Coal Combustion Residual Beneficial Use Evaluation: Fly Ash Concrete and FGD Gypsum Wallboard

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Executive Summary

Purpose

The goal of this document is to use sound science based on accepted and standard practices to determine whether the United States Environmental Protection Agency ("EPA" or "the Agency") should support the beneficial use of coal fly ash in concrete and flue gas desulfurization (FGD) gypsum in wallboard. Coal fly ash used as a direct substitute for portland cement in concrete (hereafter referred to as "fly ash concrete") and FGD gypsum used as a replacement for mined gypsum in wallboard (hereafter referred to as "FGD gypsum wallboard") are the two largest encapsulated beneficial uses of coal combustion residuals (CCRs) in the United States.

In addition, this document provides an example of how to conduct similar analyses using EPA's *Methodology for Evaluating Encapsulated Beneficial Uses of Coal Combustion Residuals* (US EPA, 2013a). This example also demonstrates an appropriate level of documentation for such analyses.

Conclusions

Based on the analysis set forth in this document, the evaluation concludes that environmental releases of constituents of potential concern (COPCs) from CCR fly ash concrete and FGD gypsum wallboard during use by the consumer are comparable to or lower than those from analogous non-CCR products, or are at or below relevant regulatory and health-based benchmarks for human and ecological receptors.

The beneficial use of CCRs, when conducted in an environmentally sound manner, can contribute significant environmental and economic benefits. Environmental benefits can include reduced greenhouse gas emissions, reduced need for disposing of CCRs in landfills, and reduced use of virgin resources. Economic benefits can include job creation in the beneficial use industry, reduced costs associated with CCR disposal, increased revenue from the sale of CCRs, and savings from using CCRs in place of other more costly materials.

Based on the conclusion of the analysis in this document stated above, and the available environmental and economic benefits, EPA supports the beneficial use of coal fly ash in concrete and FGD gypsum in wallboard. The Agency believes that these beneficial uses provide significant opportunities to advance Sustainable Materials Management (SMM).

Background

Beneficial use of industrial materials has the potential to provide economic benefits, preserve virgin resources, and avoid negative environmental impacts associated with the acquisition and processing of virgin materials. Beneficially using these materials presents significant opportunities to advance SMM and the Agency's SMM Program. The SMM Program supports the productive and sustainable use/reuse of resources throughout all stages of their lifecycles, from resources acquisition through disposal. The SMM Program seeks to avoid or minimize impacts to the environment while accounting for economic

efficiency and social considerations. CCRs are one of the industrial materials produced in the greatest quantity each year. The beneficial use of CCRs when conducted in a manner protective of human health and the environment can advance these SMM goals.

While the beneficial use of CCRs has been shown to have economic and material benefits, the environmental impacts associated with their use must also be considered. To do this, EPA's Office of Solid Waste and Emergency Response (OSWER) developed the *Methodology for Evaluating Encapsulated Beneficial Uses of Coal Combustion Residuals* (US EPA, 2013a). While in this document the Agency has used the methodology to evaluate the beneficial use of CCRs in certain encapsulated uses, this methodology can be useful to states, tribes, local governments, the public, and the regulated community for making informed decisions about any encapsulated beneficial uses of CCRs. The methodology has undergone an independent external letter peer review. A summary of the comments received from peer reviewers is available in the document *Peer Review Summary Report: Independent External Peer Review of the Preliminary Draft Report Methodology for Evaluating Encapsulated Beneficial Uses of Coal Combustion Residuals* (US EPA, 2012a). Responses to these comments are available in the document *Responses to External Peer Review Comments: Methodology for Evaluating Encapsulated Beneficial Uses of Coal Combustion Residuals* (US EPA, 2013b).

The methodology establishes a series of five steps that can be used to determine whether environmental releases of COPCs from an encapsulated beneficial use product made with CCRs are comparable to or lower than those from an analogous product made without CCRs, or are at or below relevant regulatory and health-based benchmarks developed for human and ecological receptors, during use by the consumer. The methodology allows evaluation of the range of possible encapsulated beneficial uses for any CCR. As developed, the methodology is quite flexible. The party conducting the evaluation can choose to begin at the first step and follow the methodology in the order presented or, based on the type and amount of data available on the CCR and corresponding product, can choose to begin the evaluation at any other step of the methodology. If releases of COPCs from the CCR beneficial use are found to be comparable to or lower than those from an analogous non-CCR product, or are at or below relevant regulatory and health-based benchmarks at any step of the methodology, then no further evaluation is necessary for those particular COPCs.

Summary of the Analysis

The Agency used the methodology to evaluate the potential environmental impacts associated with fly ash concrete and FGD gypsum wallboard. These beneficial use products may be variable in their composition. However, this evaluation only addressed those products that meet relevant physical and performance standards established by voluntary consensus standard bodies; that conform to specific design criteria identified in this evaluation (e.g., $\leq$ a 40 percent fly ash replacement rate); and that incorporate fly ash and FGD gypsum from common pollution control devices used in the United States. This evaluation also did not address products that contain additional additives or industrial materials that may alter releases from the products. In this specific evaluation, the Agency began with the first step and followed subsequent steps in the order presented in the methodology. The findings at each step of this specific conservative evaluation are summarized below.

Step 1 (Literature Review and Data Collection): From the available literature, the evaluation identified the following releases that may occur during use of fly ash concrete and FGD gypsum wallboard: 1) generation of dust, 2) emanation to air, 3) leaching to ground and surface water, and 4) decay of naturally occurring radionuclides (also referred to simply as radioactive decay). In addition, this literature review found several existing evaluations to be of sufficient applicability and quality to rely upon in the current evaluation. For fly ash concrete, the findings from these evaluations were used to eliminate radioactive decay from further consideration. For FGD gypsum wallboard, the findings from these existing evaluations were used to eliminate all releases from further consideration, except for emanation to air. The evaluation then used the available literature to identify COPCs for each of the remaining releases.

Step 2 (Comparison of Available Data): The Step 2 evaluation used the data identified in Step 1 to compare releases from fly ash concrete and FGD gypsum wallboard to releases from their respective analogous products (i.e., portland cement concrete and mined gypsum wallboard) during use by the consumer. The evaluation found that concentrations of silver and manganese in dust from fly ash concrete were comparable to or lower than those in dust from portland cement concrete. The evaluation also found that concentrations of arsenic, cadmium, lead, molybdenum, and thallium in leachate from fly ash concrete and portland cement concrete were comparable. Therefore, the evaluation did not carry these COPCs forward for further consideration, but retained all other COPCs from fly ash concrete and FGD gypsum wallboard for further consideration.

Step 3 (Exposure Review): The Step 3 evaluation reviewed the releases carried forward from Step 2 to identify exposures that may occur during use of the product. Where multiple exposure pathways associated with a given release were identified, the evaluation retained only those pathways likely to drive exposures for further consideration. The evaluation did not eliminate any releases or associated COPCs at this step.

Step 4 (Screening Assessment): The Step 4 evaluation conducted a screening assessment for each exposure pathway carried forward from Step 3 of the evaluation. The evaluation used conservative (i.e., likely to overestimate exposures) environmental, fate and transport, and exposure data to estimate COPC exposures that may occur during use of the CCR beneficial use products. The evaluation then compared these conservative exposure concentrations to relevant screening benchmarks to determine whether to conduct more in-depth evaluation. At the end of this step, the evaluation found that all remaining COPCs were below relevant screening benchmarks. Therefore, this evaluation did not proceed to the final Step 5 (Risk Assessment).

Conclusion: At the end of Step 4 the analysis was concluded as all identified COPCs had been eliminated in Steps 1 through 4. Thus, based on application of the methodology and the lines of evidence set forth in this document, the evaluation concludes that environmental releases of COPCs from CCR fly ash concrete and FGD gypsum wallboard during use by the consumer are comparable to or lower than those from analogous non-CCR products, or are at or below relevant regulatory and health-based benchmarks for human and ecological receptors.

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Abbreviations and Acronyms

AASHTO	American Association of State Highway and Transportation Officials
ACAA	American Coal Ash Association
ACH	Air Changes per Hour
AEA	Air Entrainment Admixture
ANSI	American National Standards Institute
ASHRAE	American Society of Heating, Refrigerating and Air Conditioning Engineers
ASTM	American Society for Testing and Materials
ATSDR	Agency for Toxic Substances and Disease Registry
AWQC	Ambient Water Quality Criteria
BCF	Bioconcentration Factor
CalEPA	California Environmental Protection Agency
CASRN	Chemical Abstract Service Registry Number
CCR	Coal Combustion Residual
COPC	Constituent of Potential Concern
CPSC	Consumer Product Safety Commission
CV-AFS	Cold Vapor Atomic Fluorescence Spectrometry
DOE	United States Department of Energy
DOT	United States Department of Transportation
EC	European Commission
Eco-SSL	Ecological Soil Screening Levels
EERC	University of North Dakota Energy and Environmental Research Center
EPA	United States Environmental Protection Agency
EPRI	Electric Power Research Institute
FBC	Fluidized Bed Combustion
FGD	Flue Gas Desulfurization
FHWA	Federal Highway Administration
HBN	Health-Based Number
HEI	Highly Exposed Individual
HELP	Hydrologic Evaluation of Landfill Performance
HPS	Health Physics Society
HQ	Hazard Quotient
IAEA	International Atomic Energy Agency
IBC	International Building Code
ICC	International Code Council
ICP	Inductively Coupled Plasma
ICRP	International Commission on Radiological Protection
IRIS	Integrated Risk Information System
IWEM	Industrial Waste Evaluation Model
LEAF	Leaching Evaluation Assessment Framework
LOI	Loss of Ignition

MCL	Maximum Contaminant Level
MDL	Method Detection Limit
ML	Minimum Level of Quantitation
NJDEP	New Jersey Department of Environmental Protection
NCRP	National Council on Radiological Protection
NOAA	National Oceanic and Atmospheric Administration
NOAEL	No Observable Affects Evaluation Level
NPDWR	National Primary Drinking Water Regulation
NRC	Nuclear Regulatory Commission
OSWER	Office of Solid Waste and Emergency Response
PCA	Portland Cement Association
PFA	Pulverized Fuel Ash
PPRTV	Provisional Peer Reviewed Toxicity Values for Superfund
PSI	Pounds per Square Inch
RAGS	Risk Assessment Guidance for Superfund
RfC	Reference Concentration
SMM	Sustainable Materials Management
SPLP	Synthetic Precipitation Leaching Procedure
T3	Third Trophic Level
T4	Fourth Trophic Level
TCLP	Toxicity Characteristic Leaching Procedure
TEL	Threshold Effect Level
TENORM	Technologically-Enhanced, Naturally Occurring Radioactive Materials
TL	Trophic Level
UNSCEAR	United Nations Scientific Committee on the Effects of Atomic Radiation
U.S.	United States
UKHPA	United Kingdom Health Protection Agency
UKNRPB	United Kingdom National Radiation Protection Board
USGS	United States Geological Survey

Introduction

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. Beneficial use, as defined in this document, is the reuse of CCRs in a product that: 1) provides a functional benefit; 2) meets relevant product specifications and performance standards for the proposed use; and 3) may replace virgin raw materials in a product on the market (referred to as an "analogous product" or "analogous non-CCR product"), thus conserving natural resources that would otherwise need to be obtained through other practices, such as extraction. The reason CCRs can be used to replace virgin materials is that they possess physical and chemical properties similar to those of the virgin materials.

The United States Environmental Protection Agency ("EPA" or "the Agency") Sustainable Materials Management (SMM) Program supports the productive and sustainable use/reuse of resources throughout all stages of their life cycles, from resource acquisition through disposal. The SMM Program seeks to avoid or minimize impacts to the environment while accounting for economic efficiency and social considerations. The beneficial use of CCRs, when done in a manner protective of human health and the environment, can advance these goals. The purpose of this beneficial use evaluation is to determine whether environmental releases of constituents of potential concern (COPCs) from these encapsulated[1] CCR products are comparable to or lower than those from analogous products made without CCRs, or are at or below relevant regulatory and health-based benchmarks for human and ecological receptors, during use by the consumer. This document details the evaluation process as well as the findings and conclusions of this beneficial use evaluation.

Selection of Beneficial Uses for Evaluation

CCR is a broad term used to refer to the byproducts generated either directly by coal combustion or as a result of applying certain pollution control devices to emissions from coal-fired combustion units. Types of CCRs generated from coal combustion include fly ash, bottom ash, boiler slag, flue gas desulfurization (FGD) waste, and fluidized bed combustion (FBC) waste. All coal-fired electric utility plants in the United States generate at least one of these types of CCRs. Each different type of CCR has unique properties and, as a result, different potential uses. This evaluation chose to examine those encapsulated beneficial uses that divert the greatest quantity of CCRs from disposal and have been most extensively studied in the literature. The following text provides the rationale for the selection of the CCR products evaluated.

[1] In the June 2010 Proposed Rule, Hazardous and Solid Waste Management System; Identification and Listing of Special Wastes; Disposal of Coal Combustion Residuals from Electric Utilities ("the 2010 Proposed CCR Disposal Rule") (75 FR 35127), the Agency defined encapsulated beneficial use as one that binds the CCRs into a solid matrix that minimizes their mobilization into the surrounding environment. Examples of encapsulated uses are concrete, wallboard, and brick. In contrast, unencapsulated beneficial uses include road embankments, structural fills, or agricultural applications (e.g., substitute for lime).

The American Coal Ash Association (ACAA) conducts a voluntary, annual survey of the coal-fired electric utility industry to track the quantities of CCRs generated and beneficially used. According to the latest survey, the electric utility industry generated nearly 110 million tons of CCRs during the 2012 calendar year. Approximately 39 million tons of these CCRs were identified by ACAA as beneficially used in either encapsulated or unencapsulated products. An additional 12.8 million tons were placed in mine-fill operations, while the remaining 57.8 million tons were disposed of in landfills and surface impoundments (ACAA, 2013).[2]

Based on the beneficial use rates reported by ACAA, the evaluation chose fly ash used as a direct substitute for portland cement during the production of concrete (referred to as "fly ash concrete") and FGD gypsum used as a replacement for mined gypsum in wallboard (referred to as "FGD gypsum wallboard") during use by the consumer. Specifically, the 2012 ACAA survey indicates that the largest encapsulated beneficial uses of CCRs, by more than a factor of two, are fly ash used in "concrete/concrete products/grout" (11.8 million tons) and FGD gypsum used in "gypsum panel products" (7.6 million tons). That is, these two beneficial uses make-up nearly 50 percent of the total amount of CCRs beneficially used on an annual basis. While fly ash and FGD gypsum may not be the only CCR or industrial material that may be beneficially used in concrete or wallboard, this evaluation does not address the beneficial use of other industrial materials. This evaluation also draws no conclusions about other beneficial uses of fly ash and FGD gypsum.

Properties of Fly Ash and FGD Gypsum

The following subsection describes the production process and properties of fly ash and FGD gypsum as well as the associated beneficial uses evaluated in this document. **Figure 1** illustrates a generalized layout of a coal-fired plant and the collection points for fly ash and FGD gypsum.

Fly Ash

Fly ash is the fraction of combusted coal that becomes suspended in plant flue gases. The fly ash available for beneficial use is captured primarily by mechanical particulate collection devices, such as an electrostatic precipitator or baghouse. The remaining fly ash that passes through these particulate collection devices either escapes into the atmosphere or is captured through sulfur dioxide [SO_2] control devices (i.e., scrubbers), resulting in its incorporation into the FGD solids. The chemical composition of the beneficially used fly ash is variable and dependent on multiple factors, such as the geographic source of the coal burned.

[2] In 2012, the survey response rate was equivalent to 59 percent of the total US 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 United States coal-fired generation capacity reported by the Energy Information Administration (EIA) in 2012. These data are available online at: http://www.eia.gov/coal/data.cfm. Reported beneficial use rates were extrapolated for the entire industry sector using the 2012 survey data, historical ACAA survey data, EIA data, and other miscellaneous data sources. The survey groups similar beneficial uses into generalized categories. As a result, a given ACAA category may include some data on beneficial uses beyond those evaluated in this document.

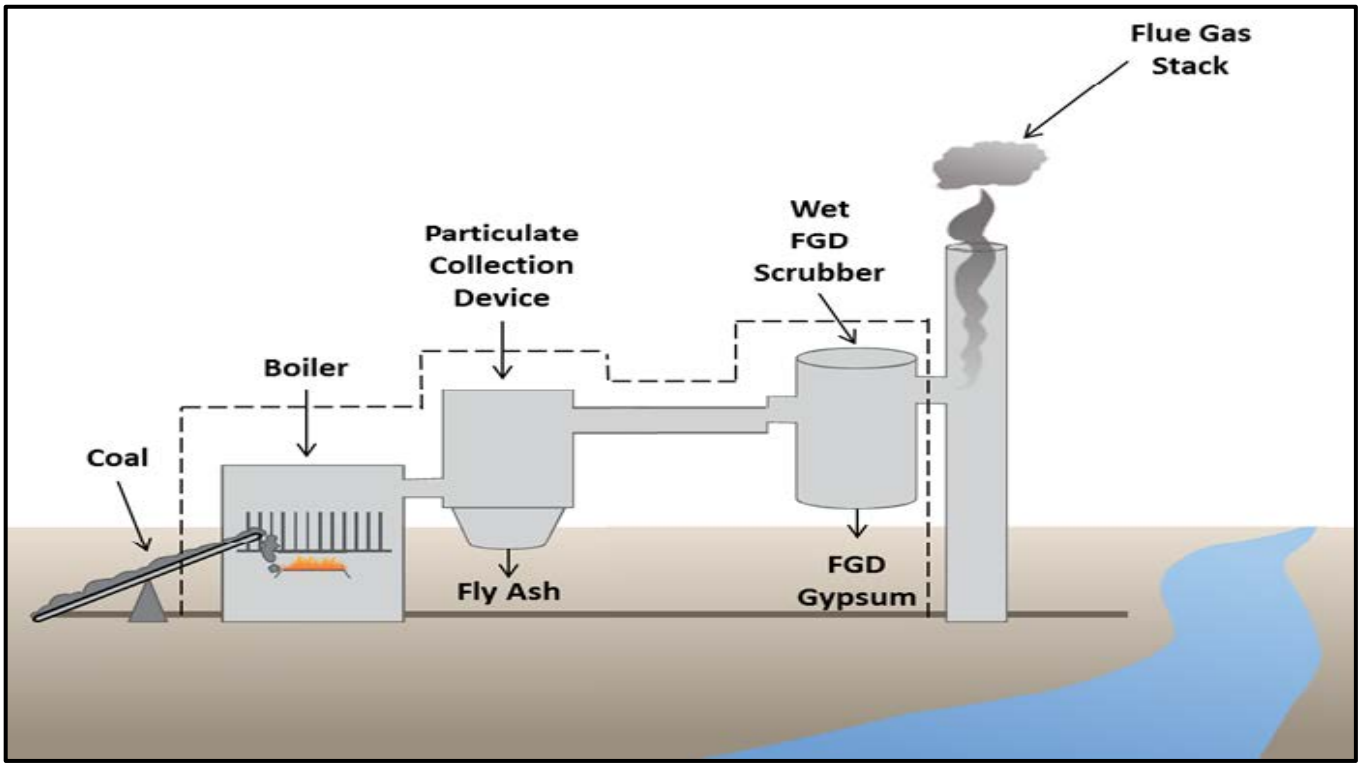


Figure 1: Diagram of generic coal combustion processes.

