles) of the data. The best nonparametric fit is selected as the one that minimized the distance between the smooth curve and the empirical curve generated by the percentiles of the data. The maximum and minimum are used to specify the range of the simulated values. Depending on the data set, there could be more than one distribution (parametric or nonparametric) that could be considered a good fit for the data. Selecting an incorrect exposure distribution model may bias the risk assessment results, producing incorrect conclusions. Therefore, the application of goodness-of-fit statistics was required to select between competing distributions and to reduce model uncertainty. One goodness-of-fit statistic available was the root mean squared error, defined as the root of the average of the squared differences between the predicted percentile and the observed percentile. The other goodness-of-fit statistic available was the Chi-square based comparison of the empirical cumulative distribution (derived from the cumulative data) or the nonparametric cumulative distribution. Graphical displays were also used to evaluate the appropriateness of the selected distribution. A plot of the observed percentiles (from the cumulative data) vs. the nonparametric cumulative distribution was created. In some cases, exposure distributions are highly skewed, and there is a probability, although small, that a combination of extreme values might be selected from the tails of the distributions. The resulting distributions are discussed in the following subsections, with the specific percentiles presented in accompanying tables. Highlighted values are those selected for use in the screening analysis.

Drinking Water Ingestion:

Drinking water intake data were obtained from Table 3-19 of the CSEFH and Table 3-38 of the EFH, for children and adults, respectively. Weighted averages of percentiles and means were calculated for the 0 to < 1 year infant (based on birth to < 1 month, 1 to < 3 months, 3 to < 6 months and 6 to < 12 months), the 16 to < 21 years age group (based on 16 to < 18 years and 18 to < 21 years) and adults (based on 20 to 44, 45 to 64, and 64 to 74 years), using the number of observations in each sub-cohort as weights. **Table B-2** presents the water ingestion data used each age cohort.

Table B-2. Drinking Water Consumption Rate Data (mL/kg-day)

Age Group (Years)	N	Mean	Percentile						
			P10	P25	P50	P75	P90	P95	P99
Infant(1)	948	71	7	25	66	104	140	164	217
1 to < 2	880	27	4	9	20	36	56	75	109
2 to < 3	879	26	4	9	21	36	52	62	121
3 to < 6	3,703	24	3	8	19	33	49	65	97
6 to < 11	1,439	17	3	6	13	23	35	45	72
11 to < 16	911	13	2	5	10	17	26	34	54
16 to < 21	700	13	2	5	10	17	27	34	61
Adult	7,616	16	2	6	12	22	34	42	64

Source: Table 3-19, CSEFH for child cohorts and Table 3-38, EFH (U.S. EPA, 2011) for adults.
1) Weighted average based on sub-cohorts presented in the CSEFH (U.S. EPA, 2008).

Drinking water consumption rate data for infants (birth to < 1 month, 1 to < 3 months, 3 to < 6 months and 6 to < 12 months) are available from Table 3-19 of the CSEFH. **Table B-3** presents the water ingestion data used for infants. As drinking water concentrations are provided by the model as annual averages, the infant consumption rates were averaged to estimate a 0 to < 1 year infant. The data were weighted by sample size because the small sample sizes did not meet minimum requirements as described in the *Third Report on Nutrition Monitoring in the United States* (IBNMRR, 1995) for numerous percentiles in numerous sub-cohorts.

Table B-3. Drinking Water Consumption Rates for Infant Sub-cohorts (mL/kg-day)

Age Group (Months)	N	Mean	Percentile						
			P10	P25	P50	P75	P90	P95	P99
Birth to < 1	37	137	11	65	138	197	235	238	263
1 to < 3	108	119	12	71	107	151	228	285	345
3 to < 6	269	80	7	27	77	118	148	173	222
6 to < 12	534	53	5	12	47	81	112	129	186
Weighted Average	948	71	7	25	66	104	140	164	217

Source: Table 3-19, *CSEFH* (U.S. EPA, 2008).

Beef Ingestion Rate:

Consumption rates for beef are presented in **Table B-4**. These data are for consumption of homegrown beef. Table 13-33 of the EFH provides data (in g WW/kg-d) for farming households by age groups (6–11 years and 12–19 years) and for adult farmers (i.e., households who farm). Data for ages 6–11 years were used for the 1 to <2, 2 to <3, 3 to < 6, and 6 to < 11 years age groups. Data for ages 12–19 years were used for the 11 to <16 and 16 to <21 years age groups.

Beef consumption rate data were adjusted to account for post-cooking losses. A mean net post-cooking loss of 29.7 percent accounts for losses from cutting, shrinkage, excess fat, bones, scraps, and juices. This value was obtained from Table 13-69 of the EFH. Values shown in Table B-4 are before these adjustments.

Table B-4. Beef Consumption Data (g WW/kg-d)

Age Group (Years)	N	Mean	Percentile								
			P01	P05	P10	P25	P50	P75	P90	P95	P99
1 to < 2	38	3.77	0.35	0.66	0.75	1.32	2.11	4.43	11.4	12.5	13.3
2 to < 3	38	3.77	0.35	0.66	0.75	1.32	2.11	4.43	11.4	12.5	13.3
3 to < 6	38	3.77	0.35	0.66	0.75	1.32	2.11	4.43	11.4	12.5	13.3
6 to <11	38	3.77	0.35	0.66	0.75	1.32	2.11	4.43	11.4	12.5	13.3
11 to <16	41	1.72	0.38	0.48	0.51	0.9	1.51	2.44	3.53	3.57	4.28
16 to < 21	41	1.72	0.38	0.48	0.51	0.90	1.51	2.44	3.53	3.57	4.28
Adult	182	2.63	0.27	0.39	0.59	0.90	1.64	3.25	5.39	7.51	11.3

Sources: Table 13-33 *EFH* (U.S. EPA, 2011).

Milk Ingestion Rate:

Consumption rates for milk are presented in **Table B-5**. Table 13-25 of the EFH provides data (in g WW/kg-d) for adult farmers. Data for children consuming homegrown milk are not available. Therefore, we used data for general population from EFH Table 11-3. These data were provided for ages 1–2, 3–5, 6–12, and 13–19 years. Data for ages 1–2 years were used for the 1 to <2 and 2 to <3 years age groups. Data for ages 3–5 were used for the 3 to <6 years age group. Data for ages 6–12 years were used for the 6 to <11 years age group. Data for ages 13–19 years were used for the 11 to <16 and 16 to <21 years age groups.

Table B-5. Milk Consumption Data (g WW/kg-d)

Age Group (Years)	N	Mean	Percentile								
			P01	P05	P10	P25	P50	P75	P90	P95	P99
1 to < 2	1,052	43.2	1.0	5.7	10.7	20.3	39.1	59.4	84.1	94.7	141.2
2 to < 3	1,052	43.2	1.0	5.7	10.7	20.3	39.1	59.4	84.1	94.7	141.2
3 to < 6	978	24.0	0.9	4.5	8.3	13.6	20.7	32.0	41.9	51.1	68.2
6 to < 11	2,256	12.9	0.5	1.5	2.6	5.6	10.8	17.8	26	31.8	42.9
11 to < 16	3,450	5.5	0.1	0.4	0.6	1.6	4.0	7.6	12.3	16.4	24.9
16 to < 21	3,450	5.5	0.1	0.4	0.6	1.6	4.0	7.6	12.3	16.4	24.9
Adult	63	17.1	0.4	0.74	3.18	9.06	12.1	20.4	34.9	44.0	80.1

Fish Consumption:

Fish consumption data were obtained from Table E-3 of U.S. EPA (2015), based on the data presented in Table 10-1 of the EFH. Values were selected for consumer-only ingestion rates for uncooked finfish (excludes shellfish because of focus on fresh water). From the available data, mean concentrations were used to represent recreational fishers, while the 95th percentile rates were used to represent subsistence fishers. **Table B-6** presents fish consumption rate data used to prepare Monte Carlo simulations.

Table B-6. Fish Intake Rates for All Ages (g/kg-day)

Age Group (Years)	Recreational Fisher	Subsistence Fisher
1 to < 2	1.60	4.90
2 to < 3	1.60	4.90
3 to < 6	1.30	3.60
6 to < 11	1.10	2.90
11 to < 16	0.660	1.70
16 to < 21	0.660	1.70
Adult	0.665	2.05

Sources: Table E-3, U.S. EPA (2015)

Exposure Duration

Exposure durations for residents were determined using data on residential occupancy from Tables 16-109 and 16-113of the EFH. The data represent the total time a person (both male and female) is expected to live at a single location, based on age. For adult residents, data reported for farm residents were used to capture highly exposed receptors within the population. For child residents, data reported for the 3-year age group were used to represent infants to < 6 years. Data reported for ages 6 and 9 were averaged to represent 6 to < 11 years. Data on ages 12 and 15 were averaged to represent 11 to < 16 years. Data reported for age 18 were used to represent 16 to < 21 years. **Table B-7** presents exposure duration data rounded to the nearest whole year. The source for the recommended EFH distribution was a Monte Carlo simulation that estimated a probability distribution for residential occupancy period based on the probability of moving and dying.

Table B-7. Human Exposure Durations (ED) for All Ages (year)

Age Group (Years)	Mean	Percentile					
		P25	P50	P75	P90	P95	P99
Infant	7	3	5	8	13	17	22
1 to < 2	7	3	5	8	13	17	22
2 to < 3	7	3	5	8	13	17	22
3 to < 6	7	3	5	8	13	17	22
6 to < 11	8	5	8	11	16	18	22
11 to < 16	9	5	9	13	16	18	23
16 to < 21	8	4	7	11	16	19	23
Adult	N/A	2	10	27	48	58	N/A

N/A – Not Available

Sources: Children: Table 16-109, EFH (U.S. EPA, 2011)
Adult farmer: Table 16-113, EFH (U.S. EPA, 2011)

Body Weight:

Weighted averages of percentiles and means were calculated for the infant age group (based on birth to < 1 month, 1 to < 3 months, 3 to < 6 months and 6 to < 12 months) and adults (based on male and female data). **Table B-8** presents body weight data. Body weight data were obtained from Table 8-3 of the EFH where data were presented by age for males and females combined.

Table B-8. Body Weight Data (kg)

Age Group (Years)	N	Mean	Percentile								
			P05	P10	P15	P25	P50	P75	P85	P90	P95
Infant	1,858	7.8	6	6.4	6.7	7.1	7.8	8.6	9.0	9.3	9.7
1 to < 2	1,176	11.4	8.9	9.3	9.7	10.3	11.3	12.4	13	13.4	14
2 to < 3	1,144	13.8	10.9	11.5	11.9	12.4	13.6	14.9	15.8	16.3	17.1
3 to < 6	2,318	18.6	13.5	14.4	14.9	15.8	17.8	20.3	22	23.6	26.2
6 to < 11	3,593	31.8	19.7	21.3	22.3	24.4	29.3	36.8	42.1	45.6	52.5
11 to < 16	5,297	56.8	34	37.2	40.6	45	54.2	65	73	79.3	88.8
16 to < 21	4,851	71.6	48.2	52	54.5	58.4	67.6	80.6	90.8	97.7	108
Adult	12,504	71.4	52.9	56.0	58.2	61.7	69.3	78.5	84.9	89.8	97.6

Sources: Table 8-3 *EFH* (U.S. EPA, 2011) for children and adults.

Fixed Parameters

Certain parameters were fixed either because the available data were not adequate to generate a full distribution or because only a single, high-end value was necessary to screen out the associated exposure pathway. **Table B-9** lists the parameters along with the value selected and source.

Table B-9. Human Exposure Factor Data Used in Modeling: Constants

Averaging Time	AT	All Ages		yr	U.S. EPA (2014)
Exposure Frequency	EF	All Ages		d/yr	U.S. EPA (2014)
Event Frequency	EV	All Ages		event/day	U.S. EPA (2014)
Event Duration	t_{Event}	All Ages		hr/event	U.S. EPA (2014)
Skin Surface Area	SA	All Child Cohorts		cm^2	U.S. EPA (2014)
Soil + Dust Ingestion Rate	CR_{Soil}	All Child Cohorts		mg/day	U.S. EPA (2014)
Protected Fruit Ingestion Rate	CR_{PFruit}	Age 1 to <2		$g_{(WW)}$/kg-day	90th Percentile from U.S. EPA (2011), Table 13-62
Exposed Fruit Ingestion Rate	CR_{EFruit}	Age 1 to <2		$g_{(WW)}$/kg-day	90th Percentile from U.S. EPA (2011), Table 13-61
Protected Vegetable Ingestion Rate	CR_{PVeg}	Age 1 to <2		$g_{(WW)}$/kg-day	90th Percentile from U.S. EPA (2011), Table 13-64
Exposed Vegetable Ingestion Rate	CR_{EVeg}	Age 1 to <2		$g_{(WW)}$/kg-day	90th Percentile from U.S. EPA (2011), Table 13-63
Root Vegetable Ingestion Rate	CR_{RVeg}	Age 1 to <2		$g_{(WW)}$/kg-day	90th Percentile from U.S. EPA (2011), Table 13-65
Arsenic Absorption Factor (Soil)	$ABS_{(soil)}$	All Ages		%	U.S. EPA (2012)
Arsenic Absorption Factor (Other Media)	$ABS_{(other)}$	All Ages		%	U.S. EPA (1991)
Beef Preparation Loss*	$Loss_{prep}$	All Ages		%	U.S. EPA (2011), Table 13-69
Trophic Level 3 Fish Consumed	F_{T3}	All Ages	36	%	U.S. EPA (2015)
Trophic Level 4 Fish Consumed	F_{T4}	All Ages	64		

* Beef preparation losses applied because measured ingestion rates reflect based on foods as brought into the household and not in the form in which they are consumed.

B.1.3 Produce and Animal Product Exposure Factors

Chemical-specific factors were used to estimate the degree to which inorganic constituents may accumulate in different plants and animals, as well as the resulting human exposures from consumption of produce and animal products (i.e., beef, milk, fish). EPA reviewed the available literature to assemble values.

Produce

Bioconcentration factors (BCFs) are used to estimate the magnitude of accumulation into produce. Where possible, EPA relied on data from field studies because of the potential for greenhouse pot studies to overpredict uptake (U.S. EPA, 1992c). However, pot studies were used when field study data were not available. In instances where both the soil and crop data were non-detect, the data were filtered out to avoid introducing excessive uncertainty. The individual data points used to calculate BCFs were drawn from three sources:

- The Gypsum Constituent Database [**Appendix A (Constituent Data)**],
- *Technical Support Document for Land Application of Sewage Sludge* (U.S. EPA, 1992c),
- *Estimating Risk from Contaminants Contained in Agricultural Fertilizers* (U.S. EPA, 1999).

EPA divided the available data into different categories of plants (e.g., protected fruit, exposed vegetable) to better capture the variability in produce consumed by both humans and livestock. Individual plant species were mapped to plant categories according to the methodology in outlined in U.S. EPA (1999). In some cases, some plants were mapped to multiple categories (e.g., corn to both grain and protected vegetable). When multiple data points were available for a given plant species, the values were averaged to prevent biasing the BCF toward those with more data. BCFs were then calculated for each plant category as the geometric mean of the values for each species in line with the recommended approach to calculate aquatic BCFs (U.S. EPA, 2016a). When no data were available from the three data sources, EPA used BCFs previously calculated in ORNL (1984). Due to the effort involved in compiling these data, values were only assembled in this document for the constituents carried forward for the screening analysis.

Animal Products

BCFs are used to estimate the amount of constituent mass that may accumulate from the water body (from dissolved and dissolved/suspended concentrations, respectively) into fish tissue. Biotransfer factors (BTFs) were used to estimate the amount of constituent mass that may accumulate from bulk soil and plant matter into beef and milk. Calculated BCFs and BTFs were drawn from the available literature:

- Primary literature: These are generally papers that may either focus on a single chemical (i.e., USFWS, 1989; Kumada et al., 1973; Lemly, 1985; Murphy et al., 1978) or contain data on multiple chemicals (i.e., Barrows et al., 1980; U.S. EPA, 1993).
- EPA databases/publications: These included ECOTOX (U.S. EPA, 2018) and the Mercury Report to Congress (U.S. EPA, 1997a).
- Other government databases/publications: These included Oak Ridge National Laboratory (ORNL), ATSDR and the Hazardous Substances Data Bank.

When sufficient data were available, separate BCFs were used for different fish trophic levels. Where data for only one trophic level of fish were available, those data were used as a surrogate for the other trophic level. Where data were only available for whole fish, those were used as a surrogate for filet. Whole fish values from the correct trophic level were preferred as surrogates to filet values from the other trophic level. So, given a TL3 whole fish value and a TL4 filet value, the TL3 whole value would be preferred to the TL4 filet as a surrogate for TL3 filet.

Table B-10 lists all of the chemical-specific parameters collected for this analysis, presented in alphabetical order based on the name of the constituent. In some instances, adequate data was not available on one or more of the parameters for a given constituent. In these cases, the analysis could not quantitatively consider exposures through the associated pathway. When the reference is a compilation, the original paper from which the value was drawn is also listed, if available. BCFs were calculated using data maintained in the EPA's FGD Gypsum Database. Using the soil to crop linkage table in the database, BCF values were calculated by dividing the crop concentration by the soil concentration reported in the literature.

Table B-10. Bioconcentration and Biotransfer Factors

Parameter	Value	Reference	Comment
Aluminum			
$\mathrm{BCF}_{\mathrm{T3}}$	3.6E+01	ECOTOX Cleveland, et al. (1991)	T4 whole fish was used for T3 filet
$\mathrm{BCF}_{\mathrm{T4}}$	3.6E+01	ECOTOX Cleveland, et al. (1991)	T4 whole fish was used for T4 filet
Antimony			
$\mathrm{BCF}_{\mathrm{T3}}$	0	Barrows et al., 1980	T3 whole fish (sunfish) was used for T3 filet.
$\mathrm{BCF}_{\mathrm{T4}}$	0	Barrows et al., 1980	T3 whole fish (sunfish) was used for T4 filet.
Arsenic			
$\mathrm{BTF}_{\mathrm{beef}}$	2.0E-03	ORNL (1984)	—
$\mathrm{BTF}_{\mathrm{milk}}$	6.0E-05	ORNL (1984)	—
$\mathrm{BCF}_{\mathrm{T3}}$	4.0E+00	Barrows et al., 1980	T3 whole fish (sunfish) used for T3 filet.
$\mathrm{BCF}_{\mathrm{T4}}$	4.0E+00	Barrows et al., 1980	T3 whole fish (sunfish) was used for T4 filet.
$\mathrm{BCF}_{\mathrm{Exfruit}}$	6.5E-03	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
$\mathrm{BCF}_{\mathrm{Exveg}}$	2.3E-02	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
$\mathrm{BCF}_{\mathrm{Forage}}$	6.4E-02	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
$\mathrm{BCF}_{\mathrm{Grain}}$	2.1E-02	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
$\mathrm{BCF}_{\mathrm{Profruit}}$	6.5E-03	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
$\mathrm{BCF}_{\mathrm{Proveg}}$	1.2E-02	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
$\mathrm{BCF}_{\mathrm{Root}}$	1.2E-02	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
$\mathrm{BCF}_{\mathrm{Silage}}$	6.4E-02	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
Barium			
$\mathrm{BCF}_{\mathrm{T3}}$	1.3E+02	ATSDR Hope, 1996	T3 whole fish (sunfish) was used for T3 filet.
$\mathrm{BCF}_{\mathrm{T4}}$	1.3E+02	ATSDR Hope, 1996	T3 whole fish (sunfish) was used for T4 filet.
Beryllium			
$\mathrm{BTF}_{\mathrm{beef}}$	1.0E-03	ORNL (1984)	—
$\mathrm{BTF}_{\mathrm{milk}}$	9.0E-07	ORNL (1984)	—
$\mathrm{BCF}_{\mathrm{T3}}$	1.9E+01	Barrows et al., 1980	T3 whole fish (sunfish) was used for T3 filet.
$\mathrm{BCF}_{\mathrm{T4}}$	1.9E+01	Barrows et al., 1980	T3 whole fish (sunfish) was used for T4 filet.
$\mathrm{BCF}_{\mathrm{Exfruit}}$	1.5E-03	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
$\mathrm{BCF}_{\mathrm{Exveg}}$	1.0E-02	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
$\mathrm{BCF}_{\mathrm{Forage}}$	4.6E-03	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
$\mathrm{BCF}_{\mathrm{Grain}}$	1.5E-03	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
$\mathrm{BCF}_{\mathrm{Profruit}}$	1.5E-03	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
$\mathrm{BCF}_{\mathrm{Proveg}}$	1.5E-03	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
$\mathrm{BCF}_{\mathrm{Root}}$	1.0E-02	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
$\mathrm{BCF}_{\mathrm{Silage}}$	4.6E-03	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.