Fly ash may be used as a partial substitute for the portland cement in concrete because it is a pozzolan, a material that reacts with calcium oxide [CaO], also known as lime, in the presence of water to produce a cementitious compound. Silica dioxide [SiO_2], aluminum trioxide [Al_2O_3], and iron oxide [Fe_2O_3] are the primary chemical constituents that may contribute to this reaction. In addition, fly ash may contain significant amounts of lime, causing it to be self-cementing in the presence of water. The American Society for Testing and Materials (ASTM) classifies fly ashes based on the amount of lime present as either Class F (low lime/high iron) or Class C (high lime/high calcium) (ASTM C618). Depending on the composition of the fly ash and the intended use of the fly ash concrete, a wide range of cement substitution rates may be considered. Class F fly ash is often used to replace portland cement at rates between 15 percent and 25 percent by mass, while Class C fly ash is often used to replace portland cement at rates between 15 percent and 40 percent by mass (PCA, 2003). However, specific replacement rates vary based both on the characteristics of the specific fly ash and on the desired characteristics of the concrete. ASTM specifies a maximum cement replacement rate of 40 percent for blended fly ash-cement mixtures based on engineering specifications (ASTM C595). Therefore, the findings of this evaluation are limited to portland cement replacement rates at or below 40 percent by mass.

Flue Gas Desulfurization Gypsum

Coal-fired plants employ a number of different air pollution control devices (generally referred to as “flue gas desulfurization units” or “scrubbers”) to reduce sulfur dioxide emissions. These devices differ in how they remove sulfur dioxide, but all generate FGD waste. This waste ranges from a dry powder to a wet sludge. FGD gypsum is a subset of the wet sludges produced by FGD units. During the generation of FGD gypsum, fly ash is initially removed from flue gas to the extent practicable using mechanical

collection devices. Next, the flue gas is sprayed with a wet limestone-based reagent. This reagent reacts with the sulfur dioxide in the flue gas, limiting the amount of the sulfur dioxide and remaining fly ash that can escape into the atmosphere. The chemical composition of the resulting sludge is dependent on the amount of oxygen available during the reaction. In the absence of oxygen, the reaction produces calcium sulfite hemihydrate [$CaSO_3 \cdot ½H_2O$]. In the presence of oxygen, the reaction produces calcium sulfate dihydrate [$CaSO_4 \cdot 2H_2O$], also known as gypsum. To convert as much of the sludge to gypsum as possible, the sludge may undergo forced oxidation, driven by air pumped into the chamber during the reaction.

FGD gypsum may be used as a full substitute for mined gypsum in wallboard (i.e., drywall) because the primary chemical constituent, calcium sulfate dihydrate, is identical in both materials. The generation of FGD gypsum may allow greater control over the chemical composition of the final gypsum product and, as a result, FGD gypsum may have higher gypsum purity than mined gypsum. However, FGD gypsum may contain some impurities that are not found in mined gypsum. Fly ash is one such impurity, and can result in accelerated wear to the production machinery and physical defects in the final products. As a result, common market specifications established by North American wallboard manufactures limit the amount of fly ash allowed in the FGD gypsum used in wallboard to one percent by mass (Henkels and Gaynor, 1996).

Overview of Methodology Steps

The beneficial use evaluation of fly ash concrete and FGD gypsum wallboard follows the steps laid out in the *Methodology for Evaluating Encapsulated Beneficial Uses of Coal Combustion Residuals* (US EPA, 2013a). This methodology has undergone an independent external letter peer review. A summary of the comments received from peer reviewers is available in the document *Peer Review Summary Report: Independent External Peer Review of the Preliminary Draft Report Methodology for Evaluating Encapsulated Beneficial Uses of Coal Combustion Residuals* (US EPA, 2012a). Responses to these comments are available in the document *Responses to External Peer Review Comments: Methodology for Evaluating Encapsulated Beneficial Uses of Coal Combustion Residuals* (US EPA, 2013b).

This methodology is flexible and allows evaluation of the range of possible encapsulated beneficial uses for any CCR. The evaluation process is divided into five individual steps. As developed, the party conducting the evaluation can choose to begin at the first step and follow the methodology in the order presented or, based on the type and amount of data available, can choose to begin the evaluation at any other step of the methodology. If, at any point in the evaluation, all releases of COPCs are found to be comparable to or lower than those from an analogous non-CCR product, or to be at or below relevant regulatory and health-based benchmarks, then no further evaluation of the CCR product is necessary. This specific evaluation began with the first step and followed subsequent steps in the order presented in the methodology.

Step 1 (Literature Review and Data Collection): This step involves the collection and review of available literature on a specific CCR and associated beneficial use. The purpose of this step is to establish whether existing evaluations are sufficient to demonstrate that releases from the CCR products under evaluation are comparable to or lower than those from analogous products, or are at or below

relevant regulatory and health-based benchmarks, and to collect data on COPCs that may be present in and released from the CCR products, but were not sufficiently addressed by existing evaluations.

Step 2 (Comparison of Available Data): This step involves using the data collected in Step 1 to conduct a comparison of the COPC releases from the CCR products with those from the analogous products that they replace. The purpose of this step is to determine whether COPC releases from the CCR products are comparable to or lower than those from an analogous product.

Step 3 (Exposure Review): This step involves the review of those COPCs that were not comparable to or lower than those from an analogous product and were carried forward from Step 2. The purpose of this step is to identify potential exposure pathways, determine whether these exposure pathways are complete, and to develop a conceptual exposure model to organize and communicate this information.

Step 4 (Screening Analysis): This step involves a screening analysis of the COPC exposures carried forward from Steps 2 and 3. This screening uses a combination of conservative (i.e., likely to overestimate exposures) environmental, fate and transport, and exposure data to estimate the magnitude of COPC concentrations at the point of exposure. The analysis then compares these conservative COPC concentrations to relevant regulatory and health-based screening benchmarks. The purpose of this step is to eliminate any COPC exposures that do not warrant further consideration with more realistic, resource intensive modeling.

Step 5 (Risk Analysis): This final step involves a revised analysis of COPC exposures that were found to be above screening benchmarks in Step 4. This analysis is intended to be more realistic than the screening analysis, and is conducted using environmental, fate and transport, and exposure data that are more representative of real world conditions. The evaluation uses these revised COPC concentrations to estimate corresponding risks. The purpose of this step is to reduce conservatisms remaining in the evaluation to a level at which a final conclusion can be made.

Step 1: Literature Review and Data Collection

This section applies the first step of the methodology to the evaluation of fly ash concrete and FGD gypsum wallboard. This step involves collecting and reviewing the available literature relevant to a specific CCR and its beneficial use. The purpose of this step is twofold. The first purpose is to determine whether any existing evaluations have already demonstrated that releases of COPCs from fly ash concrete or FGD gypsum wallboard are comparable to or lower than those from analogous products, or are at or below relevant regulatory and health-based benchmarks. The second purpose is to collect data on the COPCs present in releases from fly ash concrete and FGD gypsum wallboard that have not been sufficiently addressed by the existing evaluations. The following subsections provide a discussion of the existing evaluations used to identify the releases and associated COPCs that required further evaluation, followed by a brief summary of the data sources relied upon in the current evaluation.

1.1 Fly Ash Concrete

1.1.1 Existing Evaluations for Fly Ash Concrete

The current beneficial use evaluation reviewed all existing evaluations identified from the available literature according to the recommendations of *Summary of General Assessment Factors for Evaluating the Quality of Scientific and Technical Information* (US EPA, 2003a).[3] The focus of this review was to determine whether these existing evaluations could form the basis for defensible conclusions regarding fly ash concrete. The review determined whether the existing evaluations pertained to fly ash concrete, clearly and sufficiently explained the data and assumptions relied upon, accounted for major sources of uncertainty and variability, and had undergone an independent review in some form. The remainder of this subsection summarizes the existing evaluations used to identify releases and associated COPCs for further consideration. Under the title of each evaluation, a brief summary of relevant findings is provided. Where multiple existing evaluations were pertinent to a given topic, all the summaries are combined under a list of the evaluation titles.

US EPA (1998): *Supplemental Report to Congress on Remaining Wastes from Fossil Fuel Combustion Technical Background Document: Beneficial Use of Fossil Fuel Combustion Wastes*

This report 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 report addresses the beneficial use of CCRs, it is directly applicable to the current evaluation of fly ash concrete. Therefore, each identified type of release was retained for further consideration.

[3] EPA developed this document in response to guidelines issued by OMB (US OMB, 2002) under section 515 of the Treasury and General Government Appropriations Act for Fiscal Year 2001 (Public Law 106-554; H.R. 5658).

US EPA (1999): *Report to Congress: Wastes from the Combustion of Fossil Fuels: Volume 2 - Methods, Findings, and Recommendations*

This report reviewed all of the data available to the Agency at the time of publication on releases from CCRs generated in the United States. The data indicated that all concentrations of organic constituents, such as polyaromatic hydrocarbons and dioxins, were near or below analytical detection limits both in CCRs and in leachate released from CCRs. Based on these data, the report concluded that organic constituents are not COPCs associated with CCRs. Consideration of updated toxicity values does not alter the conclusions of this report. Furthermore, no additional data have been identified since the completion of this report that would indicate the potential for higher organic levels. Although this report addressed CCRs, the conclusions are also applicable to fly ash concrete. The beneficial use of fly ash in concrete will dilute concentrations of any organic constituents present in the fly ash through mixing with other concrete components. In addition, these organic constituents are not volatile under standard environmental conditions and are often highly hydrophobic, making a release of these complex organic compounds from fly ash concrete at rates higher than those from pure fly ash unlikely. Therefore, the current evaluation eliminated organic constituents from further consideration.

US EPA (2002): *Constituent Screening for Coal Combustion Wastes*

US EPA (2010a): *Human and Ecological Risk Assessment of Coal Combustion Wastes*

US EPA (2002), as discussed in US EPA (2010a), evaluated transport of CCRs disposed in uncovered landfills by wind and precipitation. This screening assessment found that all potential exposures were below levels of concern. Although this 2002 evaluation addressed CCRs that were disposed, the findings are also applicable to fly ash concrete. The beneficial use of fly ash in concrete will dilute constituent concentrations present in the fly ash through mixing with other raw materials, and will also reduce the rate of release because of the high strength of the intact concrete matrix. Therefore, the results of this 2002 evaluation provide a conservative estimate of dust release from concrete. However, more than a decade has passed since the 2002 screening assessment was conducted. Since that time, the Agency has obtained additional data through public comments and published literature that characterize constituent concentrations in fly ash, including data that reflect the effects of some new pollution control technologies and plant configurations installed in response to updated Clean Air Act requirements. In addition, revised toxicity values have resulted in updated human health and ecological benchmarks. As a result, the current evaluation retained generation of dust for further consideration, and identified all constituents for which sufficient data were available as COPCs for this release. These include aluminum, antimony, arsenic, barium, beryllium, boron, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, molybdenum, nickel, selenium, silver, strontium, thallium, uranium, vanadium, and zinc. While other inorganic constituents may also be present in CCRs, they were not retained for further evaluation. Some constituents were not retained because of the absence of human health and ecological toxicity values (i.e., calcium, chloride, magnesium, phosphate, potassium, sodium, silicon, sulfate, sulfide). Other constituents were not retained because the newly available data, which were either for CCRs other than fly ash or for fly ash mixed with other CCRs, were not representative of the CCRs assessed in

the current evaluation, and do not add to the information used to draw conclusions in U.S EPA (2010a) (i.e., cyanide, fluoride, nitrate/nitrite).

US EPA (2010a): *Human and Ecological Risk Assessment of Coal Combustion Wastes*

US EPA (2012b): *The Impact of Coal Combustion Fly Ash Used as a Supplemental Cementitious Material on the Leaching of Constituents from Cements and Concretes*

Kosson et al. (2013): *pH-dependent Leaching of Constituents of Potential Concern from Concrete Materials Containing Coal Combustion Fly Ash*

US EPA (2010a) conducted a national evaluation of potential human health and ecological risks resulting from releases to ground and surface water from CCRs disposed in landfills and surface impoundments. The risk assessment used leachate data from a range of different analytical methods to consider a wide range of management scenarios and environmental conditions. Leaching from disposed fly ash does not directly reflect leaching from fly ash concrete placed on the land. However, the results of US EPA (2010a) are still applicable to this evaluation of fly ash concrete based on the findings of Kosson et al. (2013), which evaluated the leaching behavior of fly ash concretes and demonstrated that these CCR products consistently leach at lower levels than fly ash alone when subjected to similar environmental conditions. Thus, the current evaluation used the findings of US EPA (2010a) to identify a conservative set of COPCs for leaching from fly ash concrete placed on the land.

The modeled results for unlined landfills provide a conservative, yet appropriate, surrogate for fly ash concrete. Although US EPA (2010a) found leaching from surface impoundments to be higher than from landfills, the effects of large hydraulic heads that drive leaching from surface impoundments would be greatly diminished by the low permeability of an intact concrete matrix. Therefore, the current evaluation did not consider leaching results for surface impoundments. The results for unlined landfills show that antimony, arsenic, boron, cadmium, lead, molybdenum, selenium, and thallium may be released at levels posing unacceptable risk to downgradient receptors. Therefore, the evaluation retained these eight constituents as COPCs for further consideration. US EPA (2012b) also conducted a conservative screening analysis by comparing the undiluted leachate from fly ash mortars and concretes to screening benchmarks. However, because US EPA (2010a) conducted more robust, full-scale modeling that took into account dilution and attenuation in the environment, it provided a more realistic estimate of exposures. Therefore, with one exception, the current evaluation did not rely on the results of US EPA (2012b) to identify COPCs. Chromium was retained as a COPC based on the screening results of US EPA (2012b). US EPA (2010a) did not evaluate cancer risks from chromium because the revised oral cancer benchmark was not available at that time.

Long et al. (2012): *Potential Indoor Air Exposures and Health Risks from Mercury Off-Gassing of Coal Combustion Products Used in Building Materials*

Long et al. (2012) measured the rate at which concrete made with and without fly ash emit mercury vapor, and used these measurements to calculate resulting indoor air concentrations in a school. The report then compared these concentrations to health-based benchmarks. The evaluation concluded that potential indoor air concentrations were below levels of concern. Because Long et al. (2012) specifically addressed releases from fly ash concrete, the results are directly applicable to the current

evaluation. However, a review of the documented methodology found that it did not address the use of fly ash concrete in residential settings, potentially resulting in an underestimation of high-end exposures. Furthermore, the relatively small number of fly ashes evaluated in the study introduces uncertainty as to whether the measured mercury emanation rates adequately characterize the range of potential emanation rates from fly ash concrete. Therefore, the current evaluation retained mercury as a COPC for emanation to air.

Ingersoll (1983): *A Survey of Radionuclide Contents and Radon Emanation Rates in Building Materials Used in the United States*

United States Geological Survey (USGS) (1997): *Radioactive Elements in Coal and Fly Ash: Abundance, Forms, and Environmental Significance*

Zielinski et al. (1998): *Uranium in Coal and Fly Ash: Abundance, Forms, and Environmental Significance*

United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) (1982, 1988, 1993; 2000; 2008): *Ionizing Radiation: Sources and Effects*

United Kingdom National Radiological Protection Board (UKNRPB) (2001): *Radiological Impact on the UK Population of Industries Which Use or Produce Materials Containing Enhanced Levels of Naturally Occurring Radionuclides: Part 1 - Coal-fired Electricity Generation*

International Atomic Energy Agency (IAEA) (2003): *Extent of Environmental Contamination by Naturally Occurring Radioactive Materials (NORM) and Technical Option for Mitigation*

United Kingdom Health Protection Agency (UKHPA) (2004): *Radiological Study of Pulverized Fuel Ash (PFA) from UK Coal-fired Power Stations*

Kolver et al. (2005a): *Radon Exhalation Of Cementitious Materials Made with Coal Fly Ash: Part 1 - Scientific Background and Testing of the Cement and Fly Ash Emanation*

Kolver et al. (2005b): *Radon Exhalation Of Cementitious Materials Made with Coal Fly Ash: Part 2 - Testing Hardened Cement Fly Ash Pastes*

National Commission for Radiological Protection and Measurements (NCRP) (2009): *Ionizing Radiation Exposure of the Population of the United States Report 160*

Kosson et al. (2013): *pH-dependent Leaching of Constituents of Potential Concern from Concrete Materials Containing Coal Combustion Fly Ash*

A broad range of domestic and international evaluations were identified that address radiation from coal, fly ash, or fly ash concrete. Although some of these evaluations do not directly address fly ash concrete, the results are applicable to the current evaluation because the beneficial use of fly ash in concrete will dilute radionuclide concentrations present in the fly ash through mixing with other raw materials. These evaluations include peer-reviewed publications, guidance documents, and voluntary standards that have been developed and reviewed by experts in the field of radiation and health physics. This body of work extends back nearly forty years and has already been well summarized through the literature reviews contained in more recent existing evaluations. Therefore, only these key recent evaluations, which form the basis for the conclusions in this beneficial use evaluation, are summarized in the following text.

International organizations have grouped radiation exposures into three different categories: (1) planned introduction and operation of sources, (2) unexpected emergencies that may arise during a planned situation or from a malicious act, and (3) situations that already exist when a decision on

control has to be taken.[4] The historical use of fly ash in concrete would fall under group (3), while the continued use would fall under group (1).[5] A number of national and international organizations develop and maintain guidance on managing public exposures to everyday sources of radiation. Evaluations of radiation exposures often incorporate the recommendations of these guidances, and several of the existing evaluations identified were conducted by one of these organizations to everyday sources of radiation. In the United States, the Nuclear Regulatory Commission (NRC) regulates source and byproduct materials, and can exempt certain products, devices, or equipment that contain radioactive materials from requiring a license (NRC, 2001). EPA has promulgated maximum contaminant limits (MCLs) for radiation levels in public drinking water, and has developed recommendations for mitigating radon levels in indoor air. The American National Standards Institute (ANSI), together with the Health Physics Society (HPS), developed Standard N13.53 for technologically enhanced sources of naturally occurring radiation, based on a review of existing radiation protection standards and guidance from relevant US and European organizations (ANSI/HPS, 2009). International organizations have also developed guidance that has been adopted by many countries. The European Commission (EC) developed Basic Safety Standards Directive 96/29, which recommends exposure limits for various sources of radioactivity and authorizes the exemption of specific practices from regulatory controls (EC, 1999; 2001). The International Commission on Radiological Protection (ICRP) developed the International System of Radiological Protection, used world-wide as the basis for radiological protection standards, legislation, guidelines, programs, and practice (ICRP, 2007). The IAEA develops safety standards and, based on these standards, issues guidance and technical documents on radiation protection. While these organizations set varying exposure limits for different scenarios (e.g., clean-up sites, medical procedures, consumer products), virtually all guidance developed incorporates the principle of "as low as reasonably achievable" to minimize public exposures while taking into account health, economic, and societal factors.

- Ingersoll (1983) measured the concentrations of uranium, thorium, and potassium present in, as well as the rate at which radon emanates from portland cement concrete and other common building materials made with virgin materials collected from ten major metropolitan areas across the United States. The study concluded that each of these building materials contribute only a small fraction of the total radon levels typically measured in US homes. The study also noted that radon measurements previously collected for fly ash concretes fell within the range measured for the portland cement concretes.
- UNSCEAR (1982, 1988, 1993, 2000, 2008) each summarized the available literature on sources of radiation exposure. These reports compiled a great deal of information on the radionuclide activities in coal, fly ash, concrete, and other building materials collected from across the globe. The data compiled in these reports show the average radionuclide activity of fly ash generated in the United States falls around the upper bound of the range measured in soils, but is generally lower than fly ashes generated in other parts of the globe. The 1988 report also summarized

[4] As discussed in International Commission on Radiological Protection Report 103 (ICRP, 2007).

[5] Fly ash has been a commonly used raw material in concrete for at least the past 80 years. The Hoover Dam is one of the first recorded projects in 1929 (http://www.lmcc.com/concrete_news/0607/five_minute_classroom_fly_ash.asp).

available literature on radon emanation from fly ash concrete. Based on a number of studies that showed either reduced or unchanged radon emanation rates from concrete amended with fly ash, this 1988 report concluded that the use of fly ash in building materials should not result in any additional radiation exposure beyond that from standard portland cement concrete.

- USGS (1997) reviewed data in the US Coal Quality Database on the uranium concentrations in coals mined from the Western United States and the Illinois Basin. This study used the data to estimate the uranium content of fly ash by assuming that concentrations were magnified by a factor of ten during combustion, based on values previously reported in the literature. This study compared the estimated uranium concentrations for fly ash to those previously reported in the literature for common rocks, such as granite and shale. Based on this comparison, USGS concluded that the radionuclide concentrations in fly ash are similar to those found in common rocks. IAEA (2003) reviewed the data presented in USGS (1997) and drew similar conclusions. USGS (1997) also summarized data later available in Zielinski et al. (1998) on leaching of uranium and radium from fly ash.[6] Zielinski et al. (1998) analyzed the physical and chemical structure of fly ash and concluded that the long-term leaching of uranium will be inhibited by the glassy structure of the ash, while long-term leaching of radium will be inhibited by the formation of insoluble sulfate complexes. Available uranium leachate samples corroborated these predictions. Fly ash samples were collected at one facility from various points in the exhaust stream and at different times. These fly ash samples ranged in pH from about 3 to 12, and had uranium concentrations toward the upper bound predicted by the US Coal Quality Database. Samples were subjected to column and batch leach tests, which showed concentrations of uranium to be below the corresponding MCL over the pH range relevant to concrete. Based on the available evidence, the study concluded that radionuclides leached from fly ash are generally below levels of concern. It is important to note that all of these data represent pure fly ash. Based on the findings of Kosson et al. (2013), incorporation of fly ash into concrete will further limit leaching. In addition, none of the measured concentrations account for dilution and attenuation that will occur in the environment.
- UKNPRB (2001) calculated potential exposures to gamma radiation and radon resulting from the use of fly ash generated within the United Kingdom in concrete. The report considered exposures to a member of the public that spends majority of their time in a small room with four walls, ceiling, and floor made of fly ash concrete. The report found that potential exposures to gamma radiation and radon from fly ash concrete do not exceed benchmarks established in EC guidance. Both UKHPA (2004) and NCRP (2009) reviewed the data presented in UKNPRB (2001) and drew similar conclusions.
- Kolver et al. (2005a) reviewed and summarized the literature on radon emanation from concrete. This study identified several additional studies published since the 1988 UNSCEAR report that found the addition of fly ash either reduced or had little effect on radon emanation. Kolver et al. (2005b) measured the rate of radon emanation from mortars made with and without fly ash, with variable fly ash replacement rates. Measurements were collected at 7, 28, and 90 days after curing.

[6] USGS (1997) used barium as a chemical analog for radium.

This study showed that the radon emanation rate from all fly ash amended mortars was less than from those made with only portland cement. The study concluded that the increased densification caused by the inclusion of fly ash was responsible for these lower emanation rates.

All of the existing evaluations identified concluded that radiation exposures from fly ash concrete are not a major source of concern. Several of these existing evaluations compared fly ash concrete to analogous products and found that the potential exposures do not represent an appreciable addition to the background radiation that the general public is subjected to on an annual basis. Naturally occurring radionuclides are present throughout the environment in food, air, water, soil, consumer products, and even the human body. All natural resources used in building construction (e.g., cement blocks, bricks, granite, soil, rocks) contain some trace level of naturally occurring radionuclides. For example, the USGS concluded that "the radioactivity of typical fly ash is not significantly different from that of more conventional concrete additives or other building materials such as granite and red brick." The NCRP concluded that exposures from living in concrete buildings containing fly ash are "similar to calculations made for individuals living in a brick and masonry home. Consequently, it is assumed that the use of [coal ash] in building materials has not substantially increased the average dose to an individual in the population residing in a building constructed with brick or masonry materials." Several of these existing evaluations also evaluated the magnitude of potential exposures that may result from fly ash concrete and found them to be in line with existing guidance. For example, the UKHPA concluded that exposures to "…members of the public from the use of [fly ash] in building materials is negligible." The cumulative body of evidence provided by these evaluations is considered sufficient to demonstrate that radiation from fly ash concrete is either comparable to that from analogous products made without CCRs, or is at or below relevant benchmarks established by national and international standard-setting and regulatory bodies. Therefore, the current evaluation eliminated radionuclides from further consideration.

1.1.2 Data Collection for Fly Ash Concrete

The review of existing evaluations discussed in **Section 1.1.1** above identified potential releases and associated COPCs that the literature had not sufficiently addressed. Therefore, the current evaluation assembled data on these remaining releases and associated COPCs from the existing evaluations and other available literature. These data form the basis for the evaluation of COPC releases from fly ash concrete conducted in subsequent steps of this evaluation. The remainder of this section enumerates the major sources from which these data were drawn. **Appendix A** provides further discussion of each data source, along with a presentation of the corresponding raw data.

- **CCR Constituent Database:** This database contains all of the data collected in support of evaluations of CCRs since the Agency's 1988 *Report to Congress* (US EPA, 1988). The database contains information on the identity and concentrations of constituents that may be present in CCRs. The Agency's most recent evaluation of CCR disposal practices provides an overview of historical data collection efforts prior to the finalization of the 2010 *CCR Risk Assessment* (US EPA, 2010a). However, since the completion of this risk assessment, EPA has incorporated new data into the database as it has become available.

- **Eckert and Guo (1998):** *Heavy Metals in Cement and Cement Kiln Dust from Kilns Co-Fired with Hazardous Waste-Derived Fuel: Application of EPA Leaching and Acid-Digestion Procedures.* This literature source contains data on COPC concentrations in portland cement.
- **Garrabrants et al. (2013):** *Effects of Coal Fly Ash Use in Concrete on Mass Transport-Based Leaching of Potential Concern.* This literature source contains data on COPC leaching rates using EPA Leaching Environmental Assessment Framework (LEAF) Method 1315 from concretes made with fly ash and portland cement.
- **Golightly et al. (2005):** *Gaseous Mercury from Curing Concretes that Contain Fly Ash: Laboratory Measurements.* This literature source contains data on mercury concentrations in fly ash, as well as mercury emanation rates from fly ash and portland cement concretes.
- **Golightly et al. (2009):** *Fly Ash Properties and Mercury Sorbent Affect Mercury Release from Curing Concrete.* This literature source contains data on mercury concentrations in fly ash and portland cement, as well as mercury emanation rates from concretes made with fly ash and portland cement.
- **Kosson et al. (2013):** *pH-dependent Leaching of Constituents of Potential Concern from Concrete Materials Containing Coal Combustion Fly Ash.* This literature source contains data on COPCs using EPA LEAF Method 1313 from concrete made with fly ash and portland cement.
- **Pflughoeft-Hassett et al. (1993):** *Comparative Leaching of Midwestern Coal Fly Ash and Cement*. This literature source contains data on COPC concentrations in portland cement.
- **Portland Cement Association (PCA) (1992):** *An Analysis of Selected Trace Metals in Cement and Kiln Dust.* This literature source contains data on COPC concentrations in portland cement.

The current evaluation considered several other potentially relevant literature sources that contain data on leaching from fly ash concrete, but did not rely upon those sources in this beneficial use evaluation because more appropriate data were available. These include: Zhang et al., 1985; Rankers and Hohberg, 1991; Kanare and West, 1993; Church et al., 1995; UKDETR, 1998; Meji et al., 2001; Zhang et al. 2001; McCann et al., 2007; and Giergiczny and Krol, 2008. These studies provide leachate data collected with single pH leach tests. Examples of single pH leach tests include the Deionized Water Leaching Test (ASTM D3987-85), the Synthetic Precipitation Leaching Procedure (SPLP), and the Toxicity Characteristic Leaching Procedure (TCLP). These single pH data do not reflect the time-dependent release of COPCs from intact concrete. In addition, the current evaluation did not incorporate the leachate data from US EPA (2012b) because the majority of these samples are from fly ash mortars. The higher porosity of mortars can result in leaching that is not considered representative of fly ash concrete.

1.1.3 Summary of Releases Identified for Fly Ash Concrete

Based on the review of the available literature, the current evaluation initially identified four potential releases from fly ash concrete that may occur during use: 1) generation of dust, 2) emanation to air, 3) leaching to ground and surface water, and 4) decay of naturally occurring radionuclides. A review of existing evaluations found them to be of sufficient quality and applicability to eliminate decay of

naturally occurring radionuclides from further consideration. **Table 1-1** provides a summary of the releases retained for further consideration, and the COPCs associated with each.

Table 1-1: Releases from Fly Ash Concrete and Associated COPCs

COPC	Dust	Ground and Surface Water	Air
Aluminum	X	--	--
Antimony	X	X	--
Arsenic	X	X	--
Barium	X	--	--
Beryllium	X	--	--
Boron	X	X	--
Cadmium	X	X	--
Chromium	X	X	--
Cobalt	X	--	--
Copper	X	--	--
Iron	X	--	--
Lead	X	X	--
Manganese	X	--	--
Mercury	X	--	X
Molybdenum	X	X	--
Nickel	X	--	--
Selenium	X	X	--
Silver	X	--	--
Strontium	X	--	--
Thallium	X	X	--
Uranium	X	--	--
Vanadium	X	--	--
Zinc	X	--	--

COPC = constituent of potential concern
X = constituent requires further evaluation
-- = constituent requires no further evaluation

1.2 FGD Gypsum Wallboard

1.2.1 Existing Evaluations for FGD Gypsum Wallboard

The current beneficial use evaluation reviewed all existing evaluations identified in the available literature according to the recommendations of *Summary of General Assessment Factors for Evaluating the Quality of Scientific and Technical Information* (US EPA, 2003a).[7] The focus of this review was to determine whether these existing evaluations could form the basis for defensible conclusions regarding FGD gypsum wallboard. The review determined whether the existing evaluations pertained to FGD gypsum wallboard, clearly and sufficiently explained the data and assumptions relied upon, accounted for major sources of uncertainty and variability, and had undergone an independent review in some form. The remainder of this subsection summarizes the existing evaluations used to identify releases and associated COPCs for further consideration. Under the title of each evaluation, a brief summary of relevant findings is provided. Where multiple existing evaluations were pertinent to a given topic, all the summaries are combined under a list of the evaluation titles. Many of the existing evaluations are the same as presented in **Section 1.1**, but with a focus on FGD gypsum wallboard.

US EPA (1998): *Supplemental Report to Congress on Remaining Wastes from Fossil Fuel Combustion Technical Background Document: Beneficial Use of Fossil Fuel Combustion Wastes*

This report 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 report addresses the beneficial use of CCRs, it is directly applicable to the current evaluation of FGD gypsum wallboard. Therefore, each identified type of release was retained for further consideration.

US EPA (1999): *Report to Congress: Wastes from the Combustion of Fossil Fuels: Volume 2 - Methods, Findings, and Recommendations*

This report reviewed all of the data available to the Agency at the time of publication on releases from CCRs generated in the United States. The data indicated that all concentrations of organic constituents, such as polyaromatic hydrocarbons and dioxins, were near or below analytical detection limits both in CCRs and in leachate released from CCRs. Based on these data, the report concluded that organic constituents are not COPCs associated with CCRs. Consideration of updated toxicity values does not alter the conclusions of this report. Furthermore, no additional data have been identified since the completion of this report that would indicate the potential for higher organic levels. These organic constituents are not volatile under standard environmental conditions (e.g., temperature and atmospheric pressure), making a release of these complex organic compounds from FGD gypsum wallboard at rates higher than those from pure FGD gypsum unlikely. Therefore, the current evaluation eliminated organic constituents from further consideration.

[7] EPA developed this document in response to guidelines issued by OMB (US OMB, 2002) under section 515 of the Treasury and General Government Appropriations Act for Fiscal Year 2001 (Public Law 106-554; H.R. 5658).

Yost et al. (2010): *Lack of Complete Exposure Pathways for Metals in Natural and FGD Gypsum*

This evaluation identified potential releases from wallboard through the development of a conceptual exposure model. The evaluation concluded that, because wallboard is sealed within the ceiling or walls of a building during use, it is not exposed to most environmental media. The only media anticipated to come in contact with wallboard during standard use is indoor air. Based on these findings, the current evaluation limited the selection of COPCs to those with the potential to be released through emanation to indoor air or radioactive decay.

Consumer Product Safety Commission (CPSC) (2010a): *Final Report on an Indoor Environmental Quality Assessment of Residences Containing Chinese Drywall*

CPSC (2010b): *Staff Preliminary Evaluation of Drywall Chamber Test Results: Reactive Sulfur Gases*

The CPSC commissioned a series of evaluations in response to concerns raised by the public over the potential for imported Chinese wallboard to cause adverse health effects. In one evaluation, air samples were collected at 41 homes that had reported problematic wallboard and 10 homes that had reported no issues (CPSC, 2010a). The evaluation found houses with problem wallboards had higher indoor air concentrations of select volatile organic compounds (e.g., n-hexane) and volatile sulfur compounds (e.g., hydrogen sulfide and carbon disulfide).[8] However, these higher levels were generally below associated EPA health benchmarks. In a subsequent study, differences in the emanation rates of volatile sulfur gases from domestic and imported wallboards were evaluated in a laboratory setting (CPSC, 2010b). The preliminary results found considerably lower emanation rates of volatile sulfur gases from all domestic wallboards compared to the problematic imported wallboards. CPSC (2010b) did not specify whether any of the domestic wallboards evaluated were made with FGD gypsum. However, CPSC (2010a) specifically compared the characteristics of domestic mined and FGD gypsum wallboards. This evaluation found that the indicators of high volatile sulfur releases associated with the problematic imported wallboard were absent from all FGD gypsum wallboard samples. Based on these findings, the current evaluation eliminated these releases from further consideration.