Table B-10. Bioconcentration and Biotransfer Factors

Parameter	Value	Reference	Comment
		Boron	
BTF_{beef}	8.0E-04	ORNL (1984)	—
BTF_{milk}	1.5E-03	ORNL (1984)	—
BCF_{T3}	—	—	—
BCF_{T4}	—	—	—
$BCF_{Exfruit}$	2.0E+00	ORNL (1984)	—
BCF_{Exveg}	4.0E+00	ORNL (1984)	—
BCF_{Forage}	4.2E+00	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Grain}	6.1E-01	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
$BCF_{Profruit}$	2.0E+00	ORNL (1984)	—
BCF_{Proveg}	2.0E+00	ORNL (1984)	—
BCF_{Root}	4.0E+00	ORNL (1984)	—
BCF_{Silage}	4.2E+00	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
		Cadmium	
BTF_{beef}	5.5E-04	ORNL (1984)	—
BTF_{milk}	1.0E-03	ORNL (1984)	—
BCF_{T3}	2.7E+02	Kumada et al., 1972	T3 whole fish (rainbow trout) used for T3 and T4 filet. Geomean of multiple values.
BCF_{T4}	2.7E+02	Kumada et al., 1972	T3 whole fish (rainbow trout) was used for T4 filet. Geometric mean of multiple values.
$BCF_{Exfruit}$	5.1E-02	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Exveg}	5.5E-01	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Forage}	2.0E-01	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Grain}	8.4E-02	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
$BCF_{Profruit}$	5.1E-02	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Proveg}	7.2E-02	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Root}	1.3E-01	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Silage}	2.1E-01	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
		Chloride	
BCF_{T3}	—	—	—
BCF_{T4}	—	—	—
		Chromium	
BTF_{beef}	5.5E-03	ORNL (1984)	—
BTF_{milk}	1.5E-03	ORNL (1984)	—
BCF_{T3}	6.0E-01	U.S. EPA, 1993	T4 filet was used for T3 filet.
BCF_{T4}	6.0E-01	U.S. EPA, 1993 derived from Buhler et al.,1977 and Calamari et al., 1982	—
$BCF_{Exfruit}$	3.3E-04	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Exveg}	8.4E-04	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Forage}	2.8E-03	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Grain}	2.1E-04	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
$BCF_{Profruit}$	3.3E-04	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Proveg}	3.3E-04	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Root}	8.1E-04	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Silage}	2.8E-03	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
		Cobalt	
BCF_{T3}	—	—	—
BCF_{T4}	—	—	—

Table B-10. Bioconcentration and Biotransfer Factors

Parameter	Value	Reference	Comment
Copper			
BCF_{T3}	0	U.S. EPA (1993)	—
BCF_{T4}	0	U.S. EPA (1993)	—
Iron			
BCF_{T3}	1.9E+01	ECOTOX Preez et al., 1993	Species is banded bream
BCF_{T4}	1.9E+01	HSDB Nakamoto and Hassler, 1992	T3 filet (banded bream) was used for T4 filet
Lead			
BCF_{T3}	4.6E+01	U.S. EPA (1993)	T3 whole fish (bluegill) was used for T3 filet
BCF_{T4}	4.6E+01	U.S. EPA (1993)	T3 whole fish (bluegill) was used for T4 filet
Manganese			
BCF_{T3}	4.0E-01	ECOTOX Litzke and Hubel, 1993	Species was common carp
BCF_{T4}	2.0E-01	ECOTOX Litzke and Hubel, 1993	Species was rainbow trout
Mercury			
BTF_{beef}	6.0E-03	Calculated from U.S. EPA, 1997a	Converted from dry to fresh weight assuming a 70 percent moisture content in beef (US. EPA, 2005d)
BTF_{milk}	2.6E-03	Calculated from U.S. EPA, 1997a	Converted from dry to fresh weight assuming a 87 percent moisture content in milk (US. EPA, 2005d)
BCF_{T3}	1.6E+06	U.S. EPA, 1997a	Methyl mercury
BCF_{T4}	6.8E+06	U.S. EPA, 1997a	Methyl mercury
$BCF_{Exfruit}$	3.1E-02	Gypsum Database, U.S. EPA, 1992; 1999	Divalent mercury. Geomean of multiple crop types.
BCF_{Exveg}	1.4E-01	Gypsum Database, U.S. EPA, 1992; 1999	Divalent mercury. Geomean of multiple crop types.
BCF_{Forage}	4.6E-01	Gypsum Database, U.S. EPA, 1992; 1999	Divalent mercury. Geomean of multiple crop types.
BCF_{Grain}	6.6E-02	Gypsum Database, U.S. EPA, 1992; 1999	Divalent mercury. Geomean of multiple crop types.
$BCF_{Profruit}$	3.1E-02	Gypsum Database, U.S. EPA, 1992; 1999	Divalent mercury. Geomean of multiple crop types.
BCF_{Proveg}	2.1E-02	Gypsum Database, U.S. EPA, 1992; 1999	Divalent mercury. Geomean of multiple crop types.
BCF_{Root}	1.4E-02	Gypsum Database, U.S. EPA, 1992; 1999	Divalent mercury. Geomean of multiple crop types.
BCF_{Silage}	4.6E-01	Gypsum Database, U.S. EPA, 1992; 1999	Divalent mercury. Geomean of multiple crop types.
Molybdenum			
BCF_{T3}	4.0E+00	USFWS, 1989	T4 filet (rainbow trout and steelhead trout) was used for T3 filet
BCF_{T4}	4.0E+00	USFWS, 1989	Geometric mean of multiple values. Species were rainbow trout and steelhead trout.
Nickel			
BCF_{T3}	8.0E-01	U.S. EPA, 1993 derived from Calamari et al., 1982	T4 filet was used as a surrogate for T3 filet
BCF_{T4}	8.0E-01	U.S. EPA, 1993 derived from Calamari et al., 1982	—
Selenium			
BTF_{beef}	1.5E-02	ORNL (1984)	—
BTF_{milk}	5.9E-03	ORNL (1984)	—
BCF_{T3}	4.9E+02	Lemly, 1985	Species were threadfin shad and blueback herring. *
BCF_{T4}	1.7E+03	Lemly, 1985	Species were threadfin shad and blueback herring. *
$BCF_{Exfruit}$	2.4E-02	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Exveg}	1.2E-02	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Forage}	1.6E-01	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Grain}	3.0E-02	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
$BCF_{Profruit}$	2.4E-02	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Proveg}	1.6E-02	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Root}	3.4E-02	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Silage}	1.6E-01	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.

Table B-10. Bioconcentration and Biotransfer Factors

Parameter	Value	Reference	Comment
Strontium			
BCF_{T3}	9.5E+00	ECOTOX Aleksanyan et al., 1978	Species was common carp
BCF_{T4}	9.5E+00	ECOTOX Aleksanyan et al., 1978	T3 filet (common carp) was used for T4 filet
Thallium			
BTF_{beef}	4.0E-02	ORNL (1984)	—
BTF_{milk}	2.0E-03	ORNL (1984)	—
BCF_{T3}	3.4E+01	Barrows et al., 1980	T3 whole fish (sunfish) was used for T3 filet
BCF_{T4}	1.3E+02	U.S. EPA, 1993 derived from Zitko et al., 1975	—
$BCF_{Exfruit}$	4.0E-04	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Exveg}	4.0E-03	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Forage}	6.5E-01	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Grain}	4.0E-04	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
$BCF_{Profruit}$	4.0E-04	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Proveg}	4.0E-04	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Root}	4.0E-03	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
BCF_{Silage}	6.5E-01	Gypsum Database, U.S. EPA, 1992; 1999	Geometric mean of multiple crop types.
Vanadium			
BCF_{T3}	3.2E+01	ECOTOX Bell et al., 1980	T4 whole fish was used for T3 filet
BCF_{T4}	3.2E+01	ECOTOX Bell et al., 1980	T4 whole fish was used for T4 filet
Zinc			
BCF_{T3}	3.5E+02	Murphy et al., 1978	T3 whole fish (bluegill) was used for T3 filet. Geomean of multiple values.
BCF_{T4}	3.5E+02	Murphy et al., 1978	T3 whole fish (bluegill) was used for T4 filet. Geomean of multiple values

* In Lemly's paper on selenium, the BCFs are reported in L/g, but based on other data in the paper, the units are clearly actually L/kg.

- ECOTOX Wright, 1977 means the value was obtained from ECOTOX, which cites Wright (1977).
- ATSDR Hope, 1996" means the value was obtained from ATSDR, which cites Hope (1996).

Table B-11 presents additional factors used to calculate the accumulation of constituents in plants and animal. Values were all drawn from the EPA guidance documents, with the exception of the fraction of soil and forage consumed by cows raised for beef and milk. These values were set to 50 percent to reflect that, under pasturing conditions, cattle would not be allowed to graze in fields. The growing season is assumed to be approximately half the year.

Table B-11. Plant and Animal Exposure Factor Data Used in Modeling: Constants

Variable		Value	Units	Citation
Moisture Adjustment Factor (MAF)	Exposed Fruit	85	%	U.S. EPA (1997b)
	Exposed Vegetable	92		
	Protected Fruit	90		
	Protected Vegetable	80		
	Root Vegetable	87		

Table B-11. Plant and Animal Exposure Factor Data Used in Modeling: Constants

Variable		Value	Units	Citation
Fraction of Media Contaminated (f)	Forage	50	%	Assumption
	Grain	100		
	Silage	100		
	Soil	50		
Beef Cattle Ingestion Rate (Q)	Forage	8.8	$kg_{(DW)}$/day	U.S. EPA (2005d)
	Grain	0.47		
	Silage	2.5		
	Soil	0.39		
Dairy Cow Ingestion Rate (Q)	Forage	13.2	$kg_{(DW)}$/day	U.S. EPA (2005d)
	Grain	3.0		
	Silage	4.1		
	Soil	0.4		

B.2 Ecological Benchmarks

Ecological receptors may be exposed through the ingestion and/or direct contact with a contaminated medium, or they may be exposed through the food chain (i.e., consuming plants and prey that have, themselves, become contaminated). The ecological benchmarks used in this beneficial use evaluation are medium-specific constituent concentrations that are intended to protect ecological receptors from adverse effects. For each medium, EPA identified values based on hierarchies designed to identify values from sources that are current, transparent, publicly available, and that have been subjected to sufficient documentation and review. In instances where multiple benchmarks were available at the same level of the hierarchy (e.g., for different species), EPA selected the lowest benchmark to ensure that the most sensitive ecological receptors were protected. The following text summarize the hierarchy used for each environmental medium and the values selected. Values were last reviewed in January 2019.

B.2.1 Soil

The ecological soil benchmarks are intended to protect plants, soils invertebrates, avian wildlife or mammalian wildlife that may be chronically exposed to constituents in soil through direct contact, as well as the ingestion of biota that live in or on the soil. The selection hierarchy is discussed below, with the identified values presented in **Table B-12**.

Tier I

EPA Ecological Soil Screening levels (Eco-SSLs) were derived through the collaborative effort of a multi-stakeholder workgroup consisting of federal, state, consulting, industry, and academic participants led by EPA. These values have undergone a rigorous and extensive quality assurance protocol to ensure that these soil screening levels represent the current state of the science for each constituent. Eco-SSLs are derived based on toxicity reference values (TRVs), which represent a receptor-class (e.g., bird, mammal) level estimate of the soil concentration at which no adverse effects are observed from chronic exposure. Where data were available, TRVs were calculated for four classes of terrestrial receptors: plants, invertebrates, birds and mammals. For birds and mammals, EPA selected several different species representing different trophic levels and dietary habits, and selected the most protective (i.e., lowest) value as the Eco-SSL for that group for each constituent. A generic food-chain model was used to estimate the relationship between the concentration of the contaminant in soil and the resulting dose for the receptor.

Tier II

United States Department of Energy (DOE) reports developed by Oak Ridge National Laboratory (ORNL, 1997a,b) that calculate benchmarks for soil invertebrates and microbial processes. These values represent Effects Range-Low (ER-L), which are calculated as the lower 10th percentile of available lowest observed effects concentration (LOEC) data from laboratory data or field surveys. When the available data was based on a lethal concentration 50% (LC_{50}) or other endpoint that includes a 50% or greater reduction in survivorship, the value was divided by a factor of five.

Table B-12. Ecological Benchmarks for Soil (Terrestrial Receptors)

Constituent	CASRN	Value (mg/kg dry weight)	Receptor	Sources
Arsenic	7440-38-2	18	Plants	U.S. EPA (2005e)
Beryllium	7440-41-7	10	Plants	ORNL (1997b)
Boron	7440-42-8	0.5	Birds	ORNL (1997b)
Cadmium	7440-43-9	0.36	Mammals	U.S. EPA (2005f)
Chromium (III)	16065-83-1	26	Birds	U.S. EPA (2008c)
Chromium (VI)	18540-29-9	0.4	Invertebrates	ORNL (1997a)
Mercury	7487-94-7	0.1	Invertebrates	ORNL (1997a)
Selenium	7446-08-4	0.52	Plants	U.S. EPA (2007a)
Thallium	7440-61-1	1.0	Mammals	ORNL (1997a)

B.2.2 Surface Water

Surface water benchmarks were selected to protect animals in water bodies that may be exposed through direct contact with surface water or through ingestion of other biota that live in the water. EPA chose aquatic criteria appropriate for species living in the freshwater bodies because coastal waters were not modeled in this risk assessment. The hierarchy is as follows, with the selected values presented in **Table B-13**.

Tier I

EPA National Recommended Surface Water Quality Criteria provide chronic benchmarks based on Criterion Continuous Concentration (CCC). These values are estimates of the highest concentration of a chemical to which an aquatic community can be exposed indefinitely without resulting in an unacceptable effect. Values are only developed when sufficient data are available, with at least eight LC50s and three CVs. First, a Final Acute Value (FAV) is calculated, which represents the fifth percentile of the distribution of 48- to 96-hour LC50 values or equivalent median EC50 values. CCCs are the FAVs divided by the Final Acute-Chronic Ratio (FACR), which is the geometric mean of quotients of at least three LC50/CV.

Tier II

The Great Lakes Initiative Clearinghouse (GLI, 2013) database contains chronic values compiled from states and tribes from around the Great Lakes. When the minimum data requirements for calculating a CCC were not met, Secondary Continuous Concentrations (SCCs) were calculated in the same manner as CCCs with an adjustment factor applied based on the quantity of data available. In instances where multiple values were available from different sources, EPA selected values based on consideration of which were highest data quality, most recent, and lowest. Individual sources for the values in the database are listed after the citation to the GLI database.

Table B-13. Ecological Benchmarks for Surface Water (Freshwater Community)

Aluminum	7429-90-5	87	U.S. EPA (1988b)
Antimony	7440-36-0	190	OHEPA (2006a)
Arsenic	7440-38-2	150	U.S. EPA (1996)
Barium	7440-39-3	220	OHEPA (2006b)
Beryllium	7440-41-7	11	NYDEC (1984)
Boron	7440-42-8	7,200	MIDEQ (2011a)
Cadmium	7440-43-9	0.72	U.S. EPA (2016b)
Chloride	16887-00-6	230,000	U.S. EPA (1986)
Chromium (III)	16065-83-1	74	U.S. EPA (1996)
Chromium (VI)	18540-29-9	11	
Cobalt	7440-48-4	19	INDEM (1999a)
Copper	7440-50-8	9	U.S. EPA (2007b)
Iron	7439-89-6	1,000	U.S. EPA (1986)
Lead	7439-92-1	2.5	U.S. EPA (1985)
Manganese	7439-96-5	93	WIDNR (2005)
Mercury (total)	7439-97-6	0.77	U.S. EPA (1996)
Molybdenum	7439-98-7	800	INDEM (1998)
Nickel	7440-02-0	52	U.S. EPA (1995d)
Selenium	7782-49-2	1.5 (lentic) 3.1 (lotic)	U.S. EPA (2016c)
Strontium	7440-24-6	5,300	OHEPA (2006c)
Thallium	7440-28-0	6	INDEM (1999b)
Vanadium	7440-62-2	27	MIDEQ (2011b)
Zinc	7440-66-6	120	U.S. EPA (1996)

B.2.3 Sediment

Sediment benchmarks were selected to protect invertebrates that may be exposed to sediment through direct contact with sediment or through ingestion of other biota that live in the sediment.

EPA chose aquatic criteria appropriate for species living in the freshwater bodies because coastal waters were not modeled in this risk assessment. The hierarchy is summarized below, with the identified values presented in **Table B-14**.

Tier I

The Florida Department of Environmental Protection (FLDEP) developed Threshold Effects Concentrations (TECs) that identify concentrations below which harmful effects on sediment-dwelling organisms are unlikely to be observed (FLDEP, 2003). TECs were derived by taking the geometric mean of Effects Range-Low (ER-L) and Threshold Effects Level (TEL) data from various sources. ER-L are calculated as the lower 10th percentile of available lowest observed effects concentration (LOEC) data from laboratory data or field surveys. TELs are calculated as the geometric mean of the 15th percentile of the effects level data set and the 50th percentile of the no effects level data set. At least three separate sources were required to develop a TEC.

Tier II

When TECs could not be calculated, EPA identified individual values from the available literature:

- ER-Ls were drawn from the National Oceanic and Atmospheric Administrations (NOAA, 1991) and the Ontario Ministry of Environment and Energy (OMEE, 1993). ER-Ls are calculated as the lower 10th percentile in the distribution of biological effects data from matching biological and chemical laboratory data or field surveys.
- No Observed Effects Concentrations (NOECs) were drawn from Washington Department of Ecology (WDOE, 2013). No Observed Effects Concentration represent the highest concentration at which no effects were identified in laboratory studies.

Table B-14. Ecological Benchmarks for Sediment

Constituent	CASRN	Value (mg/kg $_{\text{dry weight}}$)	Receptor	Source
Antimony	7440-36-0	2	Invertebrates	NOAA (1991)
Arsenic	7440-38-2	9.8	Invertebrates	FLDEP (2003)
Barium	7440-39-3	20	Invertebrates	FLDEP (2003)
Cadmium	7440-43-9	1	Invertebrates	FLDEP (2003)
Chromium	7440-47-3	43.4	Invertebrates	FLDEP (2003)
Cobalt	7440-48-4	50	Invertebrates	OMEE (1993)
Copper	7440-50-8	31.6	Invertebrates	FLDEP (2003)
Iron	7439-89-6	20,000	Invertebrates	OMEE (1993)
Lead	7439-92-1	35.8	Invertebrates	FLDEP (2003)
Manganese	7439-96-5	460	Invertebrates	OMEE (1993)
Mercury	7487-94-7	0.18	Invertebrates	FLDEP (2003)
Nickel	7440-02-0	22.7	Invertebrates	FLDEP (2003)
Selenium	7782-49-2	11	Invertebrates	WDOE (2013)
Zinc	7440-66-6	121	Invertebrates	FLDEP (2003)

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Appendix C. Use Characterization

This appendix describes the approach used to characterize how and where FGD gypsum might be applied across the continental United States. This information was used in the evaluation to characterize the environmental conditions that may be present where this secondary material is applied, the proximity to nearby receptors, and the rate that constituents may be released into surrounding soil, ground water, surface water and air. In any given year, the extent of land used for crops and the rate of gypsum application will change based on a combination of agronomic and economic considerations. Therefore, this evaluation aims to capture the maximum extent of cropland that is available based on both past and present use.

C.1 Application Rate and Frequency

To estimate the total mass of gypsum that may be applied to agricultural land, EPA reviewed peer reviewed literature, government and industry reports, and state extension services. These sources provided a mixture of current practices, recommended rates, and theoretical ranges that could provide benefits. As a result, the rates reported sometimes varied considerably among different sources. EPA considered all the sources that reported benefits, though not all sources identified measurable benefits from the same application rates. Because of uncertainty about how practices may evolve if use of FGD gypsum becomes more widespread, EPA considered both demonstrated and theoretical rates. However, as a general criterion for all uses, the USDA National Resources Conservation Service stipulates that annual applications should not exceed 5 tons/acre (USDA, 2015a). Therefore, this was set as an upper bound on average annual applications.

C.1.1 Reduction in Soluble Phosphorus

Identified literature sources that address applications to reduce soluble phosphorus include Stout et al. (2000), Brauer et al. (2005), Watts and Torbert (2009), OSU-E (2011), Endale et al. (2014), Torbert and Watts (2014), Adeli et al. (2015) and USDA (2015a). Across the various soil types and local conditions studied, application rates ranged from as low as 0.5 tons/acre to as high as 4.5 tons/acre (OSU-E, 2011). Application frequency was not discussed widely in the literature for this use. USDA staff indicated that, for applications with manure, annual application was a reasonable assumption. For application on soils containing residual high phosphorus, application every other or third year may be a more reasonable assumption (Dick, 2015; Torbert, 2015).

Table C-1 presents a summary of the application rates and frequencies for this use of FGD gypsum. To obtain reasonable bounds on application rates, EPA separately grouped the moderate and highest values reported in each of the available source and calculated an average for each. Because USDA (2015a) placed a lower bound on applications of 1 ton/acre, EPA treated this application rate as the low end. Few studies reported rates lower than this and those that did were similar. EPA assumed annual applications for all locations under the assumption of manure application.

Table C-1. Application Rates and Frequency for Phosphorus Runoff

Modeled Frequency of Application	Mass Applied (tons/acre)		
	Low	Moderate	High
1 Year	1.0	1.5	3.0

C.1.2 Nutrient Source

Identified literature sources that address potential application rates for calcium or sulfur nutrient amendments include FIPR (1995), Grichar et al. (2002), UW-E, (2003), Sumner (2007), Chen et al. (2008) and OSU-E (2011). Across the various crops and soils studied, application rates as a sulfur source range from as low as 0.03 tons/acre (FIPR, 1995) to as high as 0.27 tons/acre (OSU-E, 2011). Application rates as a calcium source tend to be greater, ranging from as low as 0.04 tons/acre (FIPR, 1995) to 2 tons/acre (Chen et al., 2008). However, the highest application rates associated with peanuts and tomatoes are unlikely to be applied every year. State Extension Services from outside the southeast state that peanuts and tomatoes are grown in rotation to control disease and pests (UM/UW-E, 1991; TAMU-E, 2015; UC-IPM, 2013; APC, 2015). APC (2015) states that farmers are often successful using a two- or three-year rotation with either soybeans, cotton or corn. UC-IPM (2013) suggests growing tomatoes in a two- or three-year rotation.

Table C-2 presents a summary of the application rates and frequencies for this use of FGD gypsum. To obtain reasonable bounds on application rates, EPA separately grouped the lowest, moderate and highest values for both calcium and sulfur reported in each available source and calculated an average value for all three. The application frequency, particularly the higher calcium applications, is anticipated to be every two years at most.

Table C-2. Application Rates and Frequency for Nutrient Amendment

Modeled Frequency of Application	Mass Applied (tons/acre)		
	Low	Moderate	High
2 Year	0.2	0.8	1.7

C.1.3 Sodic Soils

The identified literature sources that address use in sodic soil include KSU-E (1992), PNE (2007), ASCE (2012), CSU-E, (2012) and USDA (2015a). Several sources provided equations to calculate necessary applications rates as a function of soil cation exchange capacity, initial and target sodium adsorption ratio, bulk density, and/or soil depth. Some sources applied these equations to realistic soil conditions and provided recommended application rates. These values ranged from a low of 1 ton/acre to a high of 10 tons/acre (PNE, 2007; OSU-E, 2011). Application frequency was not discussed widely in the literature for this use. USDA staff indicated that at the higher rates reported in the literature, applications may occur every 10 to 20 years (Chaney, 2016).