Long et al. (2012): *Potential Indoor Air Exposures and Health Risks from Mercury Off-Gassing of Coal Combustion Products Used in Building Materials*

Long et al. (2012) measured the rate at which wallboard made with and without FGD gypsum emits mercury vapor and used these measurements to calculate resulting indoor air concentrations in a home and a school. The report then compared these concentrations to available health-based benchmarks. This evaluation concluded that potential indoor air concentrations were below levels of concern. Because this evaluation specifically addressed releases from FGD gypsum wallboard, the results are directly applicable to the current evaluation. However, the relatively small number of FGD gypsum samples evaluated introduces uncertainty as to whether the mercury emanation rates measured adequately characterize the range of potential emanation rates from fly ash concrete. Therefore, the current evaluation retained mercury as a COPC for emanation to air.

[8] The evaluation focused on organics, such as n-hexane and formaldehyde, which are associated with adhesives and additives added to wallboard during production, rather than with FGD gypsum.

Wollenburg and Smith (1962): *Portland Cement for a Low-Counting Facility*

Lindeken and Coles (1977): *The Radium-226 Content of Agricultural Gypsums*

Zikovsky and Kennedy (1992): *Radioactivity of Building Materials Available in Canada*

Henkels and Gaynor (1996): *Characterizing Synthetic Gypsum for Wallboard Manufacture*

Roper et al (2013): *Analysis of Naturally-Occurring Radionuclides in Coal Combustion Fly Ash, Gypsum, and Scrubber Residue Samples*

A number of existing evaluations were identified that address radiation from FGD gypsum and FGD gypsum wallboard from a broad range of domestic and international sources. Although some of these evaluations do not directly address FGD gypsum wallboard, the results are applicable to the current evaluation because the beneficial use of FGD gypsum in wallboard is unlikely to appreciably alter radionuclide concentrations in the raw material. These evaluations include peer-reviewed publications, guidance documents, and voluntary standards that have been developed and reviewed by experts in the field of radiation and health physics. This body of work extends back nearly forty years, and has already been well summarized through the literature reviews contained in more recent existing evaluations. Therefore, only these key recent evaluations, which form the basis for the conclusions in this beneficial use evaluation, are summarized in the following text.

A number of national and international organizations develop and maintain guidance for managing public exposures to everyday sources of radiation, such as building materials. Evaluations of radiation exposure are often based on the recommendations of these guidances. The relevant organizations are discussed in **Section 1.1.1**.

- Roper et al. (2013) measured the activity of radionuclides in the uranium and thorium decay series, as well as potassium-40, from samples of FGD gypsum collected across the United States. The study found that FGD gypsum activities fell within the range of activities previously reported from extensive sampling of European mined gypsum. Consideration of additional activity data reported in the literature for North American mined gypsum and wallboard does not result in different conclusions (Wollenberg and Smith, 1962; Lindeken and Coles, 1977; Zikovsky and Kennedy; 1992).

The existing evaluations identified demonstrate that the potential exposures to gamma radiation from FGD gypsum wallboard are comparable to those from mined gypsum wallboard. No existing evaluations were identified that evaluated radon emanation from FGD or mined gypsum. However, because the radionuclide concentrations in raw FGD and mined gypsum are comparable; the physical structures of the finished products are nearly identical (Henkels and Gaynor, 1996); and the finished products are used in the same way, this evaluation concludes that the rate of radon emanation also will be comparable. Therefore, the current evaluation eliminated radionuclides from further consideration.

1.2.2 Data Collection for FGD Gypsum Wallboard

The review of existing evaluations discussed in **Section 1.2.1** identified one potential release and a single associated COPC that the existing evaluations had not sufficiently addressed. Therefore, the current evaluation assembled data on this remaining release and the associated COPC from the existing

evaluations and other available literature. These data form the basis for the evaluation of COPC releases from FGD gypsum wallboard conducted in subsequent steps of this evaluation. The remainder of this section enumerates the major sources from which these data were drawn. **Appendix A** provides a further discussion of each data source, along with a presentation of the corresponding raw data.

- **Electric Power Research Institute (2010):** *Public Comments to Hazardous and Solid Waste Management System; Identification and Listing of Special Wastes; Disposal of Coal Combustion Residuals from Electric Utilities.* This literature source contains data on mercury concentrations in FGD gypsum and mined gypsum.
- **Gypsum Association (2010):** *Public Comments to Hazardous and Solid Waste Management System; Identification and Listing of Special Wastes; Disposal of Coal Combustion Residuals from Electric Utilities.* This literature source contains data on mercury concentrations in FGD gypsum and mined gypsum.
- **Kairies et al. (2006):** *Mercury in Gypsum Produced from Flue Gas Desulfurization.* This literature source contains data on mercury concentrations in FGD gypsum and FGD gypsum wallboard.
- **Shock et al. (2009):** *Evaluation of Potential for Mercury Volatilization from Natural and FGD Gypsum Products Using Flux-Chamber Tests.* This literature source contains data on mercury concentrations in and mercury emanation rates from FGD gypsum and mined gypsum wallboards.
- **US EPA (2009a):** *Characterization of Coal Combustion Residues from Electric Utilities – Leaching and Characterization Data.* This literature source contains data on mercury concentrations in FGD gypsum.
- **United States Department of Energy (DOE) (2008):** *Fate of Mercury in Synthetic Gypsum Used for Wallboard Production.* This literature source contains data on mercury concentrations in FGD gypsum, mined gypsum, and FGD gypsum wallboard.
- **Yost et al. (2010):** *Lack of Complete Exposure Pathways for Metals in Natural and FGD Gypsum.* This literature source contains data on mercury concentrations in FGD gypsum and mined gypsum.

1.2.3 Summary of Releases Identified for FGD Gypsum Wallboard

Based on the review of the available literature, the current evaluation initially identified four potential releases from FGD gypsum wallboard that may occur during use: 1) generation of dust, 2) emanation to air, 3) leaching to ground and surface water, and 4) decay of naturally occurring radionuclides. A review of existing evaluations found them to be of sufficient quality and applicability to eliminate all releases from further consideration except for emanation to air. The one COPC identified for this release was mercury.

1.3 Conclusions of Step 1

Based on a review of the available literature, the current evaluation identified COPCs that may be released from fly ash concrete and FGD gypsum wallboard, but have not been sufficiently addressed by existing evaluations. For fly ash concrete, potential releases retained for further consideration were those to dust, ground and surface water, and air. COPCs for dust include aluminum, antimony, arsenic,

barium, beryllium, boron, cadmium, chromium, cobalt, copper, iron, lead, mercury, manganese, molybdenum, nickel, selenium, silver, strontium, thallium, uranium, vanadium, and zinc. COPCs for ground and surface water include antimony, arsenic, boron, cadmium, chromium, lead, molybdenum, selenium, and thallium. The one COPC identified for emanation to air was mercury. For FGD gypsum wallboard, the single potential release retained was to air. The one COPC identified for this release was mercury. **Table 1-2** provides a summary of the releases and associated COPCs.

Table 1-2: List of COPCs Remaining Following Step 1

COPC	Fly Ash Concrete			FGD Gypsum Wallboard
	Dust	Ground and Surface Water	Air	Air
Aluminum	X	--	--	--
Antimony	X	X	--	--
Arsenic	X	X	--	--
Barium	X	--	--	--
Beryllium	X	--	--	--
Boron	X	X	--	--
Cadmium	X	X	--	--
Chromium	X	X	--	--
Cobalt	X	--	--	--
Copper	X	--	--	--
Iron	X	--	--	--
Lead	X	X	--	--
Manganese	X	--	--	--
Mercury	X	--	X	X
Molybdenum	X	X	--	--
Nickel	X	--	--	--
Selenium	X	X	--	--
Silver	X	--	--	--
Strontium	X	--	--	--
Thallium	X	X	--	--
Uranium	X	--	--	--
Vanadium	X	--	--	--
Zinc	X	--	--	--

COPC = constituent of potential concern
FGD = flue gas desulfurization
X = constituent requiring further evaluation
-- = constituent requiring no further evaluation

Step 2: Comparison of Available Data

This section applies the second step of the methodology to the current beneficial use evaluation of fly ash concrete and FGD gypsum wallboard, focused on the COPCs identified in Step 1 (Literature Review and Data Collection). This evaluation aggregated and used data from the literature to compare the range of potential COPC concentrations and determine whether releases from fly ash concrete and FGD gypsum wallboard are comparable to or lower than those from analogous products. This section details the comparisons conducted, the assumptions built into these comparisons, and the results. **Appendix A** presents the raw data used in these comparisons.

2.1 Releases from Fly Ash Concrete and Portland Cement Concrete

This subsection presents the comparisons of potential COPC releases from fly ash concrete and the analogous product, portland cement concrete, during use. The type and amount of data available determined the types of comparisons conducted. These comparisons considered all available lines of evidence to determine whether releases of COPCs from fly ash concrete are comparable to or lower than those from portland cement concrete. The evaluation retained COPCs with the potential to be released from fly ash concrete at rates that are higher than those from portland cement concrete, or for which portland cement concrete data are not available, for further consideration in subsequent steps of the evaluation.

2.1.1 Generation of Dust

Dust is generated during use when disturbances to the concrete matrix results in the transport of particulate matter away from the encapsulated matrix. The COPC concentrations in these releases will be similar to those in the source concrete because they are both composed of the same materials. The current evaluation did not identify sufficient data on the range of COPC concentrations in finished fly ash concrete and portland cement concrete to compare these products directly. Instead, the evaluation used COPC concentrations in raw fly ash and portland cement as a surrogate in this comparison. A surrogate is defined in *Methodology for Evaluating Encapsulated Beneficial Uses of Coal Combustion Residuals* as "data on one variable that can be used to reliably approximate the behavior of another variable and, as a result, can substitute for that variable in the comparison" (US EPA, 2013a). Because the substitution of portland cement with fly ash is the primary difference in the compositions of these two types of concrete, any difference in COPC concentrations in these raw materials will drive differences in the resulting concrete dust. The evaluation drew fly ash data from the CCR Constituent Database and portland cement data from three sources (PCA, 1992; Pflughoeft-Hassett et al., 1993; and Eckert and Guo, 1998). This evaluation used a statistical comparison to determine whether differences in the two data sets were significant.

This evaluation used ProUCL Version 4.1.01 to compare the distribution of different COPC concentrations found in fly ash and portland cement (US EPA, 2010b,c). This statistical software allows for appropriate consideration of datasets with non-detect values. To help select the most appropriate

statistical test to use in the comparison, this evaluation used both the Shapiro-Wilk and Lilliefors Goodness-of-Fit tests to determine the general shape of the different COPC data distributions. This shape is important because it provides information on which statistical descriptors (e.g., standard deviation) are appropriate to use when comparing the datasets. Visual inspection of the distributions shows that most are highly skewed to the right and, based on a user-specified confidence level of 95 percent, both the Shapiro-Wilk and Lilliefors Goodness-of-Fit tests indicate that none of the COPC datasets fit a normal, lognormal, or gamma distribution. Because the datasets did not fit one of these established probability distributions, this evaluation applied a non-parametric hypothesis test that does not assign a distribution to the datasets.

This evaluation found the Gehan test to be the most appropriate non-parametric hypothesis test because the datasets for some COPCs contain large numbers of non-detects with varying detection limits. The Gehan test was applied to each dataset with a null hypothesis (H_0) that the median COPC concentration in portland cement is greater than or equal to that in fly ash, and the alternative hypothesis (H_a) that the median COPC concentration in portland cement is less than that in fly ash. Thus, the result of the hypothesis test is a p-value. This statistical value is a measurement of confidence in H_0. The p-value is compared to a user-specified confidence level representing the acceptable likelihood of incorrectly rejecting H_0. When the p-value is lower than the specified confidence level, H_0 is rejected in favor of H_a. This evaluation applied a confidence level of 95 percent ($\alpha = 0.05$). The evaluation retained COPCs with a p-value of less than 0.05 for further consideration. **Table 2-1** provides the results of these comparisons. Because some of the p-values are extremely small, the table truncates p-values beyond 0.0001 for ease of presentation.

Table 2-1: Comparison of COPC Concentrations in Fly Ash and Portland Cement (mg/kg)

Constituent	Fly Ash		Portland Cement		p-value
	Detection Frequency	Median	Detection Frequency	Median	
Aluminum	22 / 22	87,833	11 / 11	26,250	< 0.0001
Antimony	38 / 42	6.2	2 / 94	0.10	< 0.0001
Arsenic	97 / 100	50.1	64 / 109	12.4	< 0.0001
Barium	61 / 61	1,189	98 / 98	205	< 0.0001
Beryllium	24 / 32	10.5	93 / 94	0.98	< 0.0001
Boron	34 / 34	403	3 / 4	42.5	< 0.0001
Cadmium	70 / 91	1.3	42 / 98	0.03	< 0.0001
Chromium	91 / 91	107	109 / 109	58.6	< 0.0001
Cobalt	49 / 49	45.3	12 / 15	10.0	< 0.0001
Copper	47 / 48	108	15 / 15	36.0	< 0.0001
Iron	23 / 23	27,514	0 / 0	N/A	N/A
Lead	98 / 99	55.0	88 / 109	6.3	< 0.0001
Manganese	48 / 48	219	15 /15	465	0.99

mg/kg = milligrams per kilogram
N/A = Not applicable; available data not sufficient to conduct comparison

Table 2-1: Comparison of COPC Concentrations in Fly Ash and Portland Cement (mg/kg)

Constituent	Fly Ash		Portland Cement		p-value
	Detection Frequency	Median	Detection Frequency	Median	
Mercury	73 / 87	0.17	25 / 98	0.01	< 0.0001
Molybdenum	44 / 46	16.3	1 / 4	5.0	< 0.0001
Nickel	75 / 76	76.7	97 / 109	25.0	< 0.0001
Selenium	71 / 79	8.8	6 / 98	2.0	< 0.0001
Silver	16 / 26	0.55	93 / 98	8.6	1.00
Strontium	22 / 22	795	0 / 0	N/A	N/A
Thallium	19 / 27	2.1	32 / 94	0.30	< 0.0001
Uranium	10 / 19	5.5	0 / 0	N/A	N/A
Vanadium	43 / 43	267	15 / 15	64.0	< 0.0001
Zinc	51 / 51	141	14 / 15	64.0	0.04

mg/kg = milligrams per kilogram
N/A = Not applicable; available data not sufficient to conduct comparison

Based on the statistical evaluation, the median concentrations of aluminum, antimony, arsenic, barium, beryllium, boron, cadmium, chromium, cobalt, copper, lead, mercury, molybdenum, nickel, selenium, thallium, vanadium, and zinc were found to be higher in fly ash than in portland cement. Therefore, the evaluation retained each of these COPCs for further consideration. Iron, strontium, and uranium were also retained because no portland cement data were available for comparison. Concentrations of manganese and silver were found to be less than or equal to those in portland cement. As a result, the evaluation did not carry either of these COPCs forward for further consideration.

2.1.2 Leaching to Ground and Surface Water

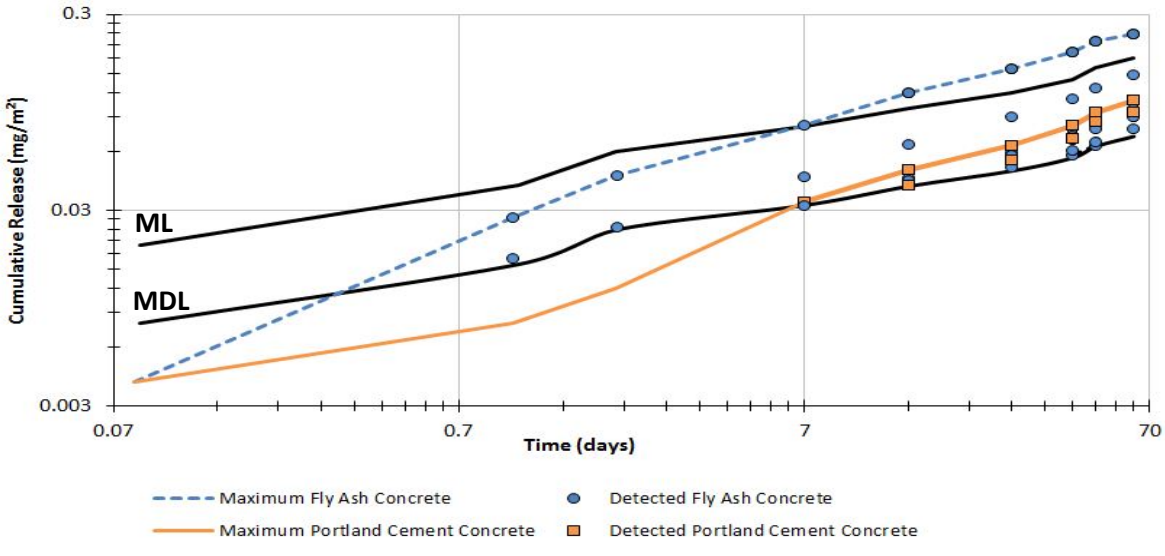
Leaching occurs during use when COPCs diffuse out of the concrete matrix and into surrounding liquids. The current evaluation applied the data from Garrabrants et al. (2013) as the most relevant to this release pathway. Garrabrants et al. (2013) used the EPA Leaching Evaluation Assessment Framework (LEAF) Method 1315 to estimate the cumulative release of COPCs from monolithic concrete blocks that had been allowed to cure for three months prior to sampling. Because these data provide information on releases of COPCs as a function of time, they allow a more precise evaluation of concrete leaching behavior. The available data consist of three sets of concrete samples. The first and second sets were concrete with a 20 percent and 45 percent fly ash replacement, respectively. The third set was micro-concrete with a 45 percent fly ash replacement.[9] For each of these sample sets, the study also collected a single control sample of portland cement concrete. The current evaluation includes a direct comparison of these data because the type and amount of available data do not support a robust statistical comparison.

[9] Micro-concretes have a similar composition as standard concrete, but lack large aggregate. The micro-concrete mixture is intended to mimic the rheological properties of standard concrete. These samples were included in Kosson et al. (2013) and Garrabrants et al. (2013) to evaluate their use as a surrogate for standard concrete in leaching studies.