Table C-3 presents a summary of the application rates and frequency modeled for this use of FGD gypsum. Because of the relatively few numerical rates reported in the literature, EPA selected low, moderate and high rates based on those reported in OSU-E (2011). EPA assumed that applications would occur, on average, every 10 years.

Table C-3. Application Rates and Frequency for Sodic Soils

Modeled Frequency of Application	Mass Applied (tons/acre)		
	Low	Moderate	High
10 Years	1.0	5.0	10.0

C.1.4 Improve Infiltration

Literature sources that address potential application rates to improve infiltration include Ben-Hur et al. (1992), FIPR (1995), UC-E (1997), Yu et al. (2003), Amezketa et al. (2005), OSU-E (2011), Buckley and Wolkowski (2014) and USDA (2015a). Application rates reported in the literature ranged from as low as 0.01 tons/acre (FIPR, 1995) to as high as 4.5 tons/acre (OSU-E, 2011). For application frequency, multiple sources reported that applications should occur on an annual basis until the problem is remedied. Applications may then continue on a more intermittent basis as necessary afterwards (FIPR, 1995; UC-E, 1997; USDA, 2015a).

Table C-4 presents a summary of the application rates and frequency modeled for this use of FGD gypsum based on available data. To obtain reasonable bounds on application rates, EPA separately grouped the lowest, moderate and highest values reported in each of the available source and calculated an average for each. The range of values agree well with those recommended in USDA (2015a). Applications are assumed to occur on an annual basis for the full duration of application.

Table C-4. Application Rates and Frequencies

Modeled Frequency of Application	Mass Applied (tons/acre)		
	Low	Moderate	High
1 Year	0.25	0.75	2.0

C.1.5 Aluminum Toxicity

Literature sources that address application rates to address subsoil aluminum toxicity as a source of calcium or sulfur include Feldhake and Ritchey (1996), Miller and Sumner (1997), Toma et al. (1999), Farina et al. (2000a,b), Ritchey and Snuffer (2002), Chen et al. (2005) and Caires et al. (2011). Application rates reported in the literature ranged from as low as 0.4 tons/acre (Ritchey and Snuffer, 2002) to as high as 15.6 tons/acre (Toma et al., 1999). For application frequency, Caires et al. (2011) found that "…about 10% of Ca from gypsum was still adsorbed in the upper 10 cm of soil several years post application." Both Farina et al. (2000a) and Miller and Sumner (1997) characterized effects as lasting 10 years. When Toma et al. (1999) studied the longevity of effects from two earlier studies they found gypsum applications still effective after around 15 years. Limestone may also be applied along with the FGD gypsum to counteract the displacement of magnesium and potassium.

Table C-5 presents a summary of the application rates and frequency modeled for this use of FGD gypsum. To obtain reasonable bounds on application rates, EPA separately grouped the lowest and highest values reported in each of the available source and calculated an average for each. For this use, few sources reported a moderate application rate. Therefore, EPA used the "normal" application rate reported by OSU-E (2011) based on a review of the literature. These applications on an annual basis with applications spread out between with the same amount each time. Based on the frequencies reported by different sources, EPA assumed that applications would occur, on average, every 10 years.

Table C-5. Application Rates and Frequencies

Frequency of Application	Mass Applied (tons/acre)		
	Low	Moderate	High
10 Years	1.5	3.0	11.0

C.2 Field Boundaries

The size and location of agricultural fields provide important information about where and how much gypsum may be applied across the landscape. To estimate field boundaries, EPA initially relied on the Common Land Unit (CLU) and Cropland Data Layer (CDL) datasets:

- A CLU is the smallest unit of land associated with USDA management programs that has a permanent, contiguous boundary, a common land cover and land management, and both a common owner and producer. CLU boundaries are delineated from relatively stable features such as fence lines, roads, and/or waterways. The most recent CLU data is from 2008. The USDA has since restricted access to subsequent CLU polygons following of the enactment of The Food, Conservation, and Energy Act of 2008 (Public Law 110-234). The available shapefile does not contain any data about land use and so the shapefile alone is not sufficient for the purpose of defining field boundaries.
- The Cropland Data Layer (CDL) blends extensive field data and satellite information to produce a detailed raster map aggregated into the following ten generalized groups: corn, cotton, rice, soybeans, wheat, vegetables and ground fruit, orchards and vineyards, other grains, other row crops, and other crops. These data have been produced annually since 2008. For this evaluation, EPA used five years' worth of data from between 2010 and 2015 to capture the maximum extent of cropland. The available raster data is generated at a lower resolution than the CLU polygons, which adds greater uncertainty about exact borders. Therefore, the raster data alone is not sufficient for the purpose of defining field boundaries.

Figure C-1 presents the overlap of CLU and CDL data across the country. Areas covered in grey are those where CLU polygons are available (CLU data is not available for every state). The area covered in green are the total extent of cropland predicted by CDL raster data.

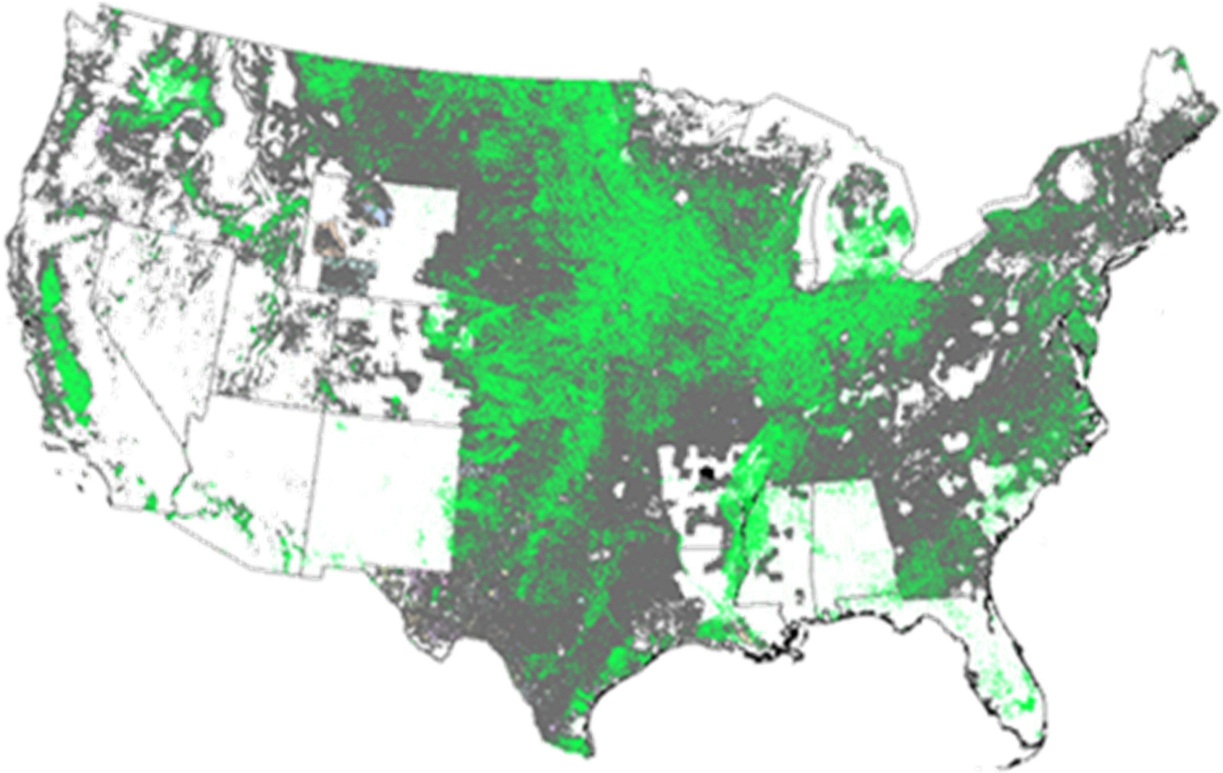

Figure C-1: Comparison of Coverage for CLU and CDL Datasets

For the purposes of this analysis, fields were divided in two categories based on the available data: those with and without CLU data. CLUs are provided on a county-by-county basis within each state. In some states, not all counties were delineated. In total, there were four states with no CLU data (AL, DE, FL and NM), 22 states for which all counties had CLU data, and 21 states that possessed a mixture of counties with and without CLU data. No field boundaries were developed for Alaska, California or Hawaii because these states fell outside the economic feasibility zone. EPA used similar approaches to delineate boundaries for fields in areas with and without CLU data. For areas with CLU data, the CDL raster data was overlain on top of the CLU polygons and used along with supplementary datasets to predict which CLU polygons were most likely to be cropland. The following data sources were used together with CDL and CLU data to refine the field boundaries:

- County boundary polygons from the U.S. Census cartographic boundary files. This layer was used so that data processing could be conducted and aggregated on a county-by-county basis.
- High-resolution National Hydrography Dataset (NHD) Plus dataset. These flowlines and water body polygons were used to identify the location of streams and lakes where cropland is unlikely.

- TIGER roadways by state (U.S. DOC, 2013). The roadway lines were used to identify areas likely to be roads, shoulders and ditches where cropland is unlikely.
- National Land Cover Dataset (NLCD) (USGS, 2011). These layers were used to identify areas of the landscape covered by impervious surfaces (e.g., building, parking lot) and forests where cropland is unlikely. Raster files were extracted by counties and converted to polygons.

Through trial and error paired with visual inspection against satellite imagery, EPA identified the combination of variables that best captured the extent of cropland. Once, these variables had been identified, EPA applied the same approach to areas without CLU data. In these areas, the CDL raster file was converted directly into field polygons and used along with supplementary datasets to predict areas most likely to be cropland. The datasets were managed as described below:

- County borders were used as a hard boundary on field area because CLU data is reported on a county-by-county basis. When field polygons crossed country lines, fields were split along that line and assigned to the county it is located. This process also allowed processing of field data in more manageable units (county rather than state).
- Polygons with a total area less than four acres were removed from the dataset. A review of the polygons overlain on satellite imagery found these areas most likely to reflect noise in the CDL datasets or small features (e.g., buildings) in the CLU dataset that would skew estimates of field size lower.
- CLU polygon with less than 50% overlap with the CDL raster was deleted. This value was selected based on trial and error to eliminate issues, such as bleed over of raster data from adjacent polygons. In these instances, the raster area may be continuous and larger than four acres, but only present around the periphery of a polygon.
- NHD flowlines were overlain on the polygons. If the flowline intersected with a CLU polygon, the CLU was assumed to be a water body along with any associated buffer areas and the entire polygon was removed. If the flowline intersected with a non-CLU polygon, the line was treated as a natural barrier between fields and simply subtracted out.
- Fields with both an area-to-perimeter ratio less than 60 and overlap with the CDL raster greater than 50% were merged together with adjacent polygons when the polygon was surrounded by potential cropland on more than one side. Visual inspection against satellite imagery found that these areas likely reflect terracing and other practices intended to prevent erosion.
- The shape and size of each remaining polygon was used to identify remaining areas that are unlikely to be cropland. EPA removed isolated polygons when the compactness, calculated as $4\pi\left(\frac{\text{Area}}{\text{Per}\quad\text{er}}\right.$, was < 0.25 or the area-to-perimeter ratio was < 30 for CLU polygons and < 15 for CDL polygons. Different values were used for CLU and CDL fields because of the blockier polygons formed by the CDL data. These metrics were used together to identify long and

narrow polygons more likely to reflect drainage ditches, buffer areas and other managed areas where crops are not grown.

- NHD water body polygons were used to capture portions of fields that contain open water. Portions of a polygon that overlapped with these areas were removed.
- TIGER roadway lines buffered on either side by 10 meters were used to capture roadway width, shoulders, rights of way and/or drainage ditches. Portions of a polygon that overlapped with these buffer areas were removed.
- NLCD polygons for impervious surfaces and forests were used to capture areas that are covered by either forests or impervious surfaces (e.g., buildings, sidewalks, parking lots) that are clearly not used as cropland. Portions of a polygon that overlapped with these areas were removed.

After applying these steps, EPA recalculated the area of the remaining polygons and removed those that had been reduced to less than four acres. The remaining polygon area was assumed to be entirely cropland. EPA conducted QA/QC on randomly selected counties from areas with and without CLU data to ensure that each data file had been correctly extracted, converted to polygons, and applied. Visual inspection of the fields overlain on satellite imagery was used to ensure that the resulting fields aligned with the apparent land use. Based on this review, EPA believes that the field boundaries provides reasonable estimate of field location and area.

This process resulted in over five million individual fields across 47 states (except CA, HI and AK) and 2,893 counties (out of 3,219). Not all of this land will be in active use in any given year as a result of economic incentives and crop rotation. EPA did not identify any data that could be used to reliably set a fraction of this land expected to be in active in any given year. Therefore, EPA relied on the delineated fields to define the maximum extent of cropland for this evaluation. In each model run, the fraction of the total cropland with FGD gypsum applied in any given year was allowed to vary anywhere from 0 to 100% based on a flat distribution.

C.3 Extent of Use

To delineate the geographic area over which FGD gypsum may be applied, EPA first defined the maximum area that it might be economical to apply the secondary material. This approach assumed that the compounding costs of purchase, transportation and application were the primary factors that determined whether FGD gypsum will be used. These costs do not consider whether any individual utility has the ability to meet the demands of the market in the surrounding area. Over a third FGD gypsum currently generated is diverted towards wallboard production (ACAA, 2018). Demand may exceed generating capacity if agricultural uses become widespread, resulting in smaller distribution areas around some utilities than considered in this evaluation. Impacts from imported sources of gypsum were assumed to be negligible.

Farmer willingness to pay for gypsum was estimated in 2011 to be between \$20 and \$25 per ton (OSU-E, 2011). Accounting for inflation, this range becomes \$21.59 to \$26.99 in 2015 dollars. Based on this calculation, it was assumed that the maximum farmers would be willing to pay for gypsum was $27 per ton. To account for the potential subsidies from USDA and other sources, EPA assumed farmers might be compensated for up to half the cost of gypsum-related costs, resulting in a total allowable cost of $54 per ton. EPA used this value to draw a distance around each source of gypsum that corresponds to this maximum cost.

To calculate this maximum distance, EPA summed the cumulative costs of purchase, transport and application. EPA first identified the sources of FGD and mined gypsum. Data on active electric utilities listed as generating FGD gypsum were drawn from the U.S. Energy Information Administration. A total of 115 utilities that generate FGD gypsum were identified. The average cost of purchase for FGD gypsum was estimated as $4.05 per ton (EIA, 2014). Data on active mines and quarries that produce mined gypsum were identified using data from Mine Safety and Health Administration. A total of 61 mines and quarries that produce mined gypsum were identified. The average cost of mined gypsum was estimated as $9.00 per ton (USGS, 2015). The cost of transport by truck was set as $0.19 per ton-mile (U.S. DOT, 2016). The transport distance was calculated as the closest straight-line distance from the source to the boundary of each county. The cost of application was based on the average field size for each county, calculated from data from the 2012 USDA Census of Agriculture based on total acreage and number of farms (USDA, 2012). The range of application rates identified for each use were considered to identify the furthest distance gypsum may be economical. The cost of application was estimated to be as the same as the cost of spreading lime at $4.39 per acre (Bongiovanni and Lowenberg-Deboer, 2000). Based on the field size and distance to closest gypsum source for each county boundary, a cumulative cost was calculated for both mined and FGD gypsum. If the cost of either material was below $54 per ton for a county, its use was considered economical. If both FGD and mined gypsum were economical for a given county, it was assumed farmers would choose the more affordable source.

Figure C-1 presents the maximum economic feasibility zone for FGD gypsum by county. Because no sources of FGD gypsum were identified in Alaska or Hawaii, these states were not considered in the analysis. A total of 87 out of 3,108 counties in the continental United States did not have sufficient information in the agricultural census and could not be assigned. Therefore, if these counties fell adjacent to one for which FGD gypsum was the most economical, then it was assumed that FGD gypsum would also be used in that county. However, if it was surrounded on all sides by an area without FGD gypsum, then it was assumed mined gypsum or no gypsum was the more economical choice.

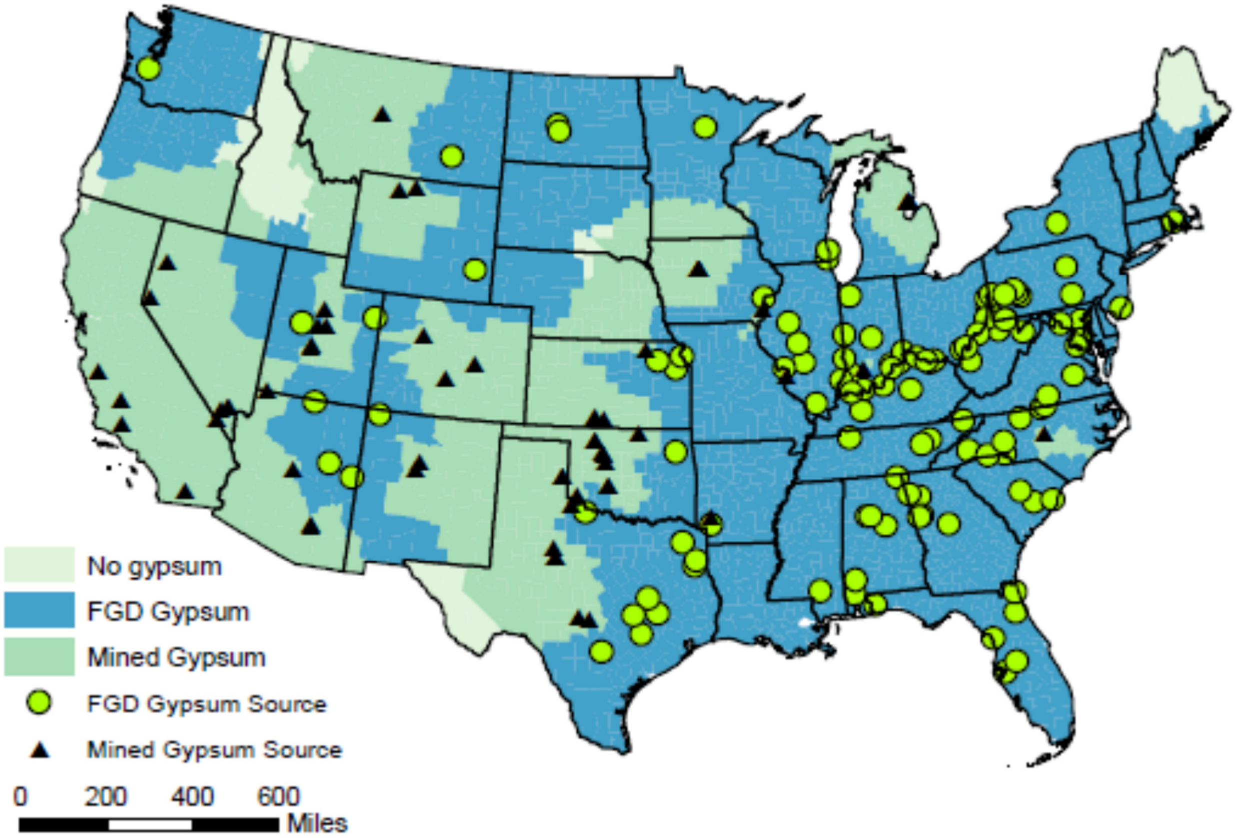

Figure C-1: Economic Feasibility Zones for FGD Gypsum

This economic feasibility zone was used as an initial boundary on the geographic extent that FGD gypsum might be applied. Because this boundary is based purely on economic feasibility, it does not indicate whether cropland is present or whether application of gypsum would provide a benefit in that area. Therefore, EPA used field boundaries together with soil and crop data to further refine the boundaries for each use. The data used to define the boundaries are described in the following subsections. These boundaries are intended to capture the widest range that this secondary material might conceivably provide a benefit and should not be construed to mean that FGD gypsum will be used over the entire areas shown.

C.3.1 Reduce Soluble Phosphorus

EPA reviewed the methods available to estimate the net amount of soluble phosphorus present in different regions of the country and concluded that the Nutrient Use Geographic Information System (NuGIS) developed by the International Plant Nutrition Institute provided the most current estimate (IPNI, 2012a,b). NuGIS used information from the USDA Census of Agriculture (USDA, 2012) to estimate the rate of phosphorus input, biological fixation, and removal by crops to obtain an annual net balance of phosphorus. Positive balances mean that there is more soluble phosphorus present than needed for crop production. A positive balance does not mean that nutrient runoff is or will become an issue, but it provides an indication of the areas where gypsum application would be most likely. The smallest relevant geographic unit available in NuGIS is a

HUC8. Therefore, EPA defined the use area as any HUC8 (and all HUC10 contained therein) with a net phosphorus greater than zero based on the most recent year data was available for that HUC8. **Figure C-2** depicts the geographic area over which FGD gypsum was evaluated to limit runoff of soluble phosphorus.

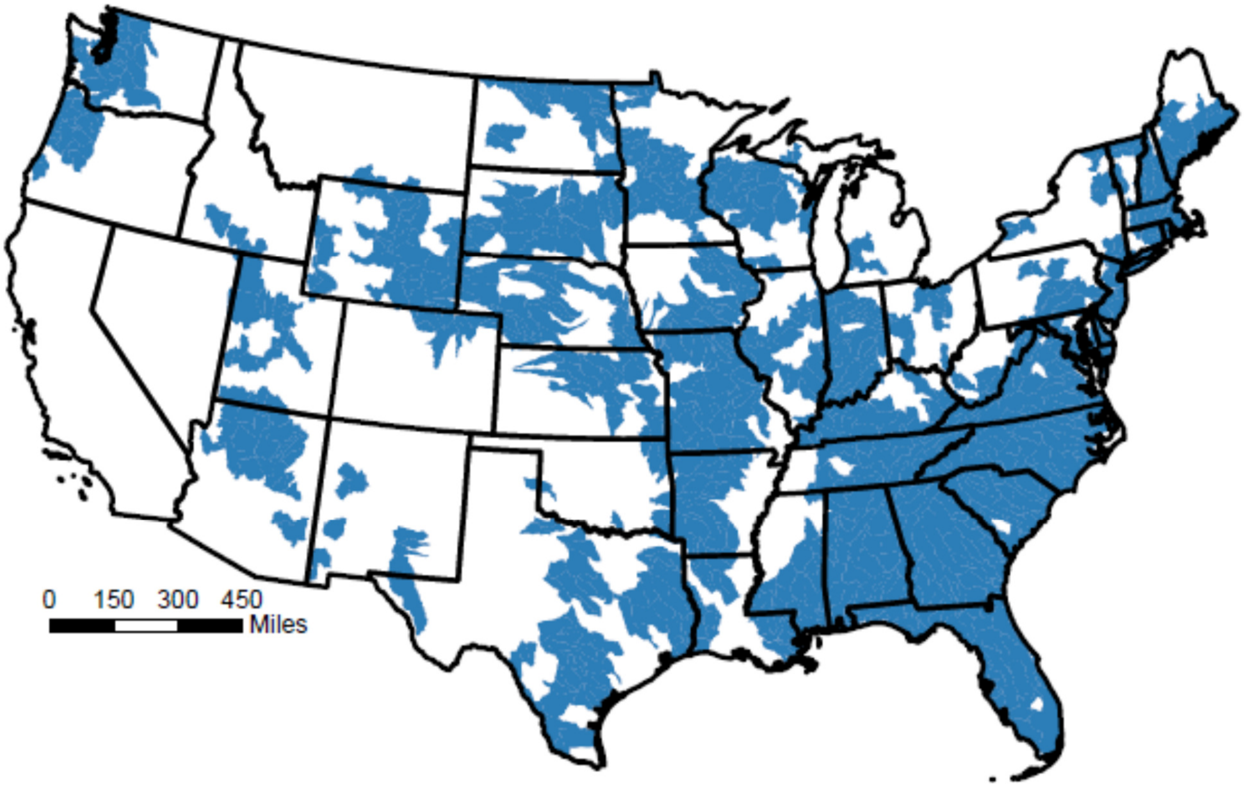

Figure C-2: Use Zone to Reduce Soluble Phosphorus

C.3.2 Nutrient Amendment

EPA consulted with USDA to identify the crops that exhibit a particular sensitivity to deficiencies of either calcium or sulfur in the soil (Boem et al., 2007; Yencho et al., 2008; DeSutter et al., 2011). Deficiencies in the soil are treated as distinct from deficiencies that result from physiological conditions or environmental stress. Based on consultation with USDA, the following crops were identified as those most likely to benefit from the application of FGD gypsum as a nutrient amendment (Chaney, 2016):

- **Calcium sensitive:** broccoli, cabbage, peanut, potato and tomato
- **Sulfur sensitive:** alfalfa, canola/rapeseed, cauliflower, mustard/kale greens, radish, sugar beet and turnip

EPA used the 2012 United States Census of Agriculture to identify counties in which these crops were grown. If more than 100 acres of any of the listed crops were grown in a county in the 2002, 2007 or 2012 census, then that county was included (USDA, 2002; 2007; 2012). This threshold was used to determine where widespread application of gypsum most likely drive risks would be most likely to occur. Because this beneficial use evaluation was organized around individual HUC, any HUC10 that overlapped with one of the identified counties was included. This could result in an overestimation of the area where gypsum is likely to be applied. **Figure C-3** depicts the geographic area over which FGD gypsum was evaluated for use as a nutrient amendment.

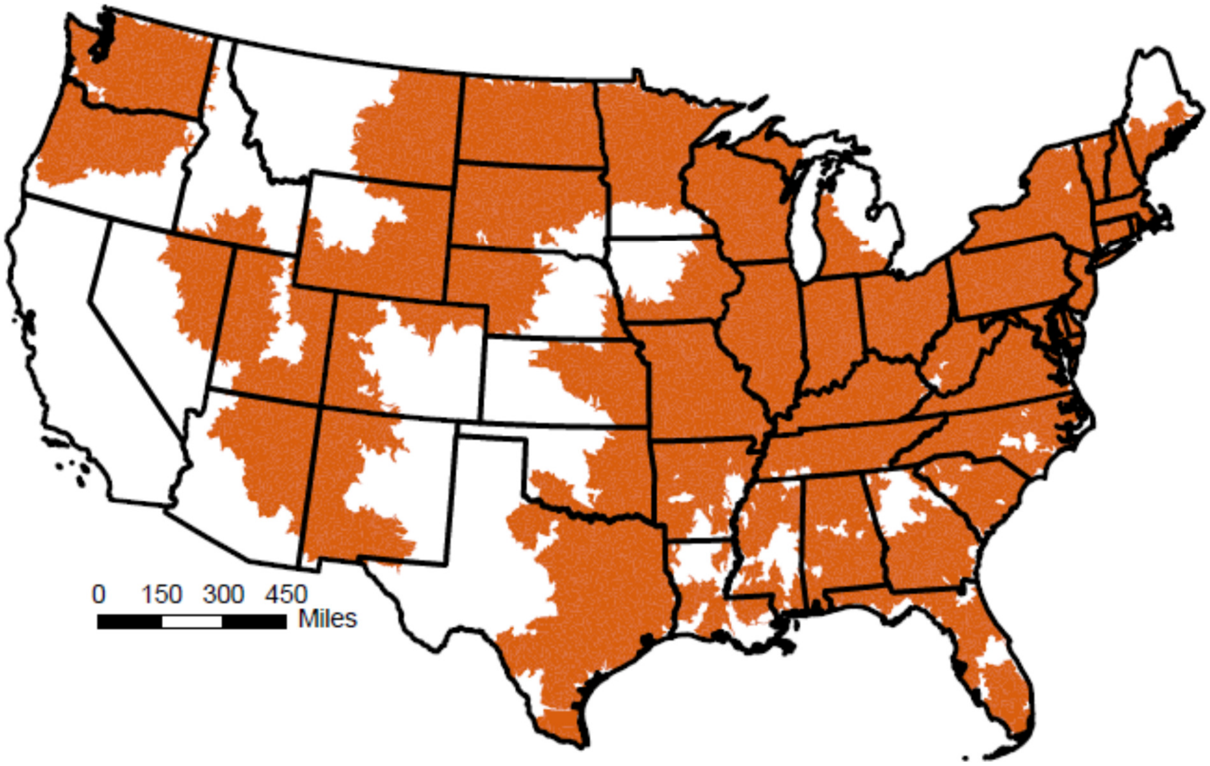

Figure C-3: Use Zone for Nutrient Amendment

C.3.3 Improve Infiltration

EPA used soil characterization data from the National Cooperative Soil Survey (NCSS) Soil Characterization Database (NCSS, 2016). This database includes site- and depth-specific chemical and physical soil characteristics. To determine regions that may be susceptible to surface crusting and reduced infiltration from calcium depletion, EPA used the measured cation exchange capacity (CEC) and percent of soil exchange sites saturated with basic cations (e.g., Ca^{+2}, Na^{+}). EPA calculated the fraction of the total exchange sites occupied by Ca^{+2} (Ca$_{\%}$) with data from Table "CEC_and_Bases" according following hierarchy, based on available data:

- Ca$_{\%}$ was calculated as the Mehlich extractable Ca^{+2} $(\text{Ca}_{\text{NH}_4})$ divided by total base saturation, measured at a pH of 7 (base$_{\text{sa}}$).
- If a sample had no measured value for base$_{\text{sa}}$, then Ca$_{\text{NH}_4}$ was divided by base saturation, measured at a pH of 8.2 (base$_{\text{ca}}$).
- If a sample had no measured value for either base$_{\text{sa}}$ or base$_{\text{ca}}$, then Ca$_{\text{NH}_4}$ was divided by total cation exchange capacity $(\text{CEC}_{\text{NH}_4})$.

- If a sample had no measured value for base_{sa} , base_{ca} or CEC_{NH_4}, then Ca_{NH_4} was divided by the sum of major extractable bases ($\text{Ca}_{NH_4} + \text{Mg}_{NH_4} + \text{Na}_{NH_4} + \text{K}_{NH_4}$).

USDA (2015b) indicates that a base saturation equal to 70% is the bottom of the balanced range. EPA therefore filtered for samples with $\text{Ca}_{\%} < 70\%$. If $\text{Ca}_{\%}$ was found to be below 70% in one or more soil sample within agricultural fields in a given HUC10, that HUC10 was included in the evaluation. There are a substantial number HUC10 for which no data are available. EPA also retained these HUC10 out of an abundance of caution. **Figure C-4** depicts the geographic area over which FGD gypsum was evaluated for use to improve infiltration.

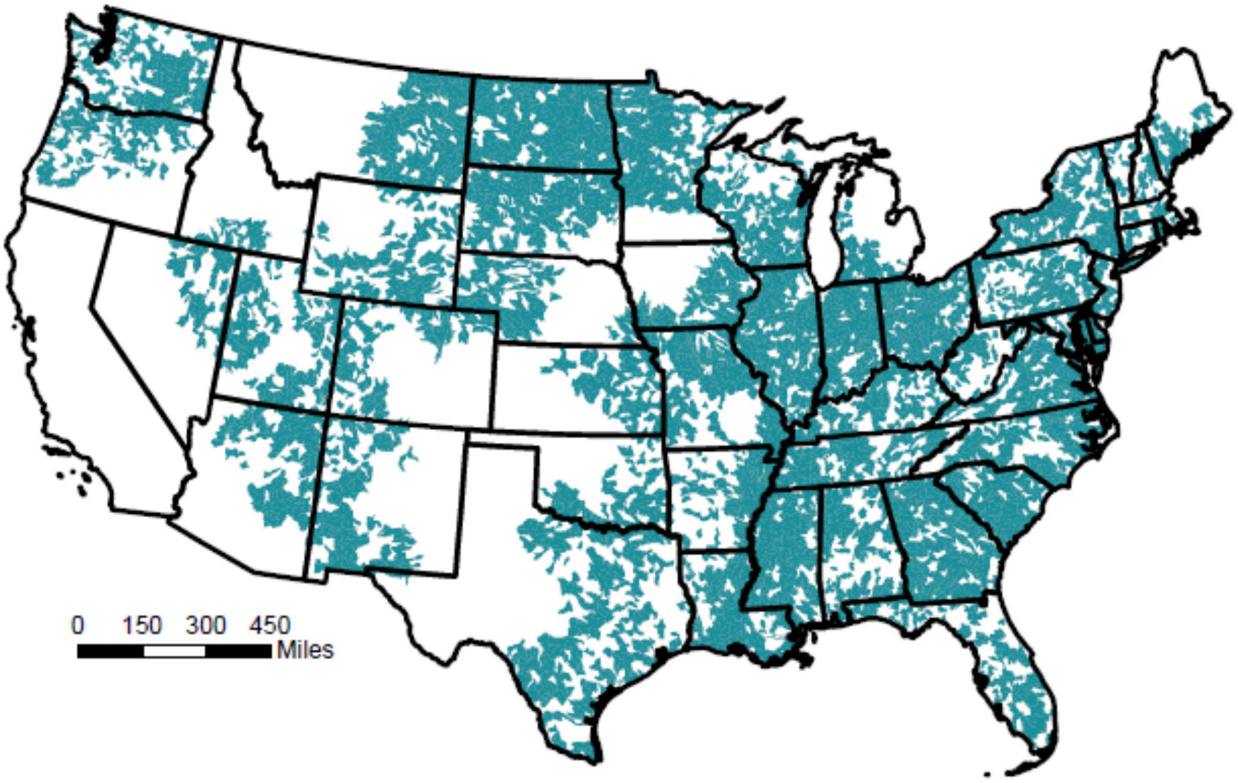

Figure C-4: Use Zone for Improved Infiltration

C.3.4 Ameliorate Sodic Soils

There is general agreement in the literature that a sodium adsorption ratio (SAR) above 13 results in sodic conditions harmful to plants, although levels below 13 have also been found to be harmful (ASCE, 2012). The Colorado State Extension recommends a final SAR below 10 after treatment with gypsum (CSU-E, 2012). EPA used data from the Soil Survey Geographic Database (USDA, 2016a) to calculate an average SAR over the top 36 inches of the soil column, intended to reflect the possible root zone. If a Soil Survey Geographic Database (SSURGO) map unit with a SAR greater than 10 overlapped with any of the agricultural field in a HUC10, the entire HUC10 was retained for further evaluation. **Figure C-5** depicts the geographic area over which FGD gypsum was evaluated for sodic soils.

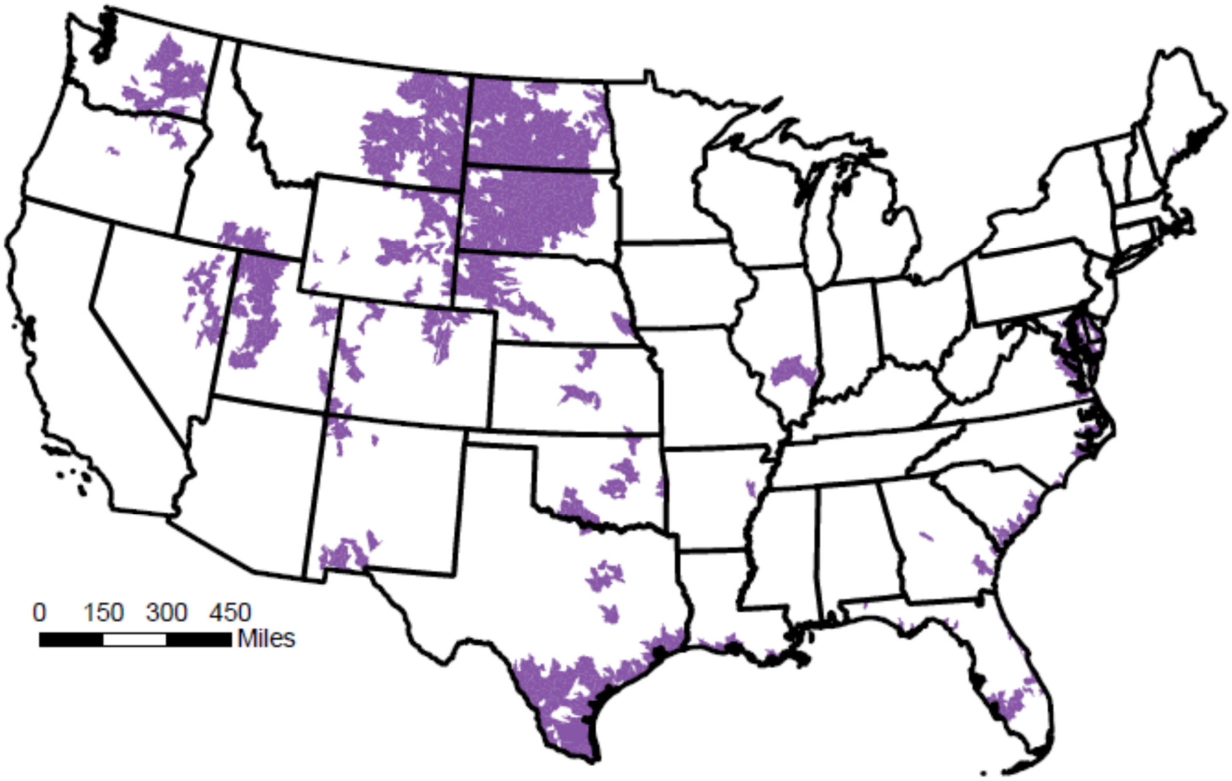

Figure C-5: Use Zone for Sodic Soils

C.3.5 Subsoil Aluminum Toxicity

The Natural Resource Conservation Service uses a taxonomic hierarchy to classify all soils in the U.S. into 11 soil orders that are subdivided into over 24,000 individual soil series. Davis (2016) advised that soils associated with subsoil Al toxicity "…would probably rest in the Ultisol or Oxisol soil order, those soils with sesquic and kaolinitic minerology and low pH subsoils." According to the NRCS Soil Series Extent Mapping Tool, Ultisols are found in the FGDG economic feasibility zone, but Oxisols are not (USDA, 2016b). EPA identified areas of potential use with data from the SSURGO, a digital soil survey that is the most detailed level of soil geographic data developed by the National Cooperative Soil Survey (USDA, 2016a). EPA used the SSURGO database to identify units in which at least 75% of the soils were Ultisols. If one of these map units overlapped with any of the agricultural field in a HUC10, the entire HUC10 was retained for further evaluation. **Figure C-6** depicts the geographic area over which FGD gypsum might be used to ameliorate aluminum toxicity.

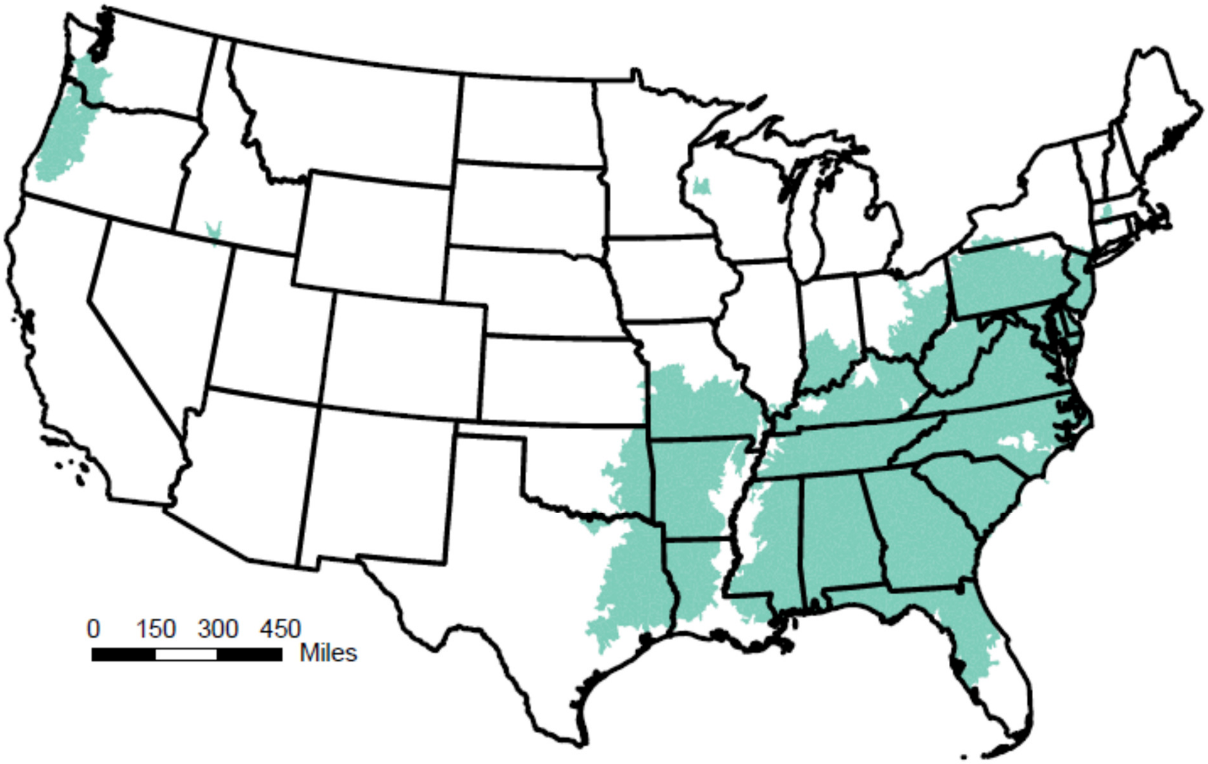

Figure C-6: Use Zone for Subsurface Aluminum Toxicity

C.4 Field Properties

EPA used the delineated field boundaries together with other databases to define the model inputs used to characterize the environmental media within and around the fields. The following text describes how the field boundaries were generally used to select data and additional criteria used to ensure the inputs were relevant and appropriate. Further discussion of how the assembled data were used in the full-scale models is discussed in **Appendix E (Probabilistic Modeling)**.

C.4.1 Distance to Receptors

Distance to receptor is a key factor in the evaluation of constituent fate and transport in ground water because greater distances allow for more dilution and attenuation prior to exposure. EPA used the National Hydrography Dataset Plus (NHDplus) flowlines. No publicly-available data is available for actual well locations. Instead, EPA used synthetic population data which estimates the most likely location of households in a given area.

Synthetic households and residents were placed to match the population distribution estimated by the LandScan USA 90-meter gridded population data set (Bhaduri et al., 2007), which distributes the US population across a grid of 90-meter square cells using a combination of satellite imagery and other geographic data layers, which include 2000 Census boundaries. The number of households is constrained by the population reported in a given 2000 Census block. Once the correct number of households is generated for a 90-meter grid cell, they are placed randomly within that 90-meter area. As a result, the synthetic population provides estimates of household locations at a finer resolution than is available from the 2000 Census data alone (Wheaton et al., 2009; Grefenstette et al., 2013). Although these household locations may not coincide with actual houses, the locations provide a representative distribution of likely home locations.

LandScan does not provide any information on which houses obtain water from private wells. Instead, EPA relied on data collected as part of the 1990 Census. For each census block, the percentage was calculated by dividing the total number of households that rely on drilled or dug wells by the total number of households. Each synthetic household location was then linked with the corresponding 1990 Census block group boundary so that the calculated percentages were transferred to each household. A SQL query was used to sum the number of households in each 1990 census block group and to calculate the total number of households on well in that block. The query then randomly selected this number of households on private within each block group and flagged each residence for use in the evaluation.

Figure C-7 provides an example of the relative location of fields used in the modeling, streams and synthetic population households. The households are coded by the source of drinking water. As can be seen in this figure, the placement of synthetic households tends to be concentrated near roadways, impervious surfaces and other indicators of human activity. In addition, in rural areas that tend to have the highest concentration of agricultural fields, the majority of homes are reported as relying on private wells as a source of drinking water. Therefore, there is minimal additional uncertainty introduced through the use of well data from the 1990 Census data.