The following figures compare the data drawn from Garrabrants et al. (2013) on the cumulative constituent mass released per unit of surface area (mg/m^2) from fly ash concrete and portland cement concrete. Garrabrants et al. (2013) examined the cumulative release of each COPC over time using EPA LEAF Method 1315. Each value listed in the following graphs represents the COPC concentration measured at a discrete time step added to the concentration measured at the previous time step. In each graph, the leachate data are plotted along with the associated method detection limit (MDL) and minimum level of quantitation (ML). The MDL is the minimum concentration that can reliably be differentiated from background noise, while the ML is the minimum concentration that can be quantified with accuracy. Following the recommendations in *Risk Assessment Guidance for Superfund (RAGS) Part A* (US EPA, 1989) and *EPA Region 3 Guidance on Handling Chemical Concentration Data near the Detection Limit in Risk Assessments* (US EPA, 1991), the current evaluation added samples that were not detected above the MDL to the subsequent time step at half the MDL. In addition, this evaluation considered samples detected above the MDL, but below the ML, to be estimated values and added those samples to the subsequent time step at the reported value. While the graphs below show an increasing MDL and ML, these values are constant for each individual time step. The summation of each discrete sample results in cumulative MDL and ML curves.

Figure 2-1 provides a comparison of antimony leaching from fly ash concrete and portland cement concrete. The amount of antimony released from two fly ash concrete samples was consistently higher than all portland cement concrete samples. The higher leaching profiles of these samples indicate the potential for fly ash concrete to leach antimony at higher rates than those of portland cement concrete.

Concrete Type	Number of Samples	Maximum Cumulative Release (mg/m^2)								
		0.08 Day	1 Day	2 Day	7 Day	14 Day	28 Day	42 Day	49 Day	63 Day
Fly Ash Concrete	10	*< 0.01*	0.03	0.05	0.08	0.12	0.16	0.19	0.22	0.24
Portland Cement Concrete	3	*< 0.01*	*< 0.016*	*< 0.024*	0.03	0.05	0.07	0.08	0.1	0.11



< Denotes a "non-detect" observation (below MDL); value estimated as half of MDL

Figure 2-1: Comparison of cumulative leaching of antimony from concrete.

Figure 2-2 presents a comparison of arsenic leaching from fly ash concrete and portland cement concrete. All samples were below the MDL for every measured time step. Therefore, the current evaluation could not identify any differences between fly ash concrete and portland cement concrete with respect to arsenic leaching.

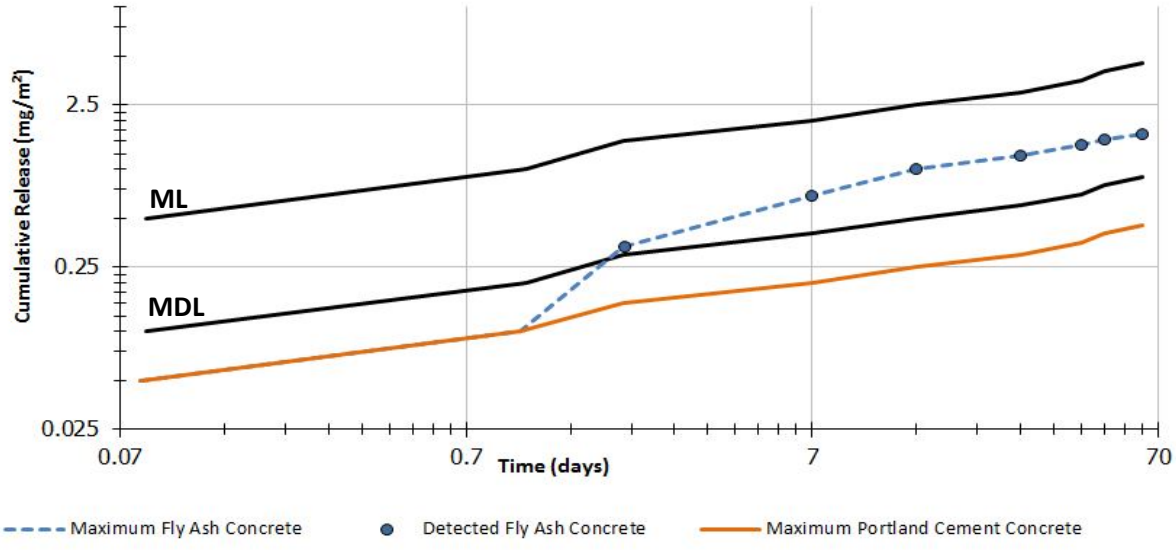
Concrete Type	Number of Samples	Maximum Cumulative Release (mg/m^2)								
		0.08 Day	1 Day	2 Day	7 Day	14 Day	28 Day	42 Day	49 Day	63 Day
Fly Ash Concrete	10	< 0.06	< 0.13	< 0.19	< 0.26	< 0.32	< 0.38	< 0.45	< 0.51	< 0.58
Portland Cement Concrete	3	< 0.06	< 0.13	< 0.19	< 0.26	< 0.32	< 0.38	< 0.45	< 0.51	< 0.58

Note: All measured samples are "non-detect" (below MDL); no graph provided

Figure 2-2: Comparison of cumulative leaching of arsenic from concrete.

Figure 2-3 presents a comparison of boron leaching from fly ash concrete and portland cement concrete. Boron was detected in one sample of fly ash concrete, but was not detected in any samples of portland cement concrete. The higher leaching profile of this one fly ash concrete sample indicates a potential for fly ash concrete to leach boron at higher rates than portland cement concrete.

Concrete Type	Number of Samples	Maximum Cumulative Release (mg/m^2)								
		0.08 Day	1 Day	2 Day	7 Day	14 Day	28 Day	42 Day	49 Day	63 Day
Fly Ash Concrete	10	< 0.10	< 0.20	0.33	0.69	1.01	1.22	1.42	1.54	1.64
Portland Cement Concrete	3	< 0.10	< 0.20	< 0.30	< 0.40	< 0.50	< 0.60	< 0.70	< 0.80	< 0.90



< Denotes a "non-detect" observation (below MDL); value estimated as half of MDL

Figure 2-3: Comparison of cumulative leaching of boron from concrete.

Figure 2-4 presents a comparison of cadmium leaching from fly ash concrete and portland cement concrete. All samples were below the MDL for every measured time step. Therefore, the current evaluation could not identify any differences between fly ash concrete and portland cement concrete with respect to cadmium leaching.

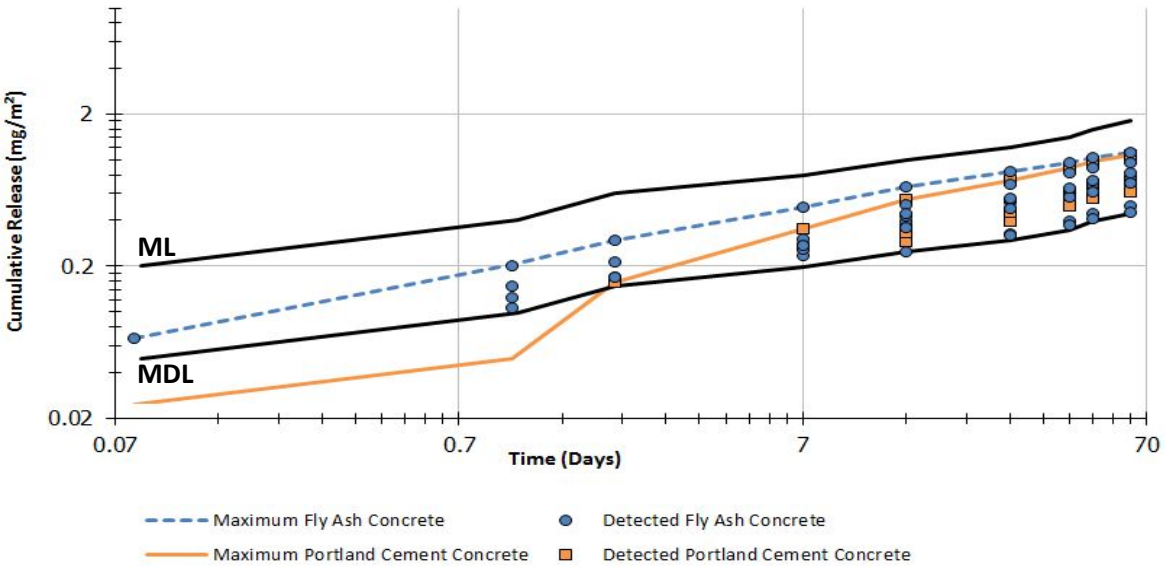
Concrete Type	Number of Samples	Maximum Cumulative Release (mg/m^2)								
		0.08 Day	1 Day	2 Day	7 Day	14 Day	28 Day	42 Day	49 Day	63 Day
Fly Ash Concrete	10	< 0.02	< 0.03	< 0.05	< 0.07	< 0.09	< 0.10	< 0.12	< 0.14	< 0.15
Portland Cement Concrete	3	< 0.02	< 0.03	< 0.05	< 0.07	< 0.09	< 0.10	< 0.12	< 0.14	< 0.15

Note: All measured samples are "non-detect" (below MDL); no graph provided

Figure 2-4: Comparison of cumulative leaching of cadmium from concrete.

Figure 2-5 presents a comparison of cumulative leaching of chromium from fly ash concrete and portland cement concrete. Several samples of fly ash concrete exhibited higher leaching rates during the first several time steps. As time progressed, releases from some portland cement concrete approach, but never exceed, those from fly ash concrete. The higher leaching profiles of these samples indicate the potential for fly ash concrete to leach chromium at higher rates than from portland cement concrete.

Concrete Type	Number of Samples	Maximum Cumulative Release (mg/m^2)								
		0.08 Day	1 Day	2 Day	7 Day	14 Day	28 Day	42 Day	49 Day	63 Day
Fly Ash Concrete	10	0.07	0.20	0.30	0.49	0.66	0.84	0.96	1.05	1.12
Portland Cement Concrete	3	< 0.05	< 0.10	0.16	0.35	0.55	0.74	0.89	0.98	1.09



< Denotes a "non-detect" observation (below MDL); value estimated as half of MDL

Figure 2-5: Comparison of cumulative leaching of chromium from concrete.

Figure 2-6 presents a comparison of lead leaching from fly ash concrete and portland cement concrete. All samples were below the MDL for every measured time step. Therefore, the current evaluation could not identify any differences between fly ash concrete and portland cement concrete with respect to lead leaching.

Concrete Type	Number of Samples	Maximum Cumulative Release (mg/m^2)								
		0.08 Day	1 Day	2 Day	7 Day	14 Day	28 Day	42 Day	49 Day	63 Day
Fly Ash Concrete	10	< 0.02	< 0.05	< 0.07	< 0.09	< 0.11	< 0.14	< 0.16	< 0.18	< 0.21
Portland Cement Concrete	3	< 0.02	< 0.05	< 0.07	< 0.09	< 0.11	< 0.14	< 0.16	< 0.18	< 0.21

Note: All measured samples are "non-detect" (below MDL); no graph provided

Figure 2-6: Comparison of cumulative leaching of lead from concrete.

Figure 2-7 presents a comparison of molybdenum leaching from fly ash concrete and portland cement concrete. All samples were below the MDL for every measured time step. Therefore, the current evaluation could not identify any differences between fly ash concrete and portland cement concrete with respect to molybdenum leaching.

Concrete Type	Number of Samples	Maximum Cumulative Release (mg/m^2)								
		0.08 Day	1 Day	2 Day	7 Day	14 Day	28 Day	42 Day	49 Day	63 Day
Fly Ash Concrete	10	< 0.08	< 0.15	< 0.23	< 0.30	< 0.38	< 0.46	< 0.53	< 0.61	< 0.68
Portland Cement Concrete	3	< 0.08	< 0.15	< 0.23	< 0.30	< 0.38	< 0.46	< 0.53	< 0.61	< 0.68

Note: All measured samples are "non-detect" (below MDL); no graph provided

Figure 2-7: Comparison of cumulative leaching of molybdenum from concrete.

Figure 2-8 presents a comparison of selenium leaching from fly ash concrete and portland cement concrete. All samples were below the MDL for every measured time step. Therefore, the current evaluation could not identify any differences between fly ash concrete and portland cement concrete with respect to selenium leaching.

Concrete Type	Number of Samples	Maximum Cumulative Release (mg/m^2)								
		0.08 Day	1 Day	2 Day	7 Day	14 Day	28 Day	42 Day	49 Day	63 Day
Fly Ash Concrete	10	< 0.05	< 0.10	< 0.16	< 0.21	< 0.26	< 0.31	< 0.36	< 0.42	< 0.47
Portland Cement Concrete	3	< 0.05	< 0.10	< 0.16	< 0.21	< 0.26	< 0.31	< 0.36	< 0.42	< 0.47

Note: All measured samples are "non-detect" (below MDL); no graph provided

Figure 2-8: Comparison of cumulative leaching of selenium from concrete.

Figure 2-9 presents a comparison of thallium leaching from fly ash concrete and portland cement concrete. All samples were below the MDL for every measured time step. Therefore, the current evaluation could not identify any differences between fly ash concrete and portland cement concrete with respect to thallium leaching.

Concrete Type	Number of Samples	Maximum Cumulative Release (mg/m^2)								
		0.08 Day	1 Day	2 Day	7 Day	14 Day	28 Day	42 Day	49 Day	63 Day
Fly Ash Concrete	10	< 0.05	< 0.10	< 0.15	< 0.20	< 0.25	< 0.30	< 0.35	< 0.41	< 0.46
Portland Cement Concrete	3	< 0.05	< 0.10	< 0.15	< 0.20	< 0.25	< 0.30	< 0.35	< 0.41	< 0.46

Note: All measured samples are "non-detect" (below MDL); no graph provided

Figure 2-9: Comparison of cumulative leaching of thallium from concrete.

In summary, the comparison of cumulative leaching data presented in Garrabrants et al. (2013) indicates that antimony, boron, and chromium have the potential to leach at higher rates from fly ash concrete than from portland cement concrete. Because these three constituents demonstrated the potential to leach at higher rates from fly ash concrete, the current evaluation retained them as COPCs for further consideration. Concentrations of arsenic, cadmium, lead, molybdenum, selenium, and thallium were below the MDL in all the measured concrete samples that were allowed to cure for three months. With the exception of selenium, these six constituents exhibited similar leaching behavior in an additional, unpublished set of samples collected by the authors of Kosson et al. (2013) and Garrabrants et al. (2013). These additional samples were identical in composition to those reported in Kosson et al. (2013) and Garrabrants et al. (2013), but were allowed to cure for only 28 days. Concentrations of arsenic, cadmium, lead, molybdenum, and thallium were all below the MDL in samples tested using Method 1315, but selenium was detected slightly above the MDL in two samples of fly ash concrete (Kosson and Garrabrants, 2012). Because selenium demonstrated the potential to leach at higher rates from fly ash concrete, the current evaluation retained it as a COPC for further consideration.

As previously noted, arsenic, cadmium, lead, molybdenum, and thallium were below the MDL in all measured concrete samples. At the most extreme, these samples represent micro-concretes that were cured for only 28 days and placed in contact with water for 14 consecutive days. Kosson et al. (2013) and Garrabrants (2013) found that while micro-concretes provide a good approximation of concrete leaching behavior, these samples generally leach at higher concentrations and rates than standard concrete. As demonstrated by these two studies, concrete leaching decreases as the concrete cures. Therefore, long-term concrete leaching behavior will be lower than measured after only 28 days. Furthermore, a 14-day contact time is much longer than the amount of time most water will remain in contact with a concrete surface, and far longer than required by other available leaching tests (e.g., TCLP and SPLP), which call for contact times of less than a day. Thus, there is no evidence that releases of these five COPCs from fly ash concrete are substantially different than those from portland cement concrete. Therefore, the evaluation did not carry leaching of arsenic, cadmium, lead, molybdenum, and thallium forward for further consideration.

2.1.3 Emanation to Indoor Air

Releases can occur during use when gases or vapors diffuse through the concrete matrix and migrate into indoor air. The single COPC identified for this type of release was mercury. The current evaluation conducted the comparison of mercury emanation rates from fly ash concrete and portland cement concrete using data from two studies (Golightly et al., 2005; 2009). Because of the limited sample data available, only a comparison of the range of emanation rates was possible. **Table 2-2** presents the results of this comparison.

Table 2-2: Comparison of Mercury Emanation Rates from Concretes (ng/m^2-hr)

Type of Sample	Sample Number	28-Day Range	Sample Number	56-Day Range
Portland Cement Concrete	2	2.8 - 4.4	1	3.5
33 Percent Fly Ash Concrete	1	11.4	0	--
55 Percent Fly Ash Concrete*	3	14.9 - 34.9	2	4.4 - 15.6

ng/m^2-hr = nanograms per meter squared – hour

-- = no data available

* = while a 55 percent fly ash replacement rate falls outside the scope of the current evaluation, this data was retained in the evaluation to capture a conservative upper bound of potential releases

This comparison shows that, during the first 28 days of curing, average mercury emanation rates from portland cement concrete may range between 2.8 and 4.4 ng/m^2-hr. These rates are lower than the 11.4 ng/m^2-hr measured for 33 percent fly ash concrete; they are also below the range of 14.9 to 34.9 ng/m^2-hr measured for 55 percent fly ash concrete. At around 56 days of curing, the reported emanation rate from portland cement concrete of 3.5 ng/m^2-hr was within the range of rates measured after 28 days. The range of emanation rates from 55 percent fly ash concrete decreased considerably to between 4.4 and 15.6 ng/m^2-hr after 56 days of curing, but remained higher than the portland cement concrete emanation rate. No data were available for 33 percent fly ash concrete at 56 days of curing. These results indicate that fly ash concrete may emit mercury at higher rates during the first 56 days than portland cement concrete. Therefore, this evaluation retained mercury as a COPC for further consideration.