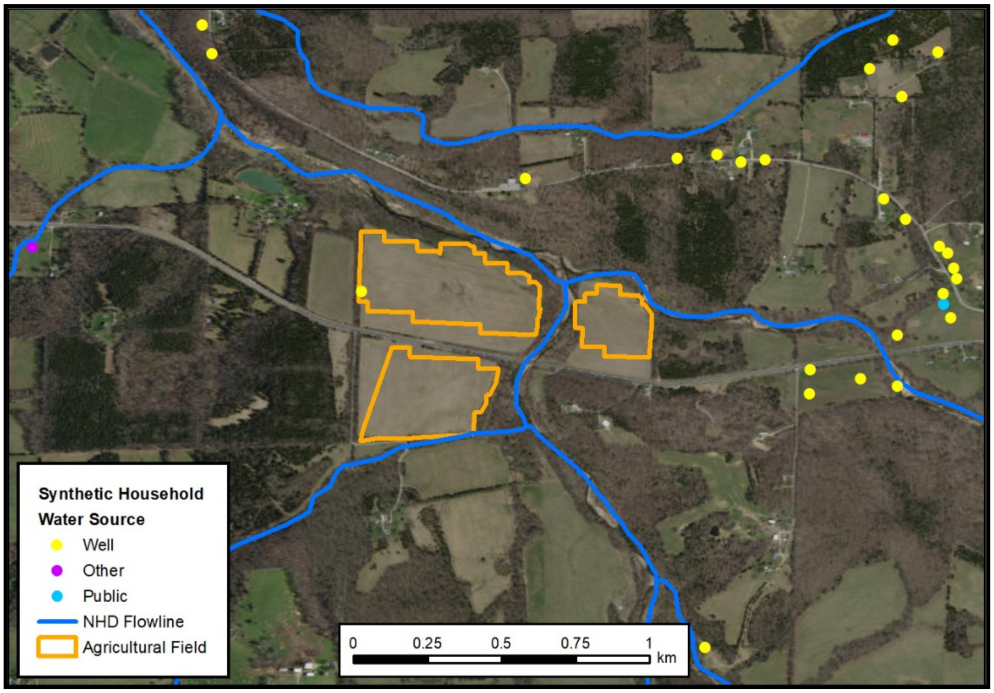

Figure C-7. Synthetic population locations by water source in the vicinity of agricultural fields.

This evaluation considered potential risks to highly exposed individuals, which are hypothetical receptors that reflect a upper bound on realistic exposures that might occur within the exposed population. To best capture these receptors, A GIS algorithm was applied to develop a distribution of distances from the each agricultural field to the nearest receptor (i.e., water body, household). Values were aggregated at the state level because it is believed that population mobility will not have a major impact on the overall distribution at this scale. Because the fate and transport is not necessarily limited by state boundaries, EPA allowed the nearest receptor to be located within an adjacent state for fields near these boundaries.

To develop distributions for nearest water body, both fields and NHD flowlines were projected to an equidistant projection to preserve distance. EPA calculated a straight-line distance from each field boundary to the nearest NHDplus flowline with the Near command in ArcGIS using the closest linear feature option. EPA aggregated distances from any HUC12 catchment that intersected the state border. The distribution of distances for headwater and mainstem streams were similar and so a single distribution was complied that included all water bodies, regardless of stream size. During review of the distributions for each state, a maximum distance was set at 1,609 meters (1 mile), chosen as the approximate 95th percentile of all measured distances. All distances greater than this maximum were capped at that value. The purpose of this maximum distance is to limit model run times and compounding uncertainty from longer distance model runs. A number of fields were found to be directly adjacent to water bodies, which may be a result of real-world conditions or because buffer areas were not identified when delineating field areas. This boundary condition has the potential to return anomalous results because there is no space for the infiltrating leachate to mix with the water table before discharge into the water body. Therefore, a setback distance of one meter was assigned to each of these fields. **Figure C-8** presents an example distributions for distance to nearest water body.

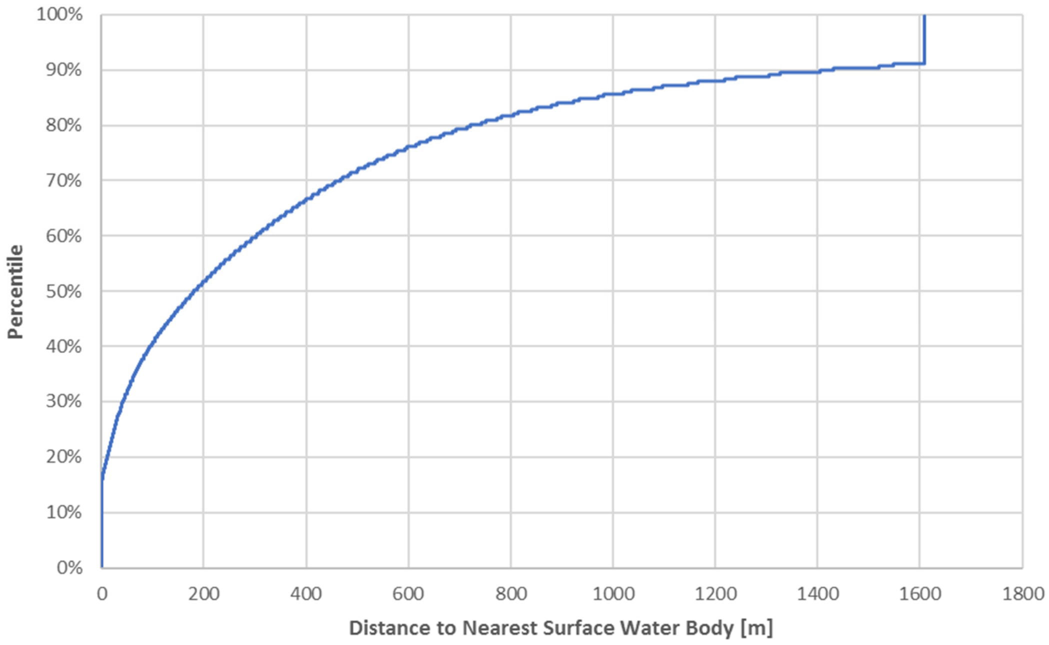

Figure C-8. Cumulative probability distribution of distance from the edge of agricultural fields to the nearest surface water body.

To calculate the distance to nearest well, both fields and the synthetic households were projected to an equidistant projection to preserve distance. EPA calculated a straight-line distance from each field boundary to the nearest household with the Near command in ArcGIS using the closest linear feature option. EPA aggregated distances from any residence within two miles of a state border. A maximum distance of 3,219 meters (2 miles) was established based on the 95th percentile of all distances to nearest households. All distances greater than this maximum were capped at that value. The purpose of this maximum distance is to limit model run times and compounding uncertainty from longer distance model runs. Because the placement of synthetic households is randomized, some were located within agricultural fields. In these instances, a default distance of 15 meters was used instead based on a minimum setback distance recommended to protect water supplies from agricultural runoff (U.S. EPA, 2002). **Figure C-9** presents an example distributions for distance to nearest residence.

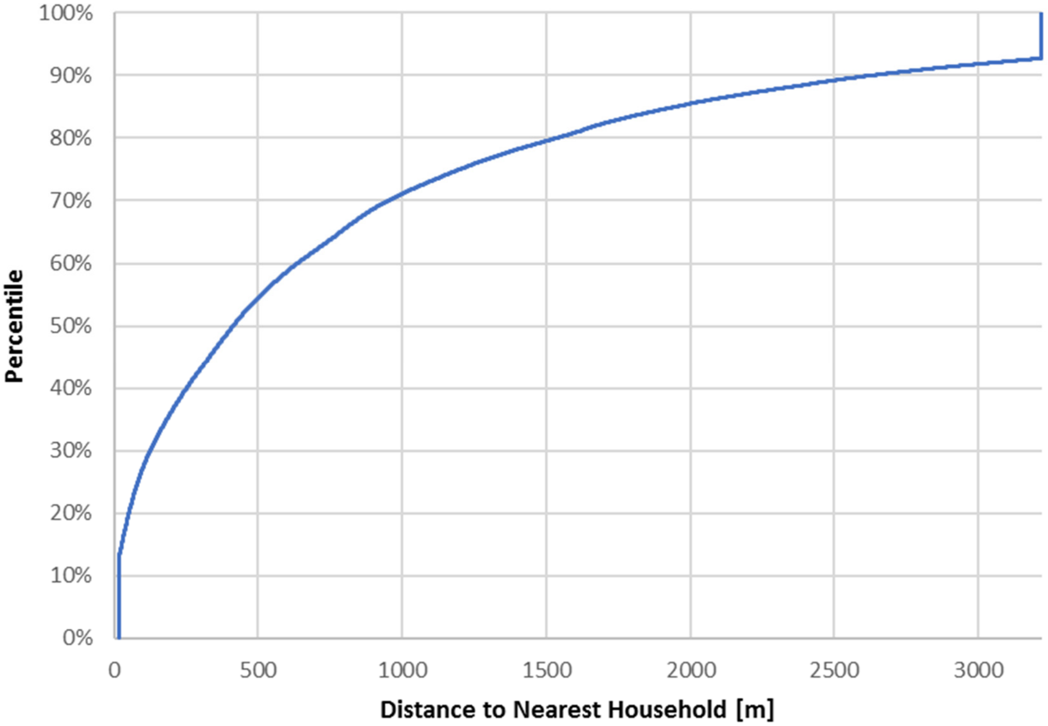

Figure C-9. Cumulative probability distributions for the distance from the edge of agricultural fields to the nearest ground water well.

C.4.1 Soil Properties

Soil properties are key factors in the evaluation of constituent fate and transport in ground water because they determine the extent to which contaminant can be released from and migrate through the soil. EPA used the Soil Survey Geographic (SSURGO) database to identify relevant soil pH, soil texture, bulk density and other inputs from within field boundaries. In instances where the soil pH reported within a field fell outside of the range of 5 to 8 considered in this evaluation, these values were filtered out of the ultimate distribution because it is unlikely that the soil would support commercial agriculture without application of other soil amendments, such as lime, to first adjust the pH. Filtering of these pH values did not affect data collected for any other variable. Further discussion on how the data from the SSURGO database were processed and incorporated into the model is discussed in **Appendix E (Probabilistic Modeling)**. **Figure C-10** presents a summary of the prevalence of different soil pH and textures used in the model runs based on all fields considered across the country. These distributions would vary for different uses of FGD gypsum.

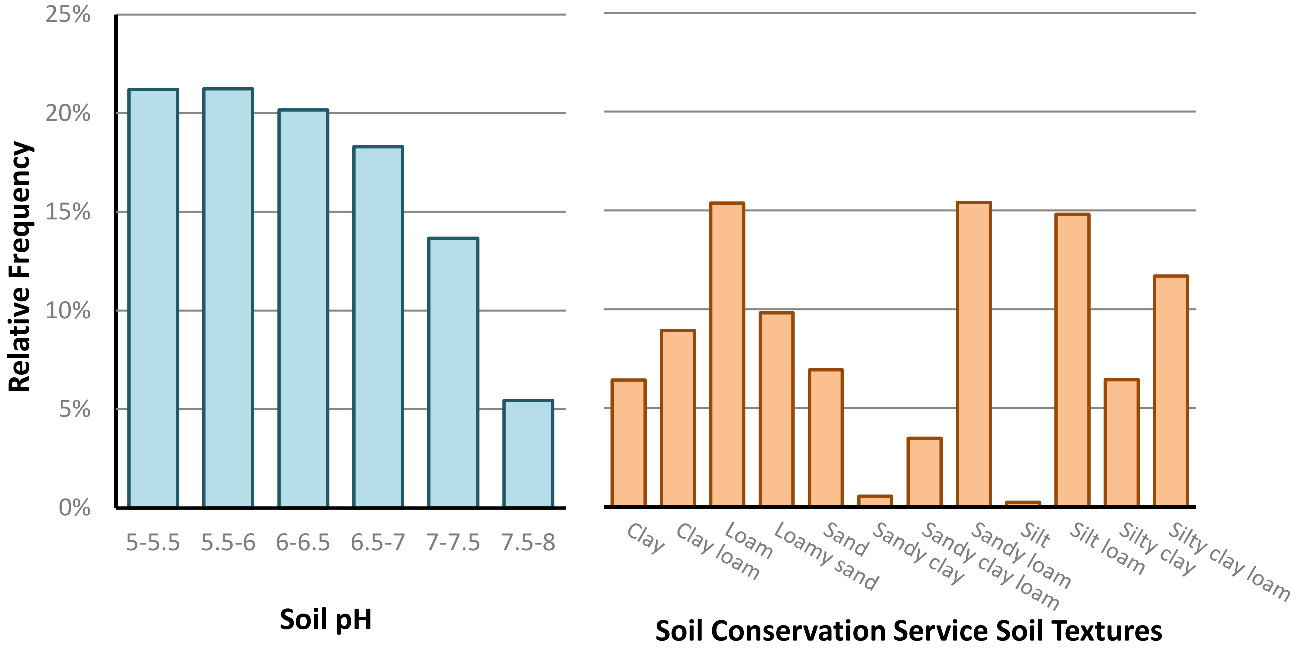

Figure C-10: National soil property distributions

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Appendix D. Screening Analysis

Modeling was performed to estimate ambient air mercury concentrations (for the inhalation pathway) and deposition rates (for the fish ingestion pathway) to evaluate potential exposures resulting from the volatilization of mercury from FGD gypsum applied to agricultural fields.

D.1 Volatilization Rate

For screening purposes, the mercury volatilization rate was set to 102.4 ng/m^2-hr. This value was calculated as the constant volatilization rate needed to deplete the 90th percentile bulk mercury concentration before the next annual application. This mercury emission rate was combined with air dispersion and deposition modeling results to estimate maximum, off-field ambient air concentrations and deposition rates used to characterize potential exposures posed to highly exposed individuals living in close proximity to the field.

D.2 Air Dispersion and Deposition Modeling

EPA conducted dispersion modeling to estimate ambient air concentrations and total combined, wet and dry vapor depositions rates for mercury. Modeling was performed with the American Meteorological Society/Environmental Protection Agency Regulatory Model (AERMOD; U.S. EPA, 2011, 2015) to produce results for the inhalation screen and to develop deposition rates to evaluate surface water impacts and estimate fish ingestion exposures.

Modeling was performed for a representative 427-acre farm scenario using five years of meteorological data. This representative FGD application scenario was formulated based on the U.S. EPA's Office of Pesticide Programs (OPP) FIRST model. This pesticide exposure assessment scenario was developed to characterize human exposures resulting from the ingestion of drinking water obtained from an index reservoir (U.S. EPA, 2008). The reservoir simulated by OPP modeling is an actual small drinking water reservoir located in Shipman, Illinois. Shipman City Lake is 13 acres in area, 9 feet deep, has a mean hydraulic residence time of 6 months, a watershed area of 427 acres, and a drainage area to capacity ratio (volume of water in the lake) of approximately 12. Under the current FGD gypsum evaluation, it is assumed that FGD gypsum is applied to the watershed (representing a crop field) for 100 years. Modeling was performed using meteorological and land use data from three representative National Weather Service (NWS) stations located in areas where FGD gypsum use is economically feasible. These locations, which represent a range of climatic conditions, included Chicago O'Hare, Charleston, S.C., and Seattle.

AERMOD modeling was conducting based on a unitized emission rate (e.g., 1 mg/m^2-s). The resulting air concentrations and deposition rates are called unitized air concentrations (e.g., µg/m^3 per unit emission rate of 1 mg/m^2-s) and unitized deposition rates (e.g., g/m^2-yr per unit emission rate of 1 mg/m^2-s). These are multiplied by the elemental mercury emission rate, along with appropriate conversion factors, to calculate the vapor-phase air concentrations and deposition rates. Modeling results for the three representative meteorological locations are presented in **Table D-1**. Considering both air concentration and total deposition outputs, Chicago O'Hare outputs were identified as the most conservative for use as input to the screening. As shown below in **Figure D-1,** Chicago O'Hare is also the NWS station associated with meteorological location where Shipman City Lake is located.

Table D-1. AERMOD Maximum Unitized Air Concentrations and Depositions Rates

Unitized Annual Average AERMOD Output	National Weather Station		
	Chicago O'Hare, IL	Charleston, S.C.	Seattle, OR
Air concentration from vapor phase ug/m^3 per $mg/(s\text{-}m^2)$	3.1E+04	3.1E+04	2.7E+04
Dry deposition from vapor phase [(g/m2/yr) per (mg/(s-m2))]	1.0	1.6	1.2
Wet deposition from vapor phase [(g/m2/yr) per (mg/(s-m2))]	1.4E-04	1.8E-04	2.5E-04
Total (wet and dry) deposition from vapor phase [(g/m2/yr) per (mg/(s-m2))]	1.0	1.6	1.2

Modeling was performed for elemental mercury using the constituent specific inputs shown in **Table D-2**. These parameters include diffusivity in air (Da) for the pollutant being modeled (cm^2/s), diffusivity in water (Dw) for the pollutant being modeled (cm^2/s), the cuticular resistance to uptake by lipids (rcl) for individual leaves (s/cm), and Henry's law constant ($Pa\text{-}m^3/mol$).These parameters are used by AERMOD in estimating dry and wet depositions for gaseous pollutants. Chronic exposures to elemental mercury (assumed to bioaccumulate as methyl mercury for fish ingestion exposures) were evaluated using outputs averaged over one year.

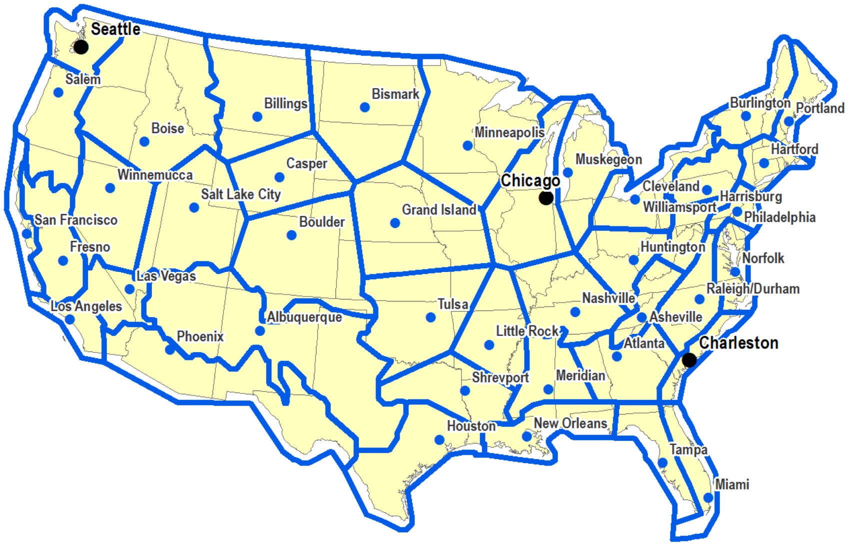

Figure D-1. Meteorological stations and regions.

Table D-2. Elemental Mercury Modeling Parameters

Parameter	Value	Reference
Diffusivity in air (Da) (cm2/s)	1.1E-02	U.S. EPA (2005b)
Diffusivity in water (Dw) (cm2/s)	3.0E-05	U.S. EPA (2005b)
Cuticular resistance to uptake by lipids (rcl) for individual leaves (s/cm)	1.0E+05	ANL (2002)
Henry's Law constant (Pa m3/mol)	7.2E+02	U.S. EPA (2005b)

Air concentrations and deposition rates were evaluated at a range of distances included in previous EPA analyses and models, such as the Multimedia, Multiple Exposure Pathway, Multiple Receptor Risk Assessment (3MRA) modeling system (U.S. EPA, 2003), the solvent wipes risk assessment (U.S. EPA, 2009), the CCR screening level analysis (U.S. EPA, 2014). The specific distances (0 m, 25 m, 50 m, 75 m, 150 m, 300 m, 500 m, 1,000 m, and 2,000 m) were selected to ensure complete coverage in the air and deposition estimates, particularly near the source of the emissions.

To capture national variability in the estimates, the air dispersion and dispersion modeling considered representative meteorological locations where FGD gypsum will likely be applied. The representative locations were selected from the 41 U.S. EPA Office of Water land application meteorological regions and locations. Because it was not feasible to run AERMOD (to obtain results for longer averaging times) for all of the 41 meteorological stations, EPA performed AERMOD modeling using meteorological and land use data for three locations where FGD gypsum could be applied. AERMOD outputs were generated for the range of receptor distances (0, 25 m, 50 m, 75 m, 150 m, 300 m, 500 m, 1,000 m, and 2,000 m) and included unitized annual average vapor air concentrations and yearly average wet and dry vapor deposition rates for elemental mercury. EPA then used these AERMOD outputs to estimate location specific ambient air concentrations and deposition rates for the chronic screening.

D.3 Air and Surface Water Impacts

The maximum ambient air concentrations were estimated using the constant emission rate of 102.4 ng/m^2-hr and the maximum, off-farm AERMOD vapor air concentration. The resulting maximum ambient air concentration was compared to the elemental mercury reference concentration, resulting in a screening ratio orders of magnitude below levels of concern.

Surface water impacts from wet and dry deposition of mercury vapor were also estimated using the constant assumed mercury volatilization rate and AERMOD deposition rates for elemental mercury. The dissolved mercury concentration was calculated using surface water equations presented in Chapter 5 and Appendix B of U.S. EPA's Human Health Risk Assessment Protocol (HHRAP) for Hazardous Combustion Facilities (U.S. EPA, 2005b). The maximum air concentration and total unitized wet and dry deposition rates of mercury from the vapor phase associated with the Chicago O'Hare NWS station presented in **Table D-1** were used with HHRAP equations to estimate the dissolved fraction of total mercury in the water body resulting from deposition and vapor phase diffusion from the air to a quiescent water body.

HHRAP Equations B-4-7, B-4-8, B-4-12 through B-4-24 (excluding B-4-14) and Equation 5-36c were used to estimate mass loadings and losses through the air-water interface and equilibrium mass partitioning between sediment, suspended and settled, and the dissolved phases. The index reservoir specific data was combined with constituent-specific information presented in **Table D-3** for dissolved mercury species, as inputs to the HHRAP equations. HHRAP default values were assumed in lieu of unknown site-specific data including the 85/15 apportionment assumption between divalent and methyl-mercury species. The dissolved surface water concentration was compared to the recreational and subsistence fish ingestion screening-level benchmarks and was found to be below a level of concern with screening ratios orders of magnitude below levels of concern.

Table D-3. Mercury Surface Water Modeling Parameters

Parameter	Value	Reference
Diffusivity in air (Da) (cm^2/s) for Divalent Mercury	5.2E-02	U.S. EPA (2014)
Diffusivity in water (Dw) (cm^2/s) for Divalent Mercury	1.8E-05	U.S. EPA (2014)
Henry's law constant ($atm{\cdot}m^3/mol$) for Divalent Mercury	7.1E-10	U.S. EPA (2014)
Suspended sediments partitioning Coefficient (Log Kd) (L/g) for Divalent Mercury	5.3	U.S. EPA (2005a), Table 5
Bed sediments partitioning Coefficient (Log Kd) (L/g) for Divalent Mercury	4.9	U.S. EPA (2005a), Table 4

D.4 References

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Appendix E: Probabilistic Analysis

The probabilistic analysis conducted for this evaluation considered risks that result from releases to soil, ground water and surface water. Modeling these pathways required calculating the flux of constituent mass to overland runoff and subsurface infiltration, as well as modeling the fate and transport each constituent within aquifers and surface water bodies. This appendix discusses the models and equations used to model concentrations in each media, as well as the data used in each.

E.1 Data Sources

The scale of data assembly, analysis and application in this evaluation was conducted at the watershed and sub-watershed scales, unless otherwise noted. Watersheds in the United States have been delineated by the U.S. Geological Survey (USGS) using a national standard hierarchical system based on surface hydrologic features and are classified into six types of hydrologic units. Watersheds and sub-watersheds are assigned the hydrologic unit code (HUC) of 10 and 12, respectively, corresponding to the number of digits in their unique identifiers. HUC10 and HUC12 will be used frequently hereafter to refer to these watershed and sub-watershed scales.

E.1.1 Soil Type

The primary data source for soil properties was the Soil Survey Geographic (SSURGO) database. SSURGO is a repository of nationwide soil properties collected by the National Cooperative Soil Survey over the last century (USDA, 2017). SSURGO data were collected at scales ranging from 1:12,000 to 1:63,360 and are linked to map unit polygons ranging between 1 and 10 acres. These map units provide the finest spatial resolution and span most of the conterminous United States. Soil attributes linked to these map unit polygons are stored within a relational database broken out by soil component and soil horizon. Each map unit contains data on the prevalence of each component and horizon within the map unit. Data extracted from SSURGO were evaluated and used at the HUC 10 scale. EPA extracted two types of data from SSURGO:

1. Measured (i.e., numerical) data are those that can be weighted by soil horizon and component. These data include pH, percent organic matter and percent silt. Measured data were extracted for the top 20 cm of the soil column (i.e., root zone) and weighted by the thickness of each horizon present to obtain a representative value for each soil component. Component values were then weighted by the relative prevalence of each component to obtain a representative value for the entire map unit.

2. Categorical data (i.e. non-numerical data) are those that can be characterized by a dominant type. These data include soil texture and hydrogeological group. First, the characteristic of the dominant horizon within each component was identified and assigned. Then, the characteristic of the dominant component was identified, and was assigned to the map unit.

Table E-1. Soil Texture Crosswalk

Detailed SSURGO Soil Texture	Basic SCS Texture	EPACMTP Soil Megatexture
Loamy Sand	Loamy Sand	Sandy Loam
Loamy Coarse Sand		
Loamy Fine Sand		
Loamy Very Fine Sand		
Sand	Sand	
Coarse Sand		
Fine Sand		
Very Fine Sand		
Sandy Loam	Sandy Loam	
Coarse Sandy Loam		
Fine Sandy Loam		
Very Fine Sandy Loam		
Silt Loam	Silt Loam	Silt Loam
Silt	Silt	
Loam	Loam	
Sandy Clay Loam	Sandy Clay Loam	
Clay Loam	Clay Loam	
Silty Clay Loam	Silty Clay Loam	Silty Clay Loam
Sandy Clay	Sandy Clay	
Silty Clay	Silty Clay	
Clay	Clay	

SCS = Soil Conservation Service

All tabular data (weighted or dominant) were joined to a 30 m gridded (raster) version of the SSURGO data, since joining to the polygon version presented multiple processing and display problems due to the very large number of map unit polygons (~36 million) in the continental United States. Producing a raster version of SSURGO soils data (using the ArcGIS Lookup command) enabled tabulations of soil parameters over several polygon features (HUC10, county).

Soil texture data provide data on bulk density, saturated water content, saturated hydraulic conductivity, and van Genuchten soil moisture retention parameters (alpha and beta) correlated on a national scale based on the work of Carsel and Parrish (1988) and Carsel et al. (1988). For measured data, GIS software was used to extract soil parameter grids within the boundaries of modeled agricultural fields. Using the extracted data, EPA calculated mean measured soil parameter values by HUC10. These average values were used as model inputs for all agricultural fields within a given HUC10. For categorical soil parameters, a distribution of values was created, which allowed for probabilistic sampling.

Table E-1 shows the crosswalk used to assign the SSURGO detailed soil textures to basic Soil Conservation Service (SCS) textures, and then to the EPACMTP megatextures. SSURGO soils are classified into 21 texture classes, which map to 12 SCS textures. EPACMTP uses three soil megatextures to represent the variability of hydrologic soil properties, so each SSURGO soil texture was crosswalked to the EPACMTP megatexture with the most similar hydrogeologic properties.

E.1.2 Hydrogeological Environment

Each HUC10 modeled in this analysis was assigned one or more hydrogeologic environment(s) from EPA's Hydrogeologic Database (HGDB) to characterize four subsurface parameters required by EPACMTP: depth to ground water, aquifer thickness, hydraulic gradient, and saturated hydraulic conductivity. The HGDB was developed by the American Petroleum Institute (Newell et al., 1989; 1990) to specify correlated empirical probability distributions of these four parameters for the 12 distinct hydrogeologic environments described in Newell et al. (1990). To assign the HGDB distributions to the HUC10s modeled in this assessment, EPA first developed a national geographic coverage of the 12 hydrogeologic environments, and then used GIS software to overlay the HUC10 locations and dimensions and assign the hydrogeologic environments to each HUC10. Of the 12 environments defined, only 9 intersected the distribution of HUC10s used in this assessment.

EPACMTP uses the HGDB for national and regional analyses. Therefore, it was necessary to assign each HUC10 to one or more hydrogeologic environments corresponding to the HGDB data set. Given the national scale of the risk assessment, only national data sets were used to delineate hydrogeologic environments, defined by an approximate 1:7,500,000 map scale. The following individual map layers were combined using GIS software to develop a single map layer for assigning the 12 hydrogeologic environments across the United States:

- **Shallowest principal aquifers** from *Principal Aquifers of the Conterminous United States, Hawaii, Puerto Rico, and the U.S. Virgin Islands* [USGS map file: aquifrp025]. 1:2,500,000 map scale, was used as the base layer in the assessment and to delineate several of the 12 hydrogeologic environments.
- **Alluvial and glacial aquifers** from *Aquifers of Alluvial and Glacial Origin* [USGS map file: alvaqfp025]. 1:2,500,000 map scale, was used to represent alluvial and glacial aquifers for the 22 states north of the southernmost line of glaciation. Note that the alluvial aquifers in this coverage are identical to those in the Hunt (1979) surficial geology layer below.
- **Surficial geology** of the conterminous United States was taken from:
    - *Surficial Geology of the Conterminous United States* [map file: geol75m]. 1:7,500,000 map scale, provided by Hunt (1979), these data were used to characterize shallow soil lithology and alluvial aquifers.
    - *The Surficial Deposits and Materials in the Eastern and Central United States (East of 102 degrees West Longitude)* [map file: sfgeoep020]. 1:1,000,00 map scale, includes the line of maximum glacial advance and represents surficial materials that accumulated or formed during the past two million years, including residual soils, alluvium, and glacial deposits.
- **Karst aquifers** from *Engineering Aspects of Karst* [map file: karst0p075], 1:7,500,000-map scale, showing karst and pseudokarst (i.e., karst-like terrain produced by processes other than the dissolution of rocks) across the United States.
- **Bedrock geology** from *Generalized Geologic Map of the United States* [map file: geolgyp075], 1:7,500,00 map scale, showing the bedrock geology at or near land surface (i.e., beneath surficial soils, alluvium and glacial deposits).
- **STATSGO soils**, 1:250,000 map scale, from the digital map and attribute data for soils.

As described below, these data layers were used to develop a national hydrogeologic environment layer in GIS for assigning an aquifer type to a point or area of interest. To create the hydrogeologic environment layer, each individual data layer described above was obtained as a GIS shapefile and processed, as needed, to ensure that coordinate systems matched and the layers could be overlain. **Table E-2** describes how the polygons comprising the 9 applicable hydrogeologic environments were developed in the GIS using these layers.

Table E-2. GIS Procedures for Assembling National Coverage of Hydrogeologic Environments.

Hydrogeologic Environment	Description
1) Metamorphic and igneous rock	Select polygons where ROCK_NAME = "igneous and metamorphic-rock aquifers" from the principal aquifer layer [aquifrp025] AND "igneous" or "metamorphic" polygons from the bedrock geology layer [geolgyp075] AND polygons from the Hunt (1979) surficial geology layer [geol75m] derived from or directly overlaying igneous or metamorphic rock.
2) Bedded sedimentary rock	Select polygons where the ROCK_NAME = "sandstone and carbonate-rock aquifers" OR AQ_NAME was "Other rocks" from the principal aquifer layer [aquifrp025] AND "sedimentary" polygons from the bedrock geology layer [geolgyp075].
3) Till over sedimentary rock	Select polygons that had a surficial geology [geol75m] = "mg: deposits of mountain glaciers", "tg: till or ground moraine" "ts: ice-laid deposits, like tg but mostly sand and silt", OR "ts/K,T: thin ice-laid deposits, like ts but thin and discontinuous. . ..", AND overlaying "sedimentary" polygons from the bedrock geology layer [geolgyp075].
4) Sand and gravel	Select polygons where AQ_NAME = "other rocks" OR "high plains aquifer" from the principal aquifer layer [aquifrp025] AND sand and gravel related names in SURFICIALG from the Hunt (1979) surficial geology layer [geol75m] (SURFICIALG contained many different types of sand and gravel deposits).
5) Alluvial valleys, basins and fans	Select polygons where AQ_NAME = "Unconsolidated sand and gravel aquifers" from the principal aquifer layer [aquifrp025] name OR surficial geology type (SURFICIALG) was either "fg: fan gravels" or "fs: fan sands" in the Hunt (1979) surficial geology layer [geol75m].
6) River valleys and floodplains with overbank deposits	Select polygons where SURFICIALG = "al: floodplain and alluvium gravel terraces" in Hunt (1979) surficial geology layer [geol75m] AND STATSGO soils with < 50% sand AND a low permeability (< 0.0147 inches per hour).
7) River valleys and floodplains without overbank deposits	Select polygons where SURFICIALG = "al: floodplain and alluvium gravel terraces" in Hunt (1979) surficial geology layer [geol75m] AND STATSGO soils that do NOT have < 50% sand AND a low permeability (< 0.0147 inches per hour).
8) Outwash	Select polygons where ORIGIN_AGE = "glaciofluvial (outwash) deposits" in the Surficial Deposits and Materials layer [sfgeomean020] AND where SURFICIALG = anything but "ts: ice-laid deposits, like tg but mostly sand and silt" in the Hunt (1979) surficial geology layer [geol75m].
9) Till and till over outwash	Select polygons from Hunt (1979) surficial geology layer [geol75,] that were not already classified hydrogeologic environment 8 AND where SURFICIALG = "w: gravel, sand and clay deposited by glacial streams adjacent to or downstream from temporary ice fronts" OR "ts: ice-laid deposits, like tg but mostly sand and silt" OR "tg: till, or ground moraine".

One or more of the nine hydrogeological environments were assigned to each HUC10 based on overlap of the environments and the HUC10 boundary. When a HUC10 spanned more than one environment, the hydrogeological environment was varied probabilistically based on relative percentage when constructing the database of field properties to simulate for that HUC10. Once hydrogeologic environments were assigned, a preprocessing run of EPACMTP was conducted to construct a set of randomly generated but correlated hydrogeologic parameters (depth to ground water, saturated hydraulic conductivity, aquifer thickness, hydraulic gradient) for each occurrence of the hydrogeologic environments in the source data files. Missing values in the HGDB data set were filled using correlations during EPACMTP execution, as described in U.S. EPA (1997).

E.1.3 Precipitation Data

Forty-one meteorological stations were chosen to represent the nine general climate regions of the continental United States. **Figure E-2** shows these stations and their boundaries. The approach used to define the forty-one regions included the following three main steps:

1. Identify contiguous areas that have similar environments, as defined by Bailey regions (Bailey et al., 1994). Bailey's ecoregions and subregions are used to associate coverage areas with meteorological stations. This hierarchical classification scheme is based primarily on rainfall regimes; subregions are delineated by elevation and other factor relevant to ecology.
2. Select one meteorological station to represent each contiguous area. Station locations were selected based on considerations of the following factors:
    - Major National Weather Service (NWS) stations were selected because these stations are expected to have high-quality equipment that is kept in good repair and is suitably sited.
    - Number of years of surface-level meteorological data available (minimum of five years). More years of data provide a more realistic long-term estimate.
    - Aimed for locations that are central within each region. All other factors being equal, a central location is expected to be most representative of the larger, contiguous region because it has the smallest average distance to all points within that region.
3. Identify the boundaries of the area to be represented by each meteorological station. Thiessen polygons, which are created by a geographic information systems (GIS) procedure that assigns every point on a map to the closest station, were used as the first step in drawing the boundaries. Meteorological boundaries were adjusted to fall along the Bailey boundaries.

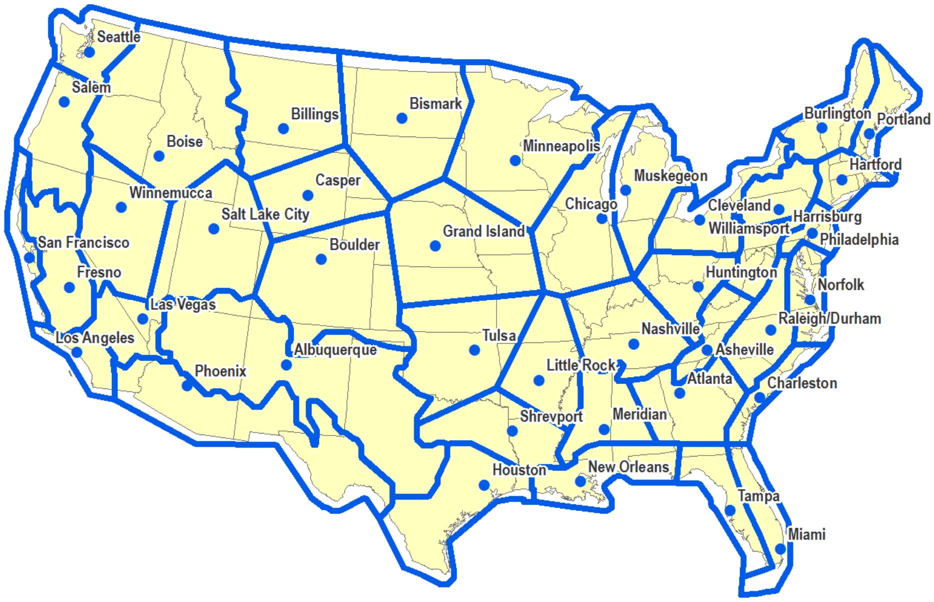

Figure E-1. Meteorological stations and regions

All available daily precipitation data associated with a HUC10 and soil characteristics for the farmland within the region were used by the Land Application Unit (LAU) model. This model retuned annual average runoff and infiltration rates over the specified timeframe. Long-term infiltration and runoff rates were used to calculate concentrations in the leachate flowing to ground and surface water. Estimates of runoff and infiltration rates always began with the first year of available precipitation data and proceeded chronologically. If the model duration exceeded the number of years of precipitation data, the available meteorological data was repeated from the first year as many times as needed.

E.1.4 Surface Water Location and Navigation

This section describes the data used to define the location of each individual surface water body and the relationship between these water bodies (i.e. flow rate and direction). This work relied on data from the Stream-Catchment database (StreamCat; Hill et al., 2016), the enhanced National Hydrography Dataset (NHDPlus; McKay et al., 2017), and the USGS Watershed Boundary Dataset (USGS, 2013). Using these data sources, EPA accumulated a number of catchment-level data attributes, including the navigation relationships for each NHDPlus catchment at the HUC12 levels and above. Assembly and management of such large amounts of data required use of automated routines performed with GIS software. The following text described how these processes were implemented, how the quality control (QC) review was conducted to ensure that the data were assembled properly, and how identified issues were addressed.

Surface water pathways were evaluated at multiple scales: catchment, HUC12, and HUC10. Catchments are typically smaller than HUC12s, which are always smaller than HUC10s. HUC10s can easily be derived from HUC12s due to the nested structure of these data. **Figure E-3** presents the conceptual model of flow from the initial catchments to the outfall of a HUC10.

Figure E-2: Conceptual Model for a HUC10 and Associated Outfalls

Headwaters: Headwaters were defined using NHDPlus catchments associated with Strahler stream orders 1 and 2. Although headwater streams have been defined elsewhere as order 3 and below, EPA limited this evaluation to those that fall entirely within the boundaries of a single HUC12. Exposures were calculated at the outfall from a headwater to any stream greater than order 2. All of the catchments upstream of that outfall were merged to obtain total drainage area and other stream properties. In total, modeling was conducted on this evaluation included a total of 178,506 separate headwater streams.

In some cases the stream order classification was anomalous. For example, in rare cases, headwater streams flowed long distances before terminating or intersecting a larger stream. This results in a catchment area that spans multiple HUC12s. Although these are real streams, a catchment area that extends beyond the bounds of a single HUC12 conflicted with the automated process used to aggregate headwater data. To address this conflict, EPA removed catchments with a drainage area greater than half of the area of the HUC12 of origin. To identify these large catchments, EPA used the "totdasqkm" field (i.e., total drainage area) in the NHDPlus table "plusflowlinevaa" (value-added attributes; McKay et al., 2017). EPA mapped each catchment to a HUC12 based on the location of its centroid and compared the catchment's total drainage area with that of the corresponding HUC12. This was encountered most frequently in particularly dry or flat areas, such as southwestern deserts and the Everglades, and were often areas outside the scope this evaluation.

Headwaters were reviewed for quality control (QC) through visual spot checks conducted across the country (10 checks performed in randomly selected states over an area approximately the size of a HUC4). One check confirmed that the headwater catchments were associated with only Order 1 and 2 stream segments. In instances that an Order 1 or 2 stream was omitted, the catchment area was compared with the HUC12 boundary to check for the issues described above. Another check ensured that drainage areas were calculated correctly by manually summing the area of each catchment along the headwater stream and comparing it to the total area calculated by the automated routine. During the QC process, it was found that the routine did not combine all the catchments along some Order 1 streams. EPA could not determine the frequency that this occurred because it would have required visual inspection of every stream. However, visual inspection conducted across multiple states as part of this QC effort indicates that it is not common (i.e., <1% of headwaters). Each of the defined headwaters was modeled as a separate headwater.

Mainstem Navigation: Mainstem streams refers to the primary route of flow through a drainage system that contains multiple stream segments. Navigation refers to the tracking of surface water flows through each individual stream segments, beginning with headwater streams and continuing downstream until reaching either a coast or a stream of Strahler order 6 or higher, which was used to denote large rivers in this evaluation. This threshold for large rivers resulted from an analysis of watershed contributions in terms of area and characteristics. Streams of order 6 began to exhibit trends that did not follow the same pattern as smaller streams, which is believed to result from the larger drainage areas and greater complexity of upstream contributing areas. This divergence was observed most clearly when the Base Flow Index (BFI) was mapped. Thus, because concentrations are likely to be lower in these larger streams as a result from dilution from a larger drainage area, EPA chose to terminate the analysis once a stream of order 6 was reached.

EPA based navigation of the hydrologic network on NHDPlus catchments (McKay et al., 2017) and a crosswalk of data between these catchments and HUC12s provided within EPA's EnviroAtlas data layers (Pickard et al., 2015). Navigation of the network using NHDPlus was completed by identifying "from" and "to" designations for each catchment. All catchments except for the very first or last should have both designations. This network was used to identify catchments located within and at the outlet of each HUC12 within the EnviroAtlas framework. EnviroAtlas navigation of HUC12s identifies three attributes: HUC12, ToHUC12 and OutletCOMID (i.e., NHDPlus catchment at outlet of a HUC12). EPA used the completed navigation for HUC12s to identify the HUC10 outfalls (i.e., when the next downstream HUC12 fell within a different HUC10, that HUC12 was designated as an outlet). Exposures were calculated either at the outfall of each HUC10 or at the outfall of an individual HUC12 if it discharged directly to a higher order stream.

Several issues were identified during the navigation process. Although NHDPlus incorporates information from the USGS Watershed Boundary Dataset, there are discrepancies between the boundaries in the two datasets that can derail automated navigation. These discrepancies were identified through QC checks run on automated navigation by review of generated tabular data, comparison of the cumulative drainage areas for outlet catchments and the associated HUC12, and visual inspection of the HUC12 routing network.