2.2 Releases from FGD Gypsum Wallboard and Mined Gypsum Wallboard

This subsection presents comparisons of potential releases from FGD gypsum wallboard and the analogous product, mined gypsum wallboard, during use. The type and amount of data available determined the type of comparison conducted. These comparisons considered all available lines of evidence to determine whether releases from FGD gypsum wallboard are comparable to or lower than those from mined gypsum wallboard. The evaluation retained COPCs with the potential to be released from FGD gypsum wallboard at rates that are higher than from mined gypsum wallboard for further consideration in subsequent steps of the evaluation.

2.2.1 Emanation to Indoor Air

Releases can occur during use when gases or vapors diffuse through wallboard and migrate into indoor air. The single COPC identified for this type of release was mercury. The evaluation conducted the comparison of mercury emanation rates from FGD gypsum wallboard and mined gypsum wallboard using the data from one study (Shock et al., 2009). Because of the limited sample data available, this evaluation only compared the range of emanation rates. **Table 2-3** presents the results of this comparison.

Table 2-3: Comparison of Mercury Emanation Rates from Wallboards (ng/m^2-hr)

Type of Sample	Number of Samples	Low	Moderate	High
Mined Gypsum Wallboard	3	0.030	0.039	0.043
FGD Gypsum Wallboard	3	0.140	0.281	0.341

FGD = flue gas desulfurization
ng/m^2-hr = nanograms per meter squared – hour

The results indicate that mercury emanation rates from FGD gypsum wallboard, which ranged between 0.14 and 0.34 ng/m^2-hr, are higher than emanation rates from mined gypsum wallboard, which ranged between 0.03 and 0.04 ng/m^2-hr. The lowest mercury emanation rate measured for FGD gypsum wallboard is nearly three times higher than the highest measured emanation rate from mined gypsum wallboard, indicating that FGD gypsum wallboard may emit mercury at higher rates than mined gypsum wallboard. Therefore, the evaluation retained mercury as a COPC for further consideration.

2.3 Conclusions of Step 2

Based on the results of the comparisons conducted in this step, the evaluation found that releases of several COPCs were comparable to or lower than those from analogous products. For the generation of dust from fly ash concrete, the evaluation eliminated manganese and silver from further consideration. For leaching from fly ash concrete, the evaluation eliminated arsenic, cadmium, lead, molybdenum, and thallium from further consideration. For releases to the air, the evaluation could not eliminate mercury for either fly ash concrete or FGD gypsum wallboard. **Table 2-4** provides a list of the remaining COPCs following this step of the evaluation.

Table 2-4: List of COPCs Remaining Following Step 2

COPC	Fly Ash Concrete			FGD Gypsum Wallboard
	Dust	Ground and Surface Water	Air	Air
Aluminum	X	--	--	--
Antimony	X	X	--	--
Arsenic	X	--	--	--
Barium	X	--	--	--
Beryllium	X	--	--	--
Boron	X	X	--	--
Cadmium	X	--	--	--
Chromium	X	X	--	--
Cobalt	X	--	--	--
Copper	X	--	--	--
Iron	X	--	--	--
Lead	X	--	--	--
Manganese	--	--	--	--
Mercury	X	--	X	X
Molybdenum	X	--	--	--
Nickel	X	--	--	--
Selenium	X	X	--	--
Silver	--	--	--	--
Strontium	X	--	--	--
Thallium	X	--	--	--
Uranium	X	--	--	--
Vanadium	X	--	--	--
Zinc	X	--	--	--

COPC = constituent of potential concern
FGD = flue gas desulfurization
X = constituent requiring further evaluation
-- = constituent requiring no further evaluation

Step 3: Exposure Review

The purpose of this section is to apply the third step of the encapsulated beneficial use methodology to the current evaluation of fly ash concrete and FGD gypsum wallboard, based on the COPCs carried forward from Step 2 (Comparison of Available Data). This section identifies the high-end chronic exposure pathways through which highly exposed individuals (HEIs) may potentially come in contact with COPCs in each identified type of release. This evaluation used this information to identify the complete exposure pathways requiring further evaluation and to develop a conceptual exposure model.

3.1 Fly Ash Concrete

This subsection describes the potential exposure pathways and receptors for each COPC that may be released from fly ash concrete during use. All of the COPCs identified in this evaluation are inorganic metals. While these constituents may change valence states, form complexes with other ions and compounds, or undergo other reactions that reduce their mobility or bioavailability, they will never naturally degrade. Once released into the environment, these constituents will persist indefinitely. Therefore, when a release occurs, exposures are theoretically possible.

3.1.1 Potential Exposure Pathways for Fly Ash Concrete

Exposure to Dust

Because of the high strength of finished concrete, releases of dust from solid concrete are generally negligible during use. However, some high abrasion environments may result in the generation of nontrivial amounts of concrete dust. The current evaluation identified the most relevant pathway as roadways exposed to traffic with studded tires. Any of the COPCs present in the concrete may be present in the resulting dust.

Ingestion of generated dust that has settled on various surfaces may occur through incidental hand to mouth contact. Wind or overland runoff may carry the dust from roadways to downgradient soils. This dust has the potential to accumulate in the surface soil over time. Based on these findings, this evaluation retained ingestion of concrete dust as an exposure pathway of potential concern for both human and ecological receptors.

Inhalation of dust may occur when wind or physical disturbances suspend the dust in air. However, studded tires are seasonal and limited to winter months in the states that permit their use. Over the course of seconds to hours, suspended dust will either disperse in the air or settle out on the ground. Significant wearing of the roadway would need to occur to generate the amount of dust necessary to sustain chronic, elevated dust concentrations in the air. A review of the air quality around Alaskan roads found that, even in the presence of traffic with studded tires, it was natural sources (e.g., wildfires) or vehicle exhaust that were the primary drivers of National Ambient Air Quality Standards exceedances for fine particulates (Zubeck et al., 2004). In addition, most roadway dust identified by this study originated from dirt and asphalt roads, rather than roads made with concrete. Because inhalation of

concrete dust is not a pathway that may drive exposures, this evaluation did not consider it in Step 4 (Screening Assessment).

Dermal contact may occur through direct contact with concrete dust that has settled on various surfaces. However, absorption through the skin is limited compared to other exposure pathways because of the relatively low lipid solubility of most metals (Paustenbach, 2000 and Hostynek et al., 1998 as cited in US EPA, 2007a). Because dermal contact with concrete dust is not a pathway that may drive exposures, this evaluation did not consider it in Step 4 (Screening Assessment).

Exposure to Ground Water

Some fraction of the precipitation that falls on concrete may infiltrate directly through any cracks present in the concrete matrix and into underlying soil. The remaining fraction of the precipitation that does not infiltrate directly through the concrete matrix may run off and infiltrate through adjacent soil. Any of the COPCs identified may be released into water that comes into contact with the concrete. All COPCs dissolved are assumed to migrate vertically through the soil column and enter the ground water table.

Ingestion of COPCs present in concrete leachate may occur if ground water is used as a source of potable water for human receptors. Leaching can occur during each precipitation event and can result in a ground water plume that contaminates downgradient private wells for multiple years. Therefore, this evaluation retained ingestion of ground water impacted by concrete leachate as an exposure pathway of potential concern for human receptors. Ecological receptors are not anticipated to have any appreciable direct contact with ground water.

None of the COPCs identified for ground water are volatile under standard environmental conditions. Therefore, inhalation of COPCs from ground water used as a source of drinking water is not a complete exposure pathway and this evaluation did not consider it in Step 4 (Screening Assessment).

Dermal contact may occur through direct contact with ground water while bathing. However, absorption through the skin is limited compared to other exposure pathways because of the relatively low lipid solubility of most metals (Paustenbach, 2000 and Hostynek et al., 1998 as cited in US EPA, 2007a). Because dermal contact with ground water is not a pathway that may drive exposures, this evaluation did not consider it in Step 4 (Screening Assessment).

Exposure to Surface Water

When precipitation falls on concrete, some fraction of this precipitation may run off overland and into downgradient water bodies. This overland runoff may also infiltrate into underlying ground water before discharging to downgradient water bodies. Some concrete structures, such as bridges and dams, may have frequent direct contact with water bodies and leach directly into surface water. Any of the COPCs identified may be released into water that comes into contact with the concrete.

Surface water used as a source of potable water is assumed to be a negligible exposure pathway for human receptors. Surface water is assumed to be routed through a municipal water treatment facility prior to consumption, reducing the levels of any COPCs present. Incidental ingestion of COPCs in surface water may occur during swimming or other activities near a water body. For human receptors, it

is assumed that these exposures are infrequent and small in comparison to exposures from intentional ingestion of ground water. However, ingestion may be a significant exposure pathway for ecological receptors that live in and around the water body. Therefore, the evaluation retained the ingestion of and direct contact with surface water as an exposure pathway of potential concern for ecological receptors. In turn, some of the fish present in these water bodies can represent a sizable portion of the diet for some human receptors. Therefore, the evaluation retained ingestion of fish as an exposure pathway of potential concern for human receptors.

As discussed for ground water, none of the COPCs identified in concrete leachate are volatile under standard environmental conditions. Therefore, inhalation of COPCs from surface water is not a complete exposure pathway and this evaluation did not consider it in Step 4 (Screening Assessment).

Dermal contact may occur through direct contact with surface water while swimming. However, absorption through the skin is limited compared to other exposure pathways because of the relatively low lipid solubility of most metals (Paustenbach, 2000 and Hostynek et al., 1998 as cited in US EPA, 2007a). Because dermal contact with surface water is not a pathway that may drive exposures, this evaluation did not consider it in Step 4 (Screening Assessment).

Exposure to Air

Concrete is a porous solid; therefore, gases and vapors are able to diffuse through the interstitial pores and emanate into indoor air. Elemental mercury is the only COPC identified that readily vaporizes within the range of standard temperature and pressure conditions found in habitable buildings.

Inhalation of mercury vapor may occur in closed indoor environments as mercury vapor accumulates due to low air circulation. Therefore, this evaluation retained the inhalation of indoor air as an exposure pathway of potential concern for human receptors. Ecological receptors are not anticipated to have any appreciable direct contact with indoor air.

This evaluation also considered dermal contact with mercury vapor to be a negligible exposure pathway. Past studies have demonstrated that the amount of inorganic mercury adsorbed through the skin is small when compared to the amount adsorbed through the lungs (Hursh et al., 1989 as cited in US EPA, 1997a). Because dermal contact with mercury vapor is not a pathway that may drive exposures, this evaluation did not consider it in Step 4 (Screening Assessment).

3.1.2 Potential Receptors for Fly Ash Concrete

Human Receptors

Due to the prevalence of concrete as a building material, human receptors may be exposed to COPCs in industrial, commercial, or residential settings. Of these receptor types, residential receptors are the most likely to be HEIs, due to the longer duration of time spent indoors, as well as the generally smaller ratio of air volume to wall surface area in residential buildings compared to offices or industrial workspaces. Residential receptors are also the only human receptors anticipated to be exposed to COPCs through ingestion of untreated ground water as a source of potable water. Commercial and industrial workspaces are generally connected to a municipal drinking water source that is treated, regulated, and monitored prior to distribution. Finally, recreational fishers may be exposed to COPCs through ingestion

of fish that have been exposed to and accumulated the COPCs from contact with surface water, sediment, and biota. **Figure 3-1** shows the conceptual exposure model developed for human receptors. Dashed lines represent exposures or receptors that may be present, but were not directly evaluated in Step 4 (Screening Assessment) because they do not drive high-end exposures.

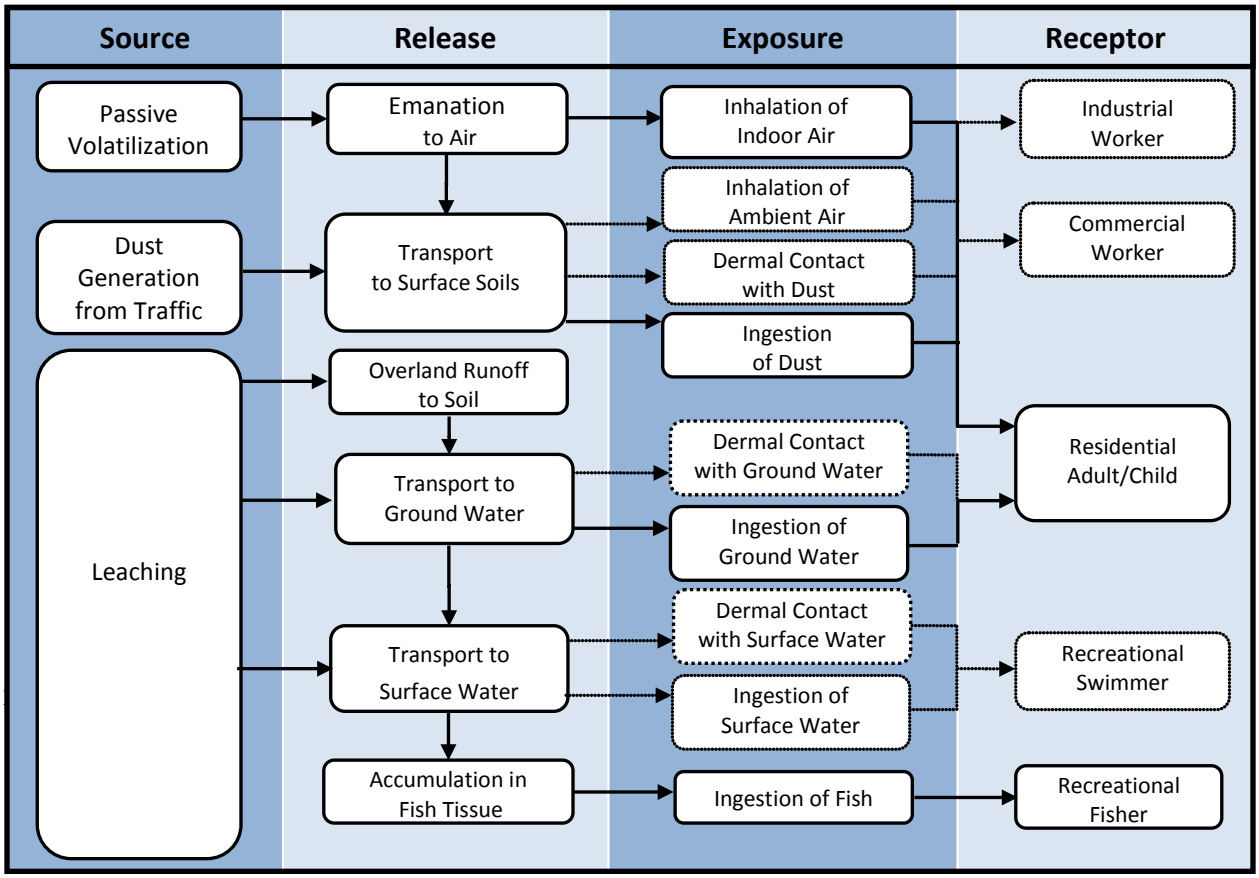


Figure 3-1: Human conceptual exposure model for fly ash concrete.

Ecological Receptors

The current evaluation identified plants, invertebrates, amphibians, fish, birds, and mammals as the relevant classes of ecological receptors. On a national scale, any of these receptor types may be present downgradient of a concrete source. The evaluation did not select specific ecological receptors at this step because the most sensitive receptor may differ on a case-by-case basis, depending on both the species and COPC present in a given environment. Instead, sensitive ecological receptors for each COPC were determined during development of screening benchmarks based on available toxicological data (see **Appendix B**). **Figure 3-2** shows the conceptual exposure model developed for ecological receptors.

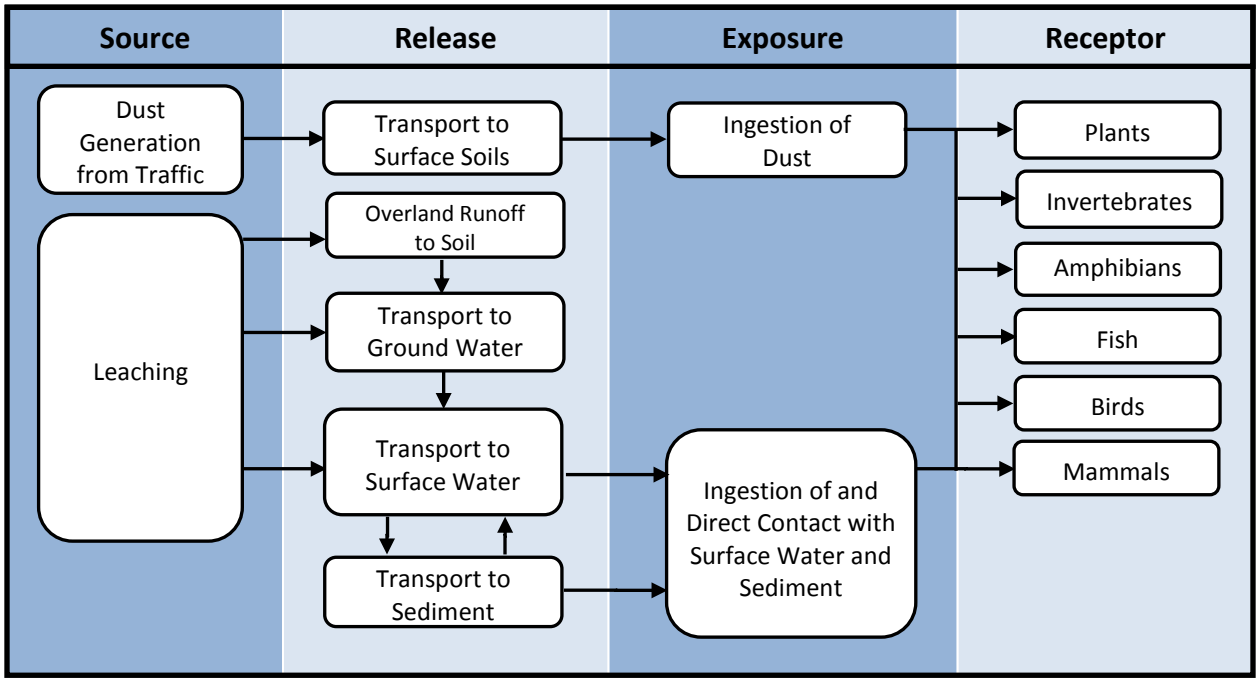


Figure 3-2: Ecological conceptual exposure model for fly ash concrete.