Closed Basins: These basins have an internal sink to which they drain (i.e., losing streams). They do not flow to a larger stream network or out to the coast. These basins may consist of any number of HUC12s (i.e., from one to several dozen). Some closed basins could be identified from the original HUC12 navigation which flagged the downstream HUC12 as "Closed Basin." Others had to be mapped and visually identified. All closed basins were removed from the modeling analysis because the disconnected hydrology introduced a great deal of uncertainty into the evaluation.

Scale Issues: There were a small number of HUC12s that were oddly shaped or that were smaller than the identified outlet catchment. Depending on the location of these HUC12s and the magnitude of the differences, the outlet catchment was either adjusted to fit the HUC12 or the HUC12 was removed from the model (i.e., coastal outfalls).

Catchment Issues: The automated process used to define outlet catchments defined for each HUC12 did not account for instances where the NHDPlus flowline through the catchment was labeled as a connector, a waterbody, a canal, or some artificial pathway (i.e., path through a waterbody to ensure continuous flow lines). Upon finding a null, outlier (e.g., extremely small cumulative drainage area for an outlet catchment), or other confounding value (e.g., negative cumulative streamflow), EPA visually inspected the specific catchment or HUC12 to determine a remedy. In almost all cases the remedy was to skip over the individual, anomalous catchment and define the next downstream catchment as the outlet. Other remedies were specific to the case and may have used the catchment upstream as the outlet or removal of the HUC12 from the analysis for one of the reasons above.

To support the spatial resolution needed for the evaluation, EPA created a navigation text file for HUC10s that mimics the file received for the EnviroAtlas navigation of HUC12s. The HUC10 file contains three fields: HUC10, ToHUC10, and OutletCOMID. The OutletCOMID is the identifier of the NHDPlus catchment at the outlet of the HUC10. This catchment corresponds to an outlet listed in the HUC12 file; therefore, an outlet HUC12 was identified by joining the two files.

HUC10s are the primary spatial resolution at which model results are aggregated for the evaluation, although individual model runs are conducted at the HUC12 scale. The routing through the hydrologic network was completed to determine cumulative impacts as a post-processing step. In total, modeling was conducted for a total of 7,999 modeled HUC10s (comprised of 32,998 HUC12s). As noted previously, the HUC12 network constructed for this modeling effort had some HUC12s that were not modeled, resulting in the fragmentation of some HUC10s. Because of this order of processing, all routing through the network can be calculated, including routing within only the fragmented pieces and the entire connected network. For instance, the drainage area captured by the outlet catchment did not always capture the entire HUC12 because of the confluence of multiple streams at the outlet. Instead, that flow is reflected in the downgradient HUC12. In addition, overlay of the Economic Feasibility Zone (EFZ) layer created disconnects and gaps within the HUC12 network. The following list summarizes the discrepancies and disconnects identified and describes the approaches used to address the issues:

- For 4,400 HUC10s (55%), all HUC12 were included and a single exposure point was modeled at the outfall of the HUC10. No additional steps were necessary to address these areas.
- For 3,599 HUC10s (45%), one or more HUC12s within the HUC10 did not contribute constituent mass to the HUC10 outfall. These HUC10s had one or more exposure point modeled at either the outfall of the HUC10 or at the outfall of individual HUC12 within the HUC10:
    - 1,635 HUC10s (20%) are intersected by stream of order 6 or higher, resulting in a series of tributaries within the HUC 10 that feed into the large-order stream. EPA modeled exposures at the outfall of each individual HUC12 that discharged into a large-order stream. HUC12s that fell along the flow path of the large-order stream were not modeled.
    - 1,964 HUC10s (25%) have one or more HUC12s that do not contribute to the HUC10 outfall due to areas without agricultural land use, areas outside the EFZ, or other similar causes. EPA modeled flow through these areas the same as every other HUC12. However, it was assumed that these areas contributed zero constituent mass to the downstream mainstem flow.

Watershed Attributes: To complete the evaluation, a number of attributes were needed for each of the assessed spatial units: HUC12s and headwaters. EPA compiled the attribute information using various base data and Value Added Attributes available within and supplemental to NHDPlus. (rather than summing the incremental flows of catchments within the HUC12)

Table E-3. Watershed attributes compiled from NHDPlus

Attribute	Source	Description
Cumulative Baseflow Index (BFI)	EPA's StreamCat Database: BFIs for the entire upstream watershed (BFIWs)	Source: EPA's Stream-Catchment database (StreamCat; Hill et al., 2016) provides geospatial attributes indexed to the NHDPlus version 2 dataset. The BFI is the ratio of base flow to total flow, expressed as a percentage. The BFIWs attribute summarizes the ratio for the entire upstream watershed. Therefore, for each headwater and HUC12 the BFIWs corresponding to the outlet catchment was chosen as the parameter value.
Stream Length	Calculated based on NHDPlus flowline length	Source: NHDPlus dataset field LengthKm in Flowline file (McKay et al., 2017) **Headwaters:** Sum the segment lengths for the individual stream segments (i.e., stream orders 1 and 2) **Headwater HUC12s:** Navigate upstream from the outlet and determine the longest path. Use the maximum length found as the stream length. **All other HUC12s (with an upstream HUC12):** Determine longest path navigated through the HUC12 from inflow to outlet by navigating the flow path from the outlet of target HUC12 upstream to the outlet of upstream HUC12. Then sum the flowline lengths to determine the stream length.
Cumulative and Incremental Streamflow	NHDPlus VAA: Q0001C, and QIncr0001C	Source: NHDPlus dataset fields Q0001C (cumulative) and QIncr0001C (incremental) in file EROM_MA0001 (McKay et al., 2017) All Flow estimates are in cubic feet per second (cfs) and represent the flow at the bottom (downstream end) of the NHDFlowline feature. All Velocity computations are in feet per second (fps) using the Jobson Method (USGS, 1996) and represent the velocity at the bottom of the NHDFlowline feature. For incremental flows, the incremental flow from each catchment within the assessment unit are summed to provide a total incremental flow for the unit (e.g., if there are 5 catchments within a HUC12 then the 5 incremental flow values are summed). For cumulative flow, the value corresponding to the outlet catchment of each assessment unit was selected.
Streamflow Velocity	V0001C	
Percent Cropland	Calculated based on field area	Source: Compiled for this evaluation (See **Appendix C: Use Characterization**) Sum of all field area contained within the corresponding headwater or HUC12 area.

EPA compiled these data based on the identified outlet catchments using an automated process. As the data were compiled into a tabular format for each assessment unit, EPA identified places where the available data were missing or anomalous. A summary of the issues by assessment unit type are described below.

- There were eight modeled headwaters (0.02%) missing BFIWs values. All eight were located in tidal areas and were removed from the analysis.
- There were 73 modeled HUC12s (0.2%) missing BFIWs values. EPA removed four of these HUC12s (0.01%) from the modeling analysis because they were identified as either tidal or closed basins. For 26 of the modeled HUC12s (0.08%), EPA used the BFIWs value reported for the next downstream catchment. Finally, for 43 of the modeled HUC12s (0.1%), EPA used an average BFIWs from all the catchments within the HUC12 because both the outlet and the

next downstream catchment were missing values. The variability of BFIWs within a single HUC12 was generally small and so an average was determined to introduce minimal additional uncertainty.

- There were 43 modeled headwaters (<0.01%) where the incremental flow values reported were less than or equal to zero. EPA removed 15 of these headwaters from the modeling analysis after visual inspection because they were located in the middle of a waterbody, such as a drainage canal or a "connector" (an artificial designation used to ensure that all streams connect), which do not reflect typical flowing streams. For the remaining 28 headwaters, EPA used the value reported for the next downstream catchment.
- There were 1,318 modeled HUC12s (4%) with incremental flow values less than zero. For 374 of these HUC12s (1%), EPA calculated the incremental flow based on the difference between the cumulative flows reported for that HUC 12 and the one immediately upgradient. For the remaining 944 HUC12s (3%), all efforts to calculate an alternate incremental flow with NHDPlus data resulted in negative values. Many of these HUC12s had lakes or other larger water bodies located at the outfall, which likely caused issues in the reported flows. Therefore, EPA assigned a fixed value of 10 cfs based on a typical values reported for HUC12s with data and best professional judgment.

Lakes: In addition to rivers and streams, an effort was made to evaluate lakes, reservoirs and other lentic water bodies, referred to collectively as "lakes" in this discussion for simplicity. To locate these lakes, EPA selected any water body classified as either a lake or reservoir in NHDPlus. EPA identified two broad classes of lakes:

- Flowthrough lakes are those that fall along the navigated stream network and contribute flow to downgradient streams. These lakes are frequently located entirely within a single HUC12. Given the larger cumulative drainage area upgradient of these lakes, the majority of the water that flows through these water bodies originates from upgradient streams. As a result, it is expected that the long-term concentrations from mixing within these lakes will be similar to adjacent streams. Thus, for the purposes of this evaluation, EPA treated these water bodies the same as streams.
- Terminal lakes are those that do not fall within the stream network. Terminal lakes were designated as such if they touched an NHD flowline with a terminal flag (i.e., NHD attribute) set to true. These water bodies were often located entirely within a single catchment. Because these water bodies receive flow primarily from an isolated drainage area, it is more likely that long-term concentrations could differ from nearby streams. However, there was not enough data (e.g., depth, volume, percent of catchment area drained) to model these water bodies without a number of assumptions that would introduce a great deal of uncertainty into model results. Thus, EPA did not model these water bodies in the evaluation.

E.1.5 Surface Water Characteristics

Data on regional surface water characteristics were collected from the legacy STORET database. STORET is the largest single source of water quality data in the United States, containing over 275 million analyses performed on more than 45 million samples collected from 800,000 stations across the country between 1960 and 1998. However, the STORET website states:

> *"The EPA does not change or filter incoming data. This means that when pulling data out of the Warehouse, users must be aware that they are responsible for screening the data for their use."*

EPA notes that there is a high degree of variability in these data due to differences in quality assurance/quality control testing, bias towards samples collected at site locations known to have contamination problems, and bias towards samples collected during critical periods (e.g., summer low flows). To account for these factors, EPA used the STORET data as discussed below.

Temperature: Median surface water temperatures were collected for each hydrologic region and assigned to each water body within that region. Median values were selected to capture reflect annualized values. **Table E-4** provides the temperatures used for each hydrologic region.

Table E-4. Regional Surface Water Temperature

Hydrologic Region	Median Surface Water Temperature (°C)
1	14
2	16
3	21
4	14
5	17
6	18
7	15
8	20
9	10
10	13
11	17
12	21
13	16
14	9
15	17
16	9
17	11*

* Legacy STORET data not available for region 17 at this time. Assigned median temperature of 11 based on professional judgment to represent cooler surface water temperature in the mountainous Pacific northwest region.

Total Suspended Solids (TSS): Minimum, maximum and weighted geometric means of these collected values were used to define log triangular distributions regionally for streams. The triangular distribution was selected because it is typically used to describe a population for which there is only limited sample data, but there is sufficient knowledge to determine that the distribution is modal, rather than uniform, as was the case here. Geometric means weighted by the annual number of measurements were used because the actual distribution around the median is unknown. Once developed, these distributions were sampled during the preparation of the input data files. **Table E-5** provides the TSS values used to define the distribution for each region, along with the number of the modeled facilities assigned to that region.

Table E-5. Surface Water Total Suspended Solids (TSS) Distributions

Hydrologic Region	Number of Annual Median Values	Annual Median TSS (log-triangular distribution)		
		Minimum	Weighted Geometric Mean	Maximum
1	33	3.2	8	40
2	38	10	32	316
3	36	6.3	25	79
4	37	6.3	25	794
5	38	4	25	100
6	28	5	16	316
7	37	32	63	1,585
8	38	50	158	316
9	35	13	32	3,162
10	38	10	126	398
11	38	25	200	794
12	35	40	79	1,995
13	37	32	200	79,433
14	38	16	158	5,012
15	37	20	200	19,953
16	33	4	16	2,512
17	37	2	6	316

Suspended Sediment Partitioning: The model partitions constituent mass between surface water, suspended solids and benthic sediment with linear partition coefficients. This approach assumes that equilibrium is maintained among these dissolved constituents within the water column and constituents in suspended solids and bed sediment. **Table E-6** provides distributions for the partitioning coefficients used by the surface water model. These distributions were derived from published empirical data presented in U.S. EPA (2005a), Allison et al. (2003) for manganese, and ORNL (1984) for iron. These data were sampled during the preparation of the input files in the Monte Carlo process.

Table E-6. Log Distribution of Sediment/Water and Suspended Solids/Water Partition Coefficients

Constituent	Sediment/Water				Suspended Solids/Water			
	Min	Mean	Max	Standard Deviation	Min	Mean	Max	Standard Deviation
Antimony	0.6	3.6	4.8	1.8	--	--	--	--
Arsenic	1.6	2.4	4.3	0.7	2.0	3.9	6.0	0.5
Cadmium	0.5	3.3	7.3	1.8	2.8	4.9	6.3	0.6
Chromium III	1.9	4.9	5.9	1.5	3.9	5.1	6.0	0.4
Chromium VI	0	1.7	4.4	1.4	3.6	4.2	5.1	0.5
Iron	--	--	--	--	N/A	1.4	N/A	N/A
Lead	2.0	4.6	7.0	1.9	3.4	5.7	6.5	0.4
Manganese	2.4	3.2	4.7	0.7	4.5	4.7	5.3	0.2
Mercury (divalent)	3.8	4.9	6.0	0.6	4.2	5.3	6.9	0.4
Mercury (methyl)	2.8	3.9	5.0	0.5	4.2	4.9	6.2	0.7
Nickel	0.3	3.9	4.0	1.8	--	--	--	--
Selenium IV	1.0	3.6	4.0	1.2	3.8	4.4	4.8	0.4
Selenium VI	-1.4	0.6	3.0	1.2	3.1	3.8	4.6	1.0
Thallium	-0.5	1.3	3.5	1.1	3.0	4.1	4.5	1.0
Zinc	1.5	4.1	6.2	1.6	--	--	--	--

-- Constituent not evaluated for this pathway.
N/A = data not available.

E.2 Water Mass Balance Model

The hydrologic module of the land application unit (LAU) model was used to estimate long-term water balance in the field. First-order partitioning was assumed to distribute soluble constituent mass between the overland and subsurface transport pathways. **Figure E-4** depicts the conceptual model for water flow used in this evaluation.

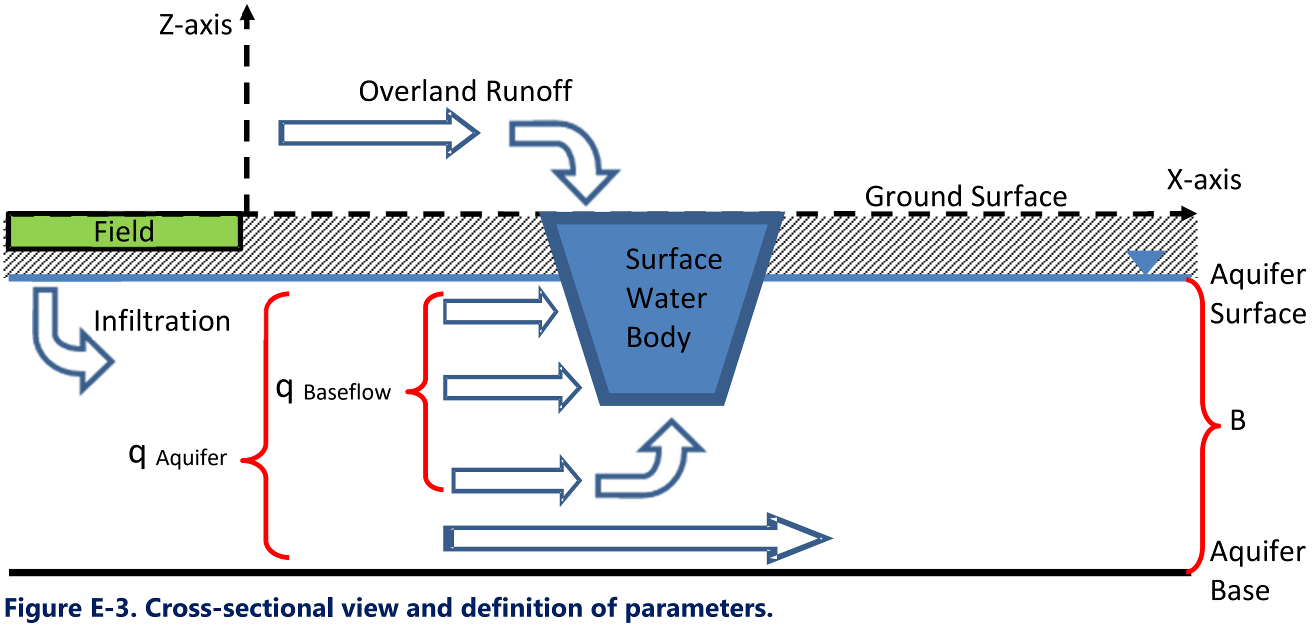

Figure E-3. Cross-sectional view and definition of parameters.

The LAU model generates estimates of long-term average runoff and infiltration. Infiltration contributes to regional aquifer flow (q_{Aquifer}) within the saturated thickness of the aquifer (B), modeled by EPACMTP; baseflow from aquifer to the surface water body ($q_{\text{ef w}}$) is derived from NHDPlus. Runoff contributes to surface water body flow. The following sections describe the hydrology model used to simulate the movement of water in and around farm fields and introduce how data from NHDPlus are used in various water balance calculations.

The LAU hydrologic model (U.S. EPA, 2003c) was used to simulate watershed runoff and ground water recharge ("infiltration"). The hydrology module is based on a daily soil moisture balance performed within the root zone of the soil column. At the end of a given day, the soil moisture in the root zone is the net moisture balance from the previous day with addition of water from precipitation and residual moisture in FGD gypsum and subtraction of water losses through runoff, infiltration and evapotranspiration.

Data on local climate and hydrogeologic environment associated with each HUC10 were used to determine long term rates of infiltration and runoff for agricultural fields within the HUC10. Precipitation is undifferentiated between rainfall and frozen precipitation; frozen precipitation is treated as rainfall on an annualized basis. As described above, available daily precipitation data from climate stations associated with a HUC10 were coordinated with average soil characteristics for the farmland within the same region and presented to the LAU module.

Potential evapotranspiration (PET) is the demand for soil moisture from evaporation and plant transpiration. When soil moisture is abundant, actual evapotranspiration (ET) equals PET. When soil moisture is limiting, ET will be less than PET. The extent to which it is less under limiting conditions has been expressed as a function of PET, available soil water, and available soil water capacity. Water that is not lost to evapotranspiration is available to runoff or infiltrate.

Runoff is based on the Soil Conservation Service curve number procedure (USDA, 1986) and is a function of current and antecedent precipitation, as well as land use. Land use catalogued by cover type (e.g., woods, meadow, impervious surfaces), treatment or practice (e.g., contoured, terraced), hydrologic condition, and hydrologic soil group.

Soil moisture in excess of the soil's field capacity, if not lost through evapotranspiration, is available for gravity drainage from the root zone as infiltration to subroot zones (Dunne and Leopold, 1978). The rate of infiltration is limited by the saturated hydraulic conductivity of the unsaturated soil (Ksat). If infiltration exceeds the Ksat, a feedback loop is triggered that increases the previously calculated runoff volume by the amount of excess soil moisture (i.e., above field capacity and Ksat). This adjustment is made to preserve water balance and assumes that the runoff curve number method, which is not highly sensitive to soil moisture, has admitted more water into the soil column than can be accommodated. After the runoff is increased to account for this excess, the ET, infiltration, and soil moisture are updated to reflect this modification and preserve the water balance. The resulting long-term average overland runoff and ground water infiltration rates determine the rate that mass is depleted from soils receiving FGD gypsum.

E.3 Source Term Model

The distribution of constituent mass present in the applied FGD gypsum and released to overland runoff and infiltration is calculated prior to ground water or surface water model runs. Dissolved concentrations are dependent on the mass balance of water calculated for each release pathway, as well as both the constituent concentrations present in and released from FGD gypsum; leaching behavior (i.e., availability or solubility-limited); soluble fraction of constituent mass; gypsum application rate, frequency and duration that are sampled from available distributions.

Measured leachate concentrations were adjusted based on leaching behavior prior to use in the model. For constituents with solubility-controlled leaching behavior, measured concentrations were used without further adjustment. For constituents with availability-limited behavior, the leachate concentration was increased to ensure depletion of the soluble content. As discussed in **Section 5 (Screening Analysis)**, this was done to account for potential for the probabilistic analysis to combine high-end values for bulk concentration, leachate concentration and soluble fraction that could result in scenario where available content does not deplete within the year. While this will result in higher leachate concentrations than will actually occur in the field, it will not result in a dramatic overestimation of releases. Any concentrations higher than that needed to deplete available content will result in faster depletion and the exact same annualized concentration. It is possible that this adjustment could push some constituents above solubility limits; however, past studies have found that similar adjustments provided a reasonable estimate of field leaching (U.S. EPA, 2014a).

Applied leachate and runoff concentrations for annually applied gypsum uses are equal to the calculated leachate concentration if available soluble mass is not depleted from soils between applications. In these instances, leaching is assumed to persist for a time required to deplete all applied soluble constituent mass. However, if the applied soluble mass depletes prior to the next application, an effective dissolved concentration is determined for both pathways such that soluble mass of one application depletes in exactly one year. The effective dissolved concentration is applied for as many years as FGD gypsum is applied to the field. This same approach is used to determine applied constituent concentrations in runoff water for ten-year application frequency uses: an effective concentration is calculated if soluble mass depletes prior to the next gypsum application; otherwise, the calculated concentration was used. The same rules for specifying how long leaching occurs. The result in all these scenarios is a constant dissolved constituent concentration release for as long as it takes to deplete the soluble mass.

Ten-year application frequencies were handled in a slightly different manner for the ground water pathway. The calculated leachate concentration was always applied regardless of how much time was required to deplete soluble constituent mass in gypsum application. If the time required to deplete the available soluble constituent mass in a single gypsum application extends beyond the next scheduled application, mass is released at a constant rate for as long as it takes to deplete all applied mass for all gypsum applications. If the soluble constituent mass from a single gypsum application depletes before the next gypsum application, the timing of soluble constituent mass releases will respect the depletion time resulting in a repeating square wave profile where each wave persists for a time equal to the depletion time for a single application, and the time between square waves equals ten years minus the depletion time.

E.4 Ground Water Model

Ground water modeling was conducted with EPACMTP (U.S. EPA, 2003a). This model consists of two steady state flow modules that address subsurface flow through unsaturated and saturated zones. Data requirements for EPACMTP ground water flow calculations are satisfied with HUC10-based distributions of soil megatextures corresponding to SSURGO classifications within the HUC, hydrogeologic environment assignments to a HUC10, and infiltration rates from the LAU module mentioned above. Specific soil and aquifer parameters are drawn from megatexture and hydrogeologic environment assignments from a preliminary simulation of EPACMTP using established distributions as described in the *Technical* and *Parameter/Data Background* documents (U.S. EPA, 2003a,b). A database of aquifer parameters is developed from the preliminary simulation and reused for all subsequent simulations of FGD gypsum Uses and constituents.

EPACMTP consists of two coupled modules that address subsurface transport through unsaturated and saturated zones. These modules treat soils as uniform, porous media and do not account for preferential pathways or facilitated transport. EPA assumed that farm drinking water wells are located upstream from surface water bodies, and so did not consider interception of ground water by surface water bodies prior to reaching the well. This is considered a reasonable assumption given the scattered spatial distribution of farm fields across the landscape.

In the unsaturated zone, the flow of water is driven primarily by gravity. Therefore, flow is modeled entirely in the vertical direction (i.e., no lateral flow). This assumption can be made because the scale of lateral migration due to dispersion will be orders of magnitude less than the scale of vertical migration through areas receiving application of FGD gypsum (U.S. EPA, 2003a). The solution to the governing equation for unsaturated zone flow yields estimates of vertical Darcy velocity and average water content used to simulate contaminant transport. Darcy velocity influences constituent advection and water content is used to determine equilibrium partitioning of constituent mass between dissolved and sorbed phases.

In the saturated zone, flow is controlled primarily by the hydraulic conductivity of the aquifer and the regional hydraulic gradient. Ground water flow velocities are the principal output of the solution to the governing equation for steady state ground water flow. Contaminant transport within the unsaturated and saturated zones requires flow velocities to advect and disperse dissolved constituent mass in the porous media. EPACMTP assumes that movement of constituent mass is driven primarily by the ground water advection. However, flow may be altered both by mounding underneath the field from high volumes of leachate, which encourages spreading of the constituent plume in all directions, and by uncontaminated recharge from precipitation that falls around the field, which increases mixing in the vertical direction. In addition to advection, EPACMTP also accounts for the mixing of ground water due to dispersion, which occurs to some degree in all directions.

During transport through both the unsaturated and saturated zones, constituents may sorb to the surrounding soils. This process is represented by soil-water partitioning coefficients (K_d values), which is the ratio of constituent mass sorbed to the soil and dissolved in solution at equilibrium. For most inorganics, K_d values are strongly dependent on the concentration present in leachate and generally decrease with higher concentrations. The K_d values used in this analysis were drawn from constituent-specific distributions of K_d values versus leachate concentrations, also known as sorption isotherms. EPA selected nonlinear isotherms as the most representative of changes in sorption as leachate concentrations decrease during transport through soil and ground water. This approach is believed to be appropriate and reasonable because the vast majority of leachate concentrations are low enough to fall in the linear region of the nonlinear isotherms. Nonlinear isotherms were generated by Metal Speciation Equilibrium Model for Surface And Ground Water (MINTEQA2) (U.S. EPA, 2001) for use with EPACMTP. The development of these isotherms is described in detail in Appendix G of the EPACMTP *Technical Background Document* (U.S. EPA, 2003a).

Leachate concentrations applied to the ground water pathway described above were used to select K_d values for the unsaturated zone module, while the soil pore water concentrations at the boundary of the unsaturated and saturated zones were used to select K_d values for the saturated zone module. Constituents with low K_d values will have low retardation factors and may move at nearly the same velocity as the ground water. Constituents with high K_d values will have high retardation factors and may move much more slowly than ground water. The subsurface migration of some constituents may be very slow, and it may take a substantial amount of time for the constituent plume to reach the downgradient receptors. As a result, the maximum concentration may not occur until thousands of years after FGD gypsum has ceased. To prevent prohibitive model run times, while not missing significant risks to potential receptors, EPA ran the model until either the observed ground water concentration of a constituent at the receptor point peaked and then fell below a model-specified minimum concentration (10^{-16} mg/L), or the model had been run for a total duration of 10,000 years.

E.5 Surface Water Model

Ground water flow velocity at the ground-surface water interface is used to estimate total aquifer flow in the vicinity of the surface water body. To predict the mass flux of a constituent from an aquifer into a stream, the distribution of constituent concentrations and volumetric ground water fluxes along the upgradient edge of the stream must be known. The volumetric ground water flux depends on the difference between the stream stage and the hydraulic head in the aquifer. If the hydraulic head in the aquifer is higher than the stream stage, the ground water from the aquifer will enter the stream as baseflow. When this occurs, the stream is said to be a gaining stream. Baseflow into each stream was estimated using the BFI, which captures mean baseflow activity for a stream, thereby supporting the assumption that all streams included in this analysis are gaining streams. **Figure E-5** depicts a generalized scenario for ground water interacting with a surface water body.

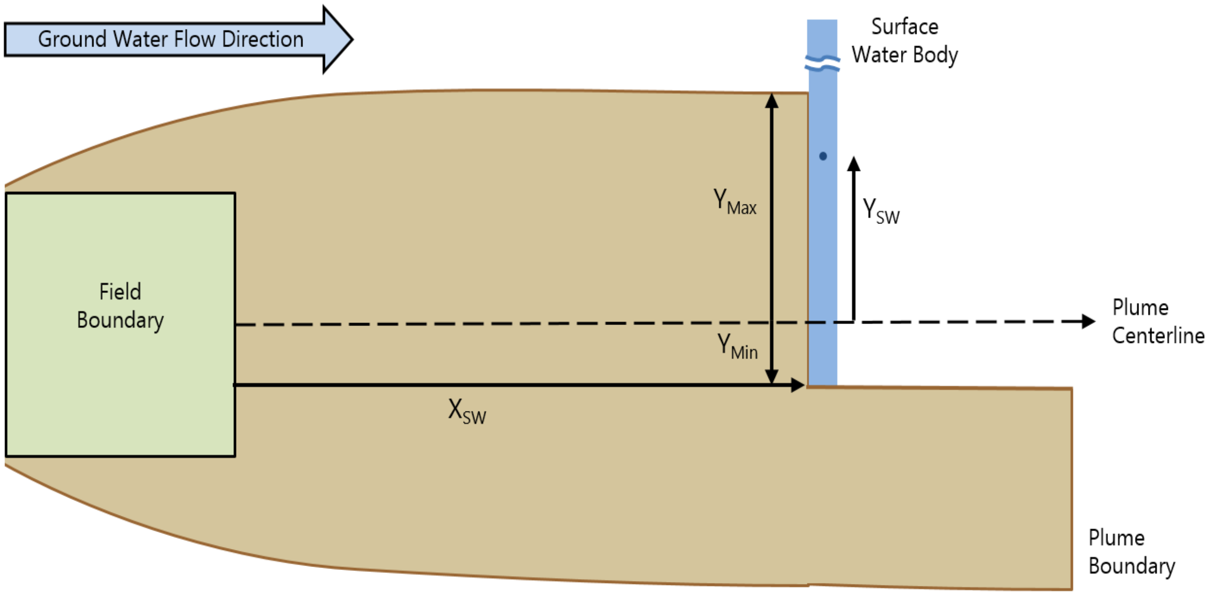

Figure E-4. Plan view of water body interception of ground water plume and definition of parameters

As shown in this figure, the water body does not always completely intercept the ground water plume. This can occur if there is a bend in a stream or if only a portion of smaller headwater streams intersect the plume. The centroid of the water body is located at the point (X_{SW}, Y_{SW}) where X_{SW} is the distance to the surface water body, and Y_{SW} is assumed to be the same distance from the plume centerline assigned to the receptor well. The total mass flux is determined using the baseflow rate and the output constituent concentration from EPACMTP according to the equation below (variable names are only defined the first time appear).

The ground water volumetric flux per unit area of stream bed is governed by the difference in ground water hydraulic head, hydraulic conductivity of streambed, and streambed thickness. Baseflow was estimated based on the BFI and flow of each water body segment. Stream flow contributed by runoff to a stream segment was calculated from the simulated runoff rate used to estimate runoff concentrations by dividing the simulated rate by the inverted BFI for that HUC. This flow rate was calculated as a first step to relate mass loading from runoff and ground water discharge, which is estimated based on flow rates from NHDplus.

(E-1) $$Q_{F}^{LAU} = \frac{q_{RO} \cdot A_{Catch} \cdot BFI}{100 - BFI} = \frac{Q_{RO}^{LAU} \cdot BFI}{100 - BFI}$$

Where:

A_{Catch} – Area of the HUC12 contributing flow to main stem [m^2]
BFI – NHDPlus mean annual base-flow index [%]
Q_{F}^{LAU} – Modeled ground water discharge (baseflow) to surface water [m^3/yr]
Q_{RO}^{LAU} – Modeled overland runoff to surface water [m^3/yr]
q_{RO} – Modeled specific runoff rate (depth) from total drainage area [m/yr]

Mass flux from ground water can be estimated by integrating the product of baseflow flux per unit area and constituent concentration over the total baseflow area. The total baseflow area is determined by the baseflow depth, D_{BF}, and the length of the water body intercepting the ground water plume.

(E-2) $$\dot{m}_{GW}^{LAU} = \left(\frac{q_{Aquifer}}{B}\right)\left(\frac{1\text{g}}{1{,}000\text{ mg}}\right)\int_{y=Y_{min}}^{y=Y_{max}}\int_{z=0}^{z=D_{BF}} C(X_{SW}, y, z, t)\, dy\, dz$$

Where:

$\dot{m}_{GW}^{HUC}$ – Incremental mass flux from ground water to HUC12 water body (g/yr)

B – Saturated thickness of the aquifer (m)

D_{BF} – Depth of baseflow (m)

$q_{Aquifer}$ – Volumetric flow rate per unit width of aquifer (m^2/yr)

Y_{Max} – Leftmost intersection of plume and stream bed with respect to ground water flow direction (m)

Y_{Min} – Rightmost intersection of plume and stream bed with respect to ground water flow direction (m)

C – Chemical concentration at surface water boundary (mg/L)

x – Horizontal distance of stream from the downgradient edge of the field (m)

X_{SW} – Distance of stream from the downgradient edge of the field (m)

y – Horizontal distance from the plume center line along a vertical plane at the upgradient side of the surface water body (m)

Y_{SW} – Distance of stream centroid from the plume centerline (m)

z – Vertical distance from the top of a vertical plane at the upgradient side of the surface water body, positive downward (m)

The ground water mass flux to surface water was treated as a direct load to the water body. The corresponding contributions from runoff were calculated separately as a mass flux to obtain the same units as contributions from ground water. The mass flux of a dissolved constituents in overland runoff to a water body was estimated more directly as the product of the calculated concentration in runoff from the field, the annualized runoff rate and the area of the field receiving FGD gypsum application (relative to the total area of the drainage area).

(E-3) $$\dot{m}_{RO}^{LAU} = C_{RO} \cdot q_{RO} \cdot A_{FGD}$$

Where:

A_{FGD} – Area of the HUC12 receiving FGD application [m^2]

C_{RO} – Concentration in runoff water [mg/L = g/m^3]

$\dot{m}_{RO}^{LAU}$ – Mass flux from runoff water [g/yr]

The mass loading calculated for ground water $\left(\dot{m}_{GW}^{LAU}\right)$ and surface water $\left(\dot{m}_{RO}^{LAU}\right)$ were summed together to obtain a total loading to the water body. To convert mass load back to a concentration, the annual flow through the headwater stream was derived from the calculated runoff volume from the entire drainage area scaled by the BFI for that area.

(E-4) $$Q_{Total}^{LAU} = Q_{RO}^{LAU}\left(\frac{100}{100 - BFI}\right) = Q_{F}^{LAU}\left(\frac{100}{BFI}\right)$$

Where:

$Q_{T\ t}^{LAU}$ – Annual average incremental flow through a stream segment [m^3/yr]

This total mass load was used together with the flow in the water body segment $\left(Q_{T\ t}^{LAU}\right)$ are used to calculate the concentration in the water column $\left(C_{wct\ t}^{HUC}\right)$ and sediment $\left(C_{b}^{HUC}\right)$ based on the steady-state model documented in EPAs *Human Health Risk Assessment Protocol (HHRAP) for Hazardous Waste Combustion Facilities* (U.S. EPA, 2005b). These equations were used to calculate the distribution of constituent mass between dissolved and sorbed phases within the water column. The following are key assumptions of the model and specifics on how it was applied in this evaluation:

- The model assumes steady-state flow and transport conditions. Long-term average annual stream flow and climatic data and assumptions were used.
- The model accounts for constituent loadings into the water body through ground water discharge and waste outfalls, and direct air deposition. The sources relevant to this beneficial use evaluation are discharges from ground water and overland runoff.
- The model estimates the rate of incorporation, or burial, of constituents into bed sediments as a function of the rate at which sediments deposit from the water column onto the surficial sediment layer. The burial rate was set to zero (a protective assumption) because of the lack of national data available to estimate this process.
- The model can incorporate separate decay rate constants for the water column and the benthic sediments to allow for consideration of decay mechanisms that remove constituents from the water body. However, because all inorganics considered are persistent in the environment, degradation was not relevant to this risk assessment.

The following equations used the mass load and stream flow to calculate the concentrations in each headwater stream. The resulting concentrations were used to calculate exposures for ecological receptors living in and around the water body.

(E-7) $$C^{HUC}_{wt\ \ t} = \frac{\partial \dot{m}^{HUC}}{Q^{LAU}_{T\ \ t} \cdot f_{w\ \ ter} \cdot \left(\frac{d_z}{d_z - d_b}\right)}$$

(E-8) $$C^{HUC}_{wct\ \ t} = C^{HUC}_{wt\ \ t} \cdot f_{w\ \ ter} \cdot \left(\frac{d_z}{d_z - d_b}\right)$$

(E-9) $$C^{HUC}_{dw} = C^{HUC}_{wt\ \ t} \cdot f_{w\ \ ter} \cdot f_{di\qquad ve} \cdot \left(\frac{d_z}{d_z - d_b}\right)$$

(E-10) $$C^{HUC}_{b} = C^{HUC}_{wt\ \ t} \cdot f_{benthic} \cdot \frac{d_z}{d_b}$$

Where:

d_b – Depth of upper benthic layer [0.03 m]
d_z – Depth of the water body [m]
C^{HUC}_{b} – Total concentration in HUC12 stream bed sediment [g/m^3 or mg/L]
C^{HUC}_{dw} – Dissolved water body concentration in HUC12 stream [g/m^3 or mg/L]
$C^{HUC}_{wt\ \ t}$ – Total water body concentration in HUC12 stream [g/m^3 or mg/L]
$f_{benthic}$ – Fraction of constituent mass sorbed to benthic sediment [unitless]
$f_{w\ \ ter}$ – Fraction of constituent mass in the total water column [unitless]

The following equations were used to calculate the fraction of constituent mass that partitions between the water (i.e., dissolved), suspended solids and benthic sediment.

(E-11) $$f_d = \frac{1}{\left(1 + (K_{d\ \ w} \cdot TSS)\left(\frac{1\ L}{1{,}000\ ml}\right)\left(\frac{1\ g}{1{,}000\ mg}\right)\right)}$$

(E-12) $$f_{w\ \ ter} = \frac{\left(\frac{d_z - d_b}{d_z}\right)\left(\frac{1}{f_d}\right)}{\left(\frac{d_z - d_b}{d_z}\right)\left(\frac{1}{f_d}\right) + \frac{d_b}{d_z}[bsp + (K_{db}\ \ \cdot bsc)]}$$

(E-13) $$f_{benthic} = \frac{\frac{d_b}{d_z}[bsp + (K_{db}\ \ \cdot bsc)]}{\left(\frac{d_z - d_b}{d_z}\right)\left(\frac{1}{f_d}\right) + \frac{d_b}{d_z}[bsp + (K_{db}\ \ \cdot bsc)]}$$

Where:

bsc – Bed sediment particle concentration [1 g/cm^3 or 1 kg/L]
bsp – Bed sediment porosity [0.6 cm^3/cm^3]
f_d – Fraction of constituent mass in water column that is dissolved [unitless]
k_b – benthic burial rate constant [1/yr]
K_{db} – Sediment-water partition coefficient [mL/g]
$K_{d\ \ w}$ – Suspended sediment-water partition coefficient [mL/g]
TSS – Total suspended solids [mg/L; **Table E-5**]

Of the constituents found in FGD gypsum, only mercury has the potential to volatilize to any appreciable degree. Additional data and equations were necessary to account for this loss pathways. When modeling mercury, the following set of equations were substituted for **Equations E-7** to account for mercury volatilization.

(E-7a) $$C^{LAU}_{wtot} = \frac{\dot{m}^{HUC}}{Q^{LAU}_{Tot} \cdot f_{water} \cdot \frac{d_z}{d_w} + (k_{wt} \cdot A_w \cdot d_z)}$$

(E-7b) $$k_{wt} = f_{water} \cdot f_{dissolve} \cdot k_v + f_{benthic} \cdot k_b$$

(E-7c) $$k_v = \frac{K_v}{d_w}$$

(E-7d) $$K_v = \left[\frac{1}{K_L} + \frac{1}{\left(K_G \cdot \frac{H}{R \cdot T_w}\right)}\right]^{-1} \cdot \theta^{(T_w - 293)}$$

(E-7e) $$K_L = \sqrt{\frac{\left(\frac{1\ \text{m}^2}{1000\ \text{cm}^2}\right) \cdot D_w \cdot u}{d_z} \cdot \left(\frac{3.1536 \times 10^7\ \text{sec}}{\text{yr}}\right)}$$

Where:

θ – Temperature correction factor [unitless; 1.026]
A_w – Surface area of water body [m^2; NHDPlus]
D_w – Diffusivity of mercury in water [1.77×10^{-5} cm^2/sec]
H – Henry's Law constant for mercury [7.1×10^{-10} $atm \cdot m^3/mol$]
K_G – Gas phase transfer coefficient for mercury [36,525 m/yr]
K_L – Liquid phase transfer coefficient [m/yr]
K_V – Overall constituent transfer coefficient from liquid to gas phase [m/yr]
k_v – Water column volatilization rate constant [1/yr]
k_{wt} – Total water body dissipation rate constant [1/yr]
R – Universal gas constant [8.205×10^{-5} $atm \cdot m^3/mol \cdot K$]
T_w – Water body temperature [K; **Table E-4**]
u – Water body current velocity [m/sec; **Table E-3**]

Calculations for the concentrations in mainstem streams used the same set of equations described above for headwater streams. There are two major differences between mainstem and headwater streams. The first is that the mass flux from overland runoff and baseflow are calculated for each entire HUC12. The second is that the total mass flux through each HUC12 outfall also includes contributions from any upstream HUC12. The mass contribution from each HUC12 to next was calculated by multiplying the total water concentration at the outfall (C^{LAU}_{wctot}) by the incremental annual average flow from NHDPlus $\left(Q^{NHD}_{Increment}\right)$ to obtain a mass flux (dissolved and adsorbed) contributed by that HUC12. At each HUC12 outfall, the mass loading from the current and all upstream HUC12 were summed together. When summing constituent mass from upgradient HUC12, the model runs used to characterize each HUC12 were allowed to vary, which resulted in combinations that captured a range of application areas and rates across the landscape. The total mass loading $\left(\dot{m}^{HUC}_{Outfall}\right)$ at each HUC10 outfall was used together with the total NHDPlus flow in that water body segment $\left(Q^{NHD}_{T\ t}\right)$ to calculate the concentration in the water column $\left(C^{HUC}_{wct\ t}\right)$ and sediment $\left(C^{HUC}_{b}\right)$. This process was done at each HUC10 outfall until the stream reached either a coastline or another stream of order 6 or higher. These resulting concentrations in each HUC10 outfall were used to calculate exposures from fish ingestion.

The use of NHDPlus flow rates to accumulate mass between HUC12 conserves concentration in the water column, but not mass. This approach was selected for several of reasons. First, the spatial resolution of weather data used in the LAU model is somewhat limited. Use of NHDPlus flow rates better capture variability in the relative contributions from adjacent HUCs. Next, there are a number of upgradient HUCs that fell outside the economic feasibility zone and so were not modeled. Use of NHDPlus avoided the need to calculate runoff over a much wider area. Finally, in areas where calculated runoff was zero, this would have resulted in zero flow from that segment of the water body. Thus, use of NHDPlus flows captured any baseflow from these areas.

E.6 Soil Model

Soil concentrations are dependent on the frequency and duration of FGD gypsum applications, the constituent concentrations present in and released from gypsum, and the fraction of constituent mass that is soluble. This model assumes that FGD gypsum is initially applied on the soil surface and eventually tilled into the earth. Therefore, long-term soil concentrations are calculated based on mixing within the top 20 cm of the soil column.

Long-term soil concentrations in farm fields receiving FGD gypsum applications are determined over up to a 100-year period assuming first-order losses of the soluble fraction of constituent mass to the subsurface (i.e., leaching) and runoff at a rate equal to the assigned leachate and runoff concentrations determined above. The general calculation for each year is as follows assuming the soil is initially free of constituent mass is:

1. For each year of application, calculate constituent mass added to field. Track soluble and insoluble fractions of applied constituent mass separately.
2. Calculate soluble losses to leaching and runoff for a year.
3. Subtract soluble losses from soluble mass fraction on soil.
4. Sum the insoluble and soluble fractions of the current year to the previous year's total soil concentration

If the total soil concentration at the end of the current year is greater than the previous maximum, update the maximum soil concentration to be equal to the current year.

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