3.2 FGD Gypsum Wallboard

This subsection describes the potential exposure pathways and receptors for mercury, the single COPC identified for FGD gypsum wallboard. While mercury may change valence states, form complexes with other ions or compounds, or undergo other reactions that reduce its mobility or bioavailability, it will not naturally degrade. Once released into the environment, mercury will persist indefinitely. Therefore, where releases are possible, so are exposures.

3.2.1 Potential Exposure Pathways for FGD Gypsum Wallboard

Exposure to Air

Wallboard is a porous solid; therefore, gases and vapors are able to diffuse through the interstitial pores and emanate into indoor air. Elemental mercury is the only COPC identified that readily vaporizes within the range of standard temperature and pressure conditions found in habitable buildings.

Inhalation of mercury vapor may occur in closed indoor environments as mercury vapor accumulates due to low air circulation. Therefore, this evaluation retained the inhalation of indoor air as an exposure pathway of potential concern for human receptors. Ecological receptors are not anticipated to have any appreciable direct contact with indoor air.

The evaluation also considered dermal contact with mercury vapor to be a negligible exposure pathway. Past studies have demonstrated that the amount of inorganic mercury adsorbed through the skin is small when compared to the amount adsorbed through the lungs (Hursh et al., 1989 as cited in US EPA, 1997a). Because dermal contact with mercury vapor is not a pathway that may drive exposures, this evaluation did not consider it in Step 4 (Screening Assessment).

3.2.2 Potential Receptors for FGD Gypsum Wallboard

Human Receptor

Due to the prevalence of FGD gypsum wallboard as a building material, human receptors may be exposed to COPCs in industrial, commercial, and residential settings. Of these receptor types, residential receptors are the most likely to be HEIs, due to the longer duration of time spent in residential buildings, as well as the generally smaller ratio of air volume to wall surface area of residential buildings compared to offices or industrial workspaces. **Figure 3-3** shows the conceptual exposure model developed for human receptors. Dashed lines represent releases, exposure pathways, or receptors that may be present, but were not directly evaluated in Step 4 (Screening Assessment) because they do not drive high-end exposures.

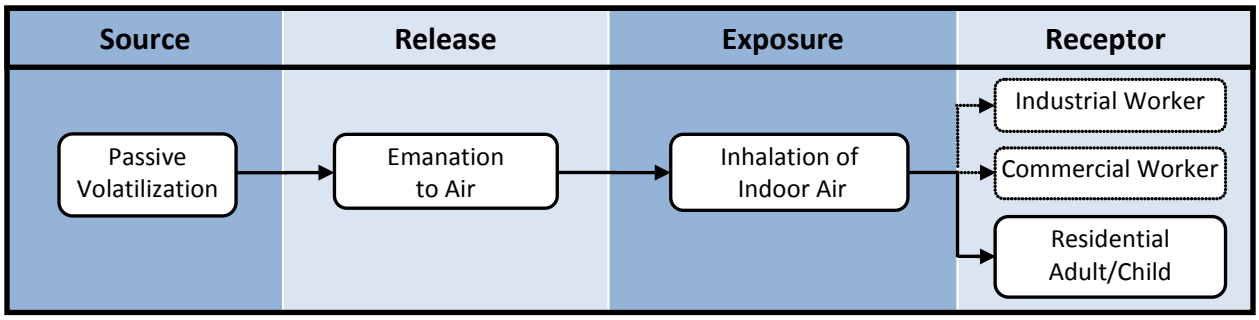


Figure 3-3: Human conceptual exposure model for FGD gypsum wallboard.

Ecological Receptors

Ecological receptors are not anticipated to have any appreciable direct contact with indoor air, and were not retained for further evaluation.

3.3 Conclusions of Step 3

The evaluation did not eliminate any releases or associated COPCs in this step based on a review of potential exposure pathways and HEIs. Therefore, all of the COPCs identified in Step 2 (Comparison of Available Data) will proceed to Step 4 (Screening Assessment). **Table 3-1** provides a list of the remaining COPCs following this step of the evaluation.

Table 3-1: List of COPCs Remaining Following Step 3

COPC	Fly Ash Concrete			FGD Gypsum Wallboard
	Dust	Ground and Surface Water	Air	Air
Aluminum	X	--	--	--
Antimony	X	X	--	--
Arsenic	X	--	--	--
Barium	X	--	--	--
Beryllium	X	--	--	--
Boron	X	X	--	--
Cadmium	X	--	--	--
Chromium	X	X	--	--
Cobalt	X	--	--	--
Copper	X	--	--	--
Iron	X	--	--	--
Lead	X	--	--	--
Manganese	--	--	--	--
Mercury	X	--	X	X
Molybdenum	X	--	--	--
Nickel	X	--	--	--
Selenium	X	X	--	--
Silver	--	--	--	--
Strontium	X	--	--	--
Thallium	X	--	--	--
Uranium	X	--	--	--
Vanadium	X	--	--	--
Zinc	X	--	--	--

COPC = Constituent of Potential Concern
FGD = Flue Gas Desulfurization
X = Constituent requiring further evaluation
-- = Constituent requiring no further evaluation

Step 4: Screening Assessment

The purpose of this section is to apply the fourth step of the encapsulated beneficial use methodology to the current evaluation of fly ash concrete and FGD gypsum wallboard, based on the COPCs carried forward from Step 3. This screening used conservative (i.e., likely to overestimate exposures) environmental, fate and transport, and exposure data to estimate COPC exposures that may occur. The evaluation compared these conservative exposure concentrations to screening benchmarks drawn from established values (e.g., ecological soil screening levels) and/or health-based values calculated for this specific evaluation based on available toxicological and exposure data (i.e., based on a cancer risk of 1×10^{-5} or a hazard quotient of 1.0). **Appendix B** provides a discussion of the considerations involved in developing an appropriate set of screening benchmarks.

4.1 Fly Ash Concrete

This subsection details the screening assessment conducted for fly ash concrete. For each exposure pathway carried forward from Step 3 (Exposure Review), this subsection discusses the different approaches used to evaluate each exposure scenario, as well as the results of the screenings.

4.1.1 Exposure to Concrete Dust

In Step 3 (Exposure Review), the evaluation concluded that the highest exposures to concrete dust result from incidental ingestion of dust that has accumulated surface soils. In this step, the evaluation estimated an upper bound for the COPC concentrations in surface soil that may result from the use of fly ash in concrete, and compared these concentrations directly to relevant screening benchmarks. First, the 90th percentile contribution of fly ash to COPC concentrations in concrete dust was calculated probabilistically by multiplying the following three distributions together:

- The range of cement use rates in concrete were drawn from the Portland Cement Association (PCA, No Date a,b). Typical values range from seven percent to 15 percent of the total concrete mass. For ease of calculation, the evaluation divided this continuous range of values into eight discrete data points in increments of one percent under the assumption that each of the data points had an equal probability of occurring.
- The range of cement replacement rates were based on the ASTM standard for blended cement concrete (ASTM C595), which limits the amount of portland cement replaced by fly ash to below 40 percent. Therefore, the evaluation selected a range of five percent to 40 percent of the cement used. For ease of calculation, the evaluation divided this continuous range of values into eight discrete data points in increments of five percent under the assumption that each of the data points had an equal probability of occurring.
- The COPC concentrations in fly ash were drawn from data in the CCR Constituent Database. **Appendix A** provides more information on each of these data sources contained in this database.

Next, the evaluation conservatively accounted for the dilution and attenuation that can occur during the transport of concrete dust to nearby surface soil. The evaluation did not identify any literature that

specifically addressed this topic, and instead relied on the findings of US EPA (2002), as discussed in US EPA (2010a). These findings pertain to unmitigated transport of ash from uncovered CCR landfills by wind and overland runoff, and show that these overland transport routes could result in CCRs accounting for up to 10 percent of nearby surface soil. Therefore, the current evaluation divided the 90th percentile dust concentrations by a dilution and attenuation factor of 10 to account for incorporation into surface soil. This represents a conservative assumption for the current evaluation because a much greater quantity of fly ash is available for transport from an uncovered landfill, compared to an intact concrete road, at any given time. **Table 4-1** presents the comparison of these soil concentrations to relevant screening benchmarks.

Table 4-1: Comparison of the Fly Ash Contribution to Concrete Dust Concentrations to Human Health and Ecological Screening Benchmarks for Soil (mg/kg)

COPC	Exposure Point Concentration	Human Soil Ingestion Benchmark	Source	Result	Ecological Soil Benchmark	Source	Result
Aluminum	407	59,443	HBN	**Screen Out**	--	--	--
Antimony	0.05	23.8	HBN	**Screen Out**	0.27	Eco-SSL	**Screen Out**
Arsenic	0.49	3.6	HBN	**Screen Out**	18.0	Eco-SSL	**Screen Out**
Barium	18.0	11,889	HBN	**Screen Out**	330	Eco-SSL	**Screen Out**
Beryllium	0.06	119	HBN	**Screen Out**	21.0	Eco-SSL	**Screen Out**
Boron	3.2	11,889	HBN	**Screen Out**	--	--	--
Cadmium	0.02	59.4	HBN	**Screen Out**	0.36	Eco-SSL	**Screen Out**
Chromium (VI)	0.59	10.8	HBN	**Screen Out**	130	Eco-SSL	**Screen Out**
Cobalt	0.29	17.8	HBN	**Screen Out**	13.0	Eco-SSL	**Screen Out**
Copper	0.85	594	HBN	**Screen Out**	28.0	Eco-SSL	**Screen Out**
Iron	153	41,610	HBN	**Screen Out**	--	--	--
Lead	0.37	400	Action Level	**Screen Out**	11.0	Eco-SSL	**Screen Out**
Mercury	0.002	17.8	HBN	**Screen Out**	--	--	--
Molybdenum	0.35	297	HBN	**Screen Out**	--	--	--
Nickel	0.59	1,189	HBN	**Screen Out**	38.0	Eco-SSL	**Screen Out**
Selenium	0.07	297	HBN	**Screen Out**	0.52	Eco-SSL	**Screen Out**
Strontium	5.0	35,666	HBN	**Screen Out**	--	--	--
Thallium	0.02	0.60	HBN	**Screen Out**	--	--	--
Uranium	0.04	178	HBN	**Screen Out**	--	--	--
Vanadium	1.5	535	HBN	**Screen Out**	7.8	Eco-SSL	**Screen Out**
Zinc	1.3	17,833	HBN	**Screen Out**	46.0	Eco-SSL	**Screen Out**

HBN = health-based number
COPC = constituent of potential concern
Eco-SSL = ecological soil screening level
mg/kg = milligrams per kilogram
-- = benchmark not available

The results of this comparison indicate that the fly ash contribution to COPC exposures from concrete falls below all relevant screening benchmarks. Based on this comparison, exposures to fly ash concrete dust do not warrant further investigation for either human or ecological receptors.

4.1.2 Exposure to Ground and Surface Water

In Step 3 (Exposure Review), the evaluation concluded that the highest exposure to concrete leachate may result from the use of impacted ground water as a source of drinking water, ingestion of fish from impacted surface water, or direct contact with impacted surface water. In this step, the evaluation conducted ground and surface water screening in two separate stages. The intent of these stepwise screenings was to eliminate COPCs that did not warrant further consideration under a more realistic, resource intensive modeling scenario.

First, a preliminary conservative screening was conducted by comparing the maximum COPC concentrations drawn from Kosson et al. (2013) and Garrabrants et al. (2013) to relevant screening benchmarks. This approach assumed that the COPC concentrations present in both ground and surface water were the same as in undiluted leachate. **Appendix C** provides a detailed discussion of the data and methods used to calculate these leachate concentrations. **Table 4-2** presents the results of the comparison to screening benchmarks for human health. For ease of comparison, all screening benchmarks were standardized to represent the ground or surface water concentrations below which no further evaluation is warranted. COPC concentrations higher than the benchmarks in this preliminary screening do not indicate the presence of elevated risks; only that further evaluation may be warranted.

Table 4-2: Comparison of Undiluted Leachate to Human Health Screening Benchmarks for Ground and Surface Water (µg/L)

COPC	Exposure Point Concentration		Tap Water Ingestion Benchmark	Source	Result		Fish Ingestion Benchmark	Source	Result
Antimony	1.1		2.8	HBN	**Screen Out**		--	--	--
Boron	4.9		1,390	HBN	**Screen Out**		--	--	--
Chromium (VI)	6.2		1.4	HBN	**Retain**		240	HBN	**Screen Out**
Selenium	1.3		34.8	HBN	**Screen Out**		3.6	HBN	**Screen Out**

COPC = constituent of potential concern
HBN = health-based number
µg/L = micrograms per liter
-- = benchmark not available

Table 4-3 presents the results of the comparison to screening benchmarks for ecological receptors. For ease of comparison, all screening benchmarks were standardized to represent the surface water concentrations below which no further evaluation is warranted. COPC concentrations higher than the benchmarks in this preliminary screening do not indicate the presence of elevated risks; only that further evaluation may be warranted. This table does not include antimony because the current evaluation did not identify a relevant screening benchmark for this COPC.

Table 4-3: Comparison of Undiluted Leachate to Ecological Screening Benchmarks for Surface Water (µg/L)

COPC	Exposure Point Concentration		Surface Water Benchmark	Source	Result		Sediment Benchmark	Source	Result
Boron	4.9		750	AWQC	**Screen Out**		--	--	--
Chromium (VI)	6.2		11.0	AWQC	**Screen Out**		744	TEL	**Screen Out**
Selenium	1.3		5.0	AWQC	**Screen Out**		--	--	--

COPC = constituent of potential concern
AWQC = ambient water quality criteria
TEL = threshold effects level
µg/L = micrograms per liter
-- = benchmark not available

The results of the comparison indicate that the concentrations of antimony, boron, and selenium fall below all relevant screening benchmarks identified for ground and surface water. Therefore, the evaluation did not retain these three constituents for further consideration. The undiluted concrete leachate concentration of chromium (VI) was higher than the human health-based number (HBN) for tap water ingestion. Therefore, a second round of screening was conducted that conservatively accounted for the dilution and attenuation that occurs in the environment prior to exposure. This evaluation used the Industrial Waste Evaluation Model (IWEM) as the most appropriate ground water model. **Appendix C** provides a detailed discussion of the inputs and assumptions used in this model. **Table 4-4** compares the modeled 90th percentile well concentration of chromium (VI) to the same HBN for tap water ingestion.

Table 4-4: Comparison of Modeled Well Concentrations to Remaining Human Health Screening Benchmark for Ground Water (µg/L)

COPC	Exposure Point Concentration		Tap Water Ingestion	Source	Result
Chromium (VI)	1.1		1.4	HBN	**Screen Out**

COPC = constituent of potential concern
HBN = health-based number
µg/L = micrograms per liter

The results of the comparison indicate that chromium (VI) falls below the relevant screening benchmark for tap water ingestion. Therefore, the evaluation did not retain chromium (VI) as a COPC for further consideration. Based on this comparison, exposures to fly ash concrete leachate do not warrant further consideration for either human or ecological receptors.

4.1.3 Exposure to Indoor Air

In Step 3 (Exposure Review), the evaluation concluded that the highest exposures to mercury vapor result from inhalation of the air inside residences constructed with fly ash concrete. In this step, the evaluation first estimated an upper bound on the mercury concentrations that may occur in indoor air as a result of the use of fly ash in concrete. **Equation 4.1** was used to probabilistically calculate a 90th percentile air concentration for a conservative exposure scenario. This equation represents a simple mass

balance equation derived under the assumption of steady-state, the idealized condition where losses and gains of mercury have reached equilibrium and the mercury concentration remains constant.

Equation (4.1) $C = \frac{E \times SA}{\alpha \times V}$

Where:

E	-	Mercury Emanation Rate (ng/m^2-hr)
α	-	Air Changes per Hour (1/hr)
SA	-	Product Surface Area (m^2)
V	-	Housing Unit Volume (m^3)
C	-	Steady-State Mercury Air Concentration (ng/m^3)

- Mercury emanation rates of 4.4 and 15.6 ng/m^2-hr measured by Golightly et al. (2005; 2009) were identified as the most appropriate values for this comparison because they represent concrete allowed to cure for 56 days. It has been shown that the concrete matrix becomes gradually denser with time for at least a year after mixing (Garboczi, 1995). A denser concrete matrix reduces the size of interstitial pores and, consequently, reduces the rate at which mercury vapor can escape to indoor air. Therefore, of the available data summarized in **Section 2.1.3**, the samples collected after 56 days are most representative of long-term exposures. The evaluation adjusted these emanation rates using the range of fly ash mercury concentrations in the CCR Constituent Database (**Appendix A**) under the assumption that the mercury emanation rate from concrete changes linearly as a function of the mercury concentration in concrete.
- The air exchange rate is the number of times that the total volume of air in a housing unit is exchanged with outside air during a given time period. Values were drawn from Koontz and Rector (1995), cited in the 1997 *Exposure Factors Handbook* (US EPA, 1997). The current evaluation incorporated the reported distribution of national air exchange rates between the 5th percentile [0.15 air changes per hour (ACH)] and the 95th percentile (1.74 ACH). The maximum air exchange rate of 23.3 ACH was omitted because it is unlikely to reflect the scenario under evaluation. No minimum air exchange rate was reported by this study.
- Product surface area is the total surface area of the CCR product exposed to indoor air. It was conservatively assumed that (at a minimum) the ceiling, floor, and four exterior walls of the residence were constructed with concrete. This evaluation assumed a square floor plan with a ceiling 2.4 m (8 ft) high. The International Building Code (IBC) was consulted to determine the total number of interior walls that may be present in a building of the size modeled (ICC, 2006b).[10] The 2006 IBC requires that the floor area of at least one room in a housing unit be 11 m^2 (120 ft^2) or larger, while all remaining habitable rooms must have floor areas of at least 6.6 m^2 (70 ft^2). Based on these parameters and the range of home unit volumes, the evaluation

[10] The International Building Codes (IBC) are building codes developed and maintained by the International Code Council. At present, many state and local governments have adopted the 2006 IBC or a more recent iteration either statewide or by an individual county.

determined that a maximum of between three and four full-length walls may be present. Adding additional walls would result in a building that is out of code. Because both sides of the interior walls are exposed to indoor air, their surface area is twice that of the external walls. The evaluation assumed that all possible surface areas between this minimum and maximum were equally likely to occur.

- Housing unit volume is the total internal volume of a housing unit. This evaluation considered volumes between 153 m^3 (5,439 ft^3) and 492 m^3 (5,439 ft^3), which are the 10^{th} percentile and average values, respectively, for owned and rented properties listed in Table 19-1 of the 2011 *Exposure Factors Handbook* (US EPA, 2011). These data were drawn from the 2011 edition because the 1997 edition only reports a median value. This evaluation used the volume of the total housing unit, rather than a single room, because the air exchange rates measured are for entire buildings, rather than individual rooms. The evaluation assumed that all possible housing volumes between these two sizes were equally likely to occur.

A 90^{th} percentile indoor air mercury concentration was calculated by probabilistically varying the inputs in **Equation 4-1**. The evaluation then compared this concentration to the relevant screening benchmark identified in **Appendix B**. The comparison results are presented in **Table 4-5**.

Table 4-5: Comparison of Indoor Air Mercury Concentration from Fly Ash Concrete to Human Health Screening Benchmark (ng/m^3)

Constituent	Exposure Point Concentration		Inhalation Benchmark	Source	Result
Mercury	100		300	RfC	**Screen Out**

ng/m^3 = nanograms per meter cubed
RfC = reference concentration

This comparison shows that the mercury concentration in indoor air resulting from fly ash concrete is below the relevant screening benchmark, even for the conservative scenario evaluated. Based on this comparison, exposures to mercury vapor emitted by fly ash concrete do not warrant further consideration for human receptors.

4.2 FGD Gypsum Wallboard

This subsection details the screening assessment conducted for FGD gypsum wallboard. Inhalation of mercury vapor was the single exposure pathway carried forward from Step 3 (Exposure Review) for this CCR product. Thus, this subsection describes the approach used to evaluate this exposure scenario, as well as the results of the screening.

4.2.1 Exposure to Indoor Air

In Step 3 (Exposure Review), the evaluation concluded that the highest exposures to mercury vapor result from inhalation of the air inside residences constructed with FGD gypsum wallboard. In this step, the evaluation first estimated an upper bound on the mercury concentrations that may occur in indoor air

as a result of the use of FGD gypsum in wallboard. **Equation 4.1** was used to probabilistically calculate a 90th percentile air concentration for a conservative exposure scenario.

- Because of the relatively small number of measurements available, the evaluation selected the highest measured mercury emanation rate in the available literature of 0.34 ng/m^2-hr (Shock et al., 2009). Unlike concrete, the internal structure of wallboard does not change appreciably after the product has been put into use. Therefore, it is unlikely that the mercury emanation rate of FGD gypsum wallboard will decrease over time like concrete. Because much of the available data on mercury concentrations were summary statistics, it was difficult to develop a distribution of emanation rates. Instead, to ensure that this screen effectively captured an upper bound of the potential releases of mercury, the current evaluation used the maximum reported mercury concentration in FGD gypsum of 3.1 mg/kg to adjust the emanation rate from Shock et al. (2009) under the assumption that the mercury emanation rate from the wallboard changes linearly as a function of the mercury concentration in the wallboard.
- The distribution of air exchange rates used was the same as described in **Section 4.1.3**.
- The distribution of product surface areas used was the same as described in **Section 4.1.3**, except it was assumed that wallboard would not be used in floor construction.
- The distribution of housing unit volumes used was the same as described in **Section 4.1.3**.

A 90th percentile indoor air mercury concentration was calculated by probabilistically varying the inputs in **Equation 4-1**. This evaluation then compared the calculated concentration to the relevant screening benchmark identified in **Appendix B**. The comparison results are presented in **Table 4-6**.

Table 4-6: Comparison of Indoor Air Mercury Concentrations from FGD Gypsum Wallboard to Human Health Screening Benchmark (ng/m^3)

Constituent	Exposure Point Concentration		Inhalation Benchmark	Source	Result
Mercury	49.0		300	RfC	**Screen Out**

ng/m^3 = nanograms per meter cubed
RfC = reference concentration

This comparison shows that the concentration of mercury in indoor air resulting from FGD gypsum wallboard is below the relevant screening benchmark, even for the conservative scenario evaluated. Based on this comparison, exposures to mercury vapor emitted by FGD gypsum wallboard do not warrant further consideration for human receptors.

4.3 Conclusions of Step 4

By the end of this step, all of the COPCs identified for fly ash concrete and FGD gypsum wallboard in Step 1 (Literature Review and Data Collection) were eliminated from further consideration. Thus, the evaluation did not proceed on to Step 5 (Risk Assessment). The analytical results of the first four steps indicate that environmental releases from these CCR products are comparable to or lower than those from analogous products, or are at or below relevant screening benchmarks. To confirm these findings, a

review was conducted to identify and characterize the major sources of uncertainty present in the evaluation. The results of this review, which are presented in **Section 5**, demonstrate that there is a high degree of confidence in the analytical results.

Section 5: Results, Uncertainties, and Conclusions

The purpose of this section is to summarize the results of the current evaluation of fly ash concrete and FGD gypsum wallboard. This evaluation was conducted according to the *Methodology for Evaluating Encapsulated Beneficial Uses of Coal Combustion Residuals* (US EPA, 2013a). Thus, this section summarizes the results of the evaluation, the uncertainties present in the evaluation, the potential impact of these uncertainties on the results of the evaluation, and ultimate conclusions regarding these beneficial uses.

5.1 Summary of Results

The primary goal of this document is to determine whether EPA should support the continued use of coal fly ash in concrete and FGD gypsum in wallboard. In addition, this document provides a clear example of how to conduct such an analysis and demonstrate an appropriate level of documentation. For these purposes, the current evaluation considered the two largest encapsulated beneficial uses of CCRs in the United States: fly ash used as a direct substitute for portland cement in concrete and FGD gypsum used as a replacement for mined gypsum in wallboard. These CCR products may be variable in their composition. However, this evaluation only addressed those products that meet relevant physical and performance standards established for these products by voluntary consensus standard bodies, that conform to specific design criteria identified in this evaluation (e.g., ≤ a 40 percent fly ash replacement rate), and that incorporate fly ash and FGD gypsum from common pollution control devices used in the United States. This evaluation also did not address products that contain additional additives or industrial materials that may alter releases from the products.

5.1.1 Fly Ash Concrete

Step 1 (Literature Review and Data Collection): From the available literature, the evaluation identified the generation of dust, emanation to air, leaching to ground and surface water and decay of naturally occurring radionuclides as potential releases that may occur from fly ash concrete during use. During the review of collected literature, the evaluation also identified several existing evaluations of sufficient applicability and quality to rely upon. This review found:

- None of the existing evaluations identified provided a sufficient rationale to eliminate COPCs from the evaluation of dust releases from fly ash concrete. As a result, a total of 23 COPCs were identified based on the chemical composition of fly ash and available toxicological data. These COPCs included: aluminum, antimony, arsenic, barium, beryllium, boron, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, molybdenum, nickel, selenium, silver, strontium, thallium, uranium, vanadium, and zinc.
- Several existing evaluations were found to be relevant to releases to ground and surface water (US EPA, 2010a; 2012a). Based on these existing evaluations, the evaluation retained a total of nine COPCs for more detailed examination. These COPCs included: antimony, arsenic, boron, cadmium, chromium, lead, molybdenum, selenium, and thallium.

- None of the existing evaluations identified provided a sufficient rationale to eliminate COPCs from the evaluation of releases to air. The constituent associated with the fly ash concrete that may volatilize under standard environmental conditions is mercury. Therefore, mercury was retained as a COPC for this release.
- Several existing evaluations were found relevant to radionuclides in fly ash concrete. The cumulative body of evidence provided by these evaluations was considered sufficient to eliminate radionuclides from further consideration.

Step 2 (Comparison of Available Data): To the extent practicable, the evaluation aggregated the data identified in Step 1 to allow a comparison of the releases of COPCs from fly ash concrete and portland cement concrete. The type of comparison conducted depended on the amount of data available. If a given COPC demonstrated the potential to be released at a higher rate from fly ash concrete than from portland cement concrete, the current evaluation retained that COPC for further evaluation in subsequent steps of the evaluation. The following comparisons were conducted:

- A statistical comparison of COPC concentrations in raw fly ash and portland cement using the ProUCL statistical software for dust releases. This evaluation used these raw materials as a surrogate for fly ash concrete and portland cement concrete, respectively. The results of the comparison showed that concentrations of manganese and silver are likely to be higher in portland cement concrete than fly ash concrete. Therefore, this evaluation did not retain these two COPCs for further consideration. The remaining 21 COPCs either demonstrated the potential to be higher in fly ash concrete, or did not have sufficient portland cement data to conduct a comparison, and were retained for further consideration.
- A direct comparison of the range of measured leachate concentrations was conducted for releases to ground and surface water. Because this leachate is the source of releases to both ground and surface water, the current evaluation used the same comparison for both media. The results of the comparison showed that all available measurements of arsenic, cadmium, lead, molybdenum, and thallium were below detection. Based on these findings, this evaluation did not retain these COPCs for further evaluation. Antimony, boron, chromium, and selenium demonstrated the potential to be released at higher rates from fly ash concrete than from portland cement concrete in one or more samples. Therefore, these four COPCs were retained for further consideration.
- A direct comparison of the range of measured mercury emanation rates was conducted for releases to air. The available data indicated the potential for mercury to be released at higher rates from fly ash concrete than from portland cement concrete. Therefore, this COPC was carried forward to the next step of the evaluation.

Step 3 (Exposure Review): This evaluation reviewed the releases carried forward from Step 2 to identify any exposures that may occur. Where the evaluation identified multiple exposure scenarios, it retained the scenarios likely to result in the highest chronic exposures for further evaluation. The evaluation identified:

- Incidental ingestion of dust generated from concrete roadways exposed to traffic with studded tires was identified as a potentially complete exposure pathway. This evaluation identified nearby residential receptors and foraging ecological receptors as the highly exposed receptors.
- Use of ground water contaminated by concrete leachate as a source of potable water was identified as a potentially complete exposure pathway for humans. The evaluation identified nearby residential human receptors as HEIs in this scenario. Ecological receptors were assumed to have negligible contact with ground water and were not retained as potential receptors.
- Ingestion of fish caught from surface water bodies that receive runoff or ground water discharge that has been contaminated by leachate from fly ash concrete was identified as a potentially complete exposure pathway. The evaluation identified recreational fishers and their families as HEIs in this scenario. This evaluation also identified direct contact and ingestion of impacted surface water as a scenario of concern for ecological receptors. Any receptor living in or near the water body may be exposed.
- Inhalation of mercury vapor in indoor air was identified as a potentially complete exposure pathway. The evaluation identified residential receptors as HEIs in this scenario.

Step 4 (Screening Assessment): This evaluation conducted a conservative screening assessment for each exposure pathway identified in Step 3. This evaluation used conservative (i.e., likely to overestimate exposures) environmental, fate and transport, and exposure data to estimate COPC concentrations at the point of exposure. It then compared these concentrations to relevant regulatory and health-based screening benchmarks to determine if more in-depth modeling was warranted.

- The evaluation calculated the 90^{th} percentile contribution of fly ash concrete to COPC concentrations in surface soil mixed with concrete dust and compared the calculated concentration to relevant screening benchmarks. The evaluation found concentrations of all COPCs to be below all relevant screening benchmarks for human and ecological receptors. Therefore, the evaluation did not retain this exposure pathway for further consideration.
- The evaluation compared the maximum leachate concentrations from fly ash concrete directly to the relevant screening benchmarks in the first stage of the screening for both ground and surface water. These undiluted leachate concentrations were below all relevant surface water screening benchmarks. Therefore, the evaluation did not retain this exposure pathway for further consideration. The undiluted leachate concentrations were also below the relevant ground water screening benchmarks for antimony, boron, and selenium. However, the undiluted concentration of chromium (VI) exceeded the screening benchmark for ingestion of ground water. Therefore, the evaluation conducted a second round of screening to conservatively account for dilution and attenuation in the environment prior to receptor exposures (see **Appendix C**). The revised chromium (VI) concentration was below all screening benchmarks. Therefore, the evaluation did not retain this exposure pathway for further consideration.
- The evaluation probabilistically calculated a 90^{th} percentile indoor air mercury concentration based on a conservative scenario. This concentration was compared to the relevant screening benchmark

and found to be lower. Therefore, this evaluation did not retain this exposure pathway for further consideration.

Conclusion: The current evaluation eliminated all of the COPCs associated with each exposure scenario by the end of Step 4. Based on these results, no further evaluation of releases of COPCs from fly ash concrete was warranted and the evaluation did not proceed to Step 5.

5.1.2 Flue Gas Desulfurization Gypsum Wallboard

Step 1 (Literature Review and Data Collection): Drawing from the available literature, this evaluation identified generation of dust, emanation to air, leaching to ground and surface water and decay of naturally occurring radionuclides as potential releases that may occur from FGD gypsum wallboard. During the review of collected literature, several existing evaluations were of sufficient applicability and quality to rely upon in the current evaluation. This evaluation identified:

- One existing evaluation was found to be relevant to releases from FGD gypsum wallboard. Based on this evaluation, the evaluation did not retain releases to dust, ground water, and surface water for further consideration.
- None of the existing evaluations identified provided a sufficient rationale to eliminate COPCs from the evaluation of releases to air. The constituent associated with the FGD gypsum wallboard that may volatilize under standard environmental conditions is mercury. Therefore, mercury was retained as a COPC for this release.
- Several existing evaluations were found to be relevant to radionuclides in FGD gypsum wallboard. The cumulative body of evidence provided by these evaluations was considered sufficient to eliminate radionuclides from further consideration.

Step 2 (Comparison of Available Data): To the extent practicable, the evaluation aggregated all of the data identified in Step 1 to allow a comparison of the releases of the one COPC, mercury, from FGD gypsum wallboard and mined gypsum wallboard. The type of comparison depended on the amount of data available. If a given COPC demonstrated the potential to be released at a higher rate from FGD gypsum wallboard than from mined gypsum wallboard, the evaluation retained that COPC for further consideration.

- A direct comparison of the range of measured mercury emanation rates was conducted for releases to air. The available data indicated the potential for mercury to be released at higher rates from FGD gypsum wallboard than from mined gypsum wallboard. Therefore, this COPC was carried forward to the next step of the evaluation.

Step 3 (Exposure Review): The evaluation reviewed releases of COPCs carried forward from Step 2 to identify any exposures that may occur. Where multiple exposure scenarios were identified, the evaluation retained the ones likely to result in the highest chronic exposures for further consideration.

- Inhalation of mercury vapor in indoor air was identified as a potentially complete exposure pathway. The evaluation identified residential receptors as HEIs in this scenario. Ecological

receptors were assumed to have negligible contact with indoor air and were not retained as potential receptors.

Step 4 (Screening Assessment): The current evaluation conducted a conservative screening assessment for the exposure scenario identified in Step 3 of the evaluation. It used conservative environmental, fate and transport, and exposure data to estimate COPC concentrations at the point of exposure. The evaluation then compared these concentrations to relevant regulatory and health-based screening benchmarks to determine if more in-depth modeling was warranted.

- The evaluation probabilistically calculated a 90^{th} percentile indoor air mercury concentration based on a conservative scenario. This concentration was compared to the relevant screening benchmark and found to be lower. Therefore, this evaluation did not retain this exposure pathway for further consideration.

Conclusion: The evaluation eliminated mercury, the one COPC associated with the exposure scenario, by the end of Step 4. Based on these results, no further evaluation of the releases of this COPC from FGD gypsum wallboard was warranted and the evaluation did not proceed to Step 5.

5.2 Sources of Uncertainty

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

- Data variability and heterogeneity introduce uncertainty when the exact range and distribution of relevant characteristics for constituents, environmental media, or receptors are not known. Variability and heterogeneity are a natural part of environmental systems that cannot be eliminated by further study. However, collection of additional data that better define these ranges and distributions can minimize the associated uncertainties.

- Models introduce uncertainty through the simplifying assumptions used to approximate real-world conditions, processes, and 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. Uncertainty can be minimized through use of the most appropriate model and by replacing any default assumptions with representative data.

- Limitations on the current state of the science may introduce uncertainty through the lack of scientific consensus or fundamental lack of knowledge of the system under evaluation. This can be the most difficult type of uncertainty to address. Neither the collection nor analysis of additional data is likely to reduce this uncertainty within the timeframe that a decision is needed.

Uncertainties in this evaluation were managed to the extent practicable by focusing the evaluation on high-end releases and exposures. In instances where the exact range of a particular variable was unknown, the evaluation relied on a conservative bounding estimate known to fall above or below the true range, as appropriate. This approach does not necessarily reduce the magnitude of uncertainties present, but does shift them in a direction that allows defensible conclusions to be drawn from the evaluation. The following subsections identify, where known, uncertainties specific to the current evaluation and the direction and magnitude of these uncertainties, as well as the potential impacts these uncertainties may have on the conclusions of the current evaluation.

5.2.1 Uncertainties for Dust Exposures

The uncertainties discussed in this section pertain to releases of dust from fly ash concrete during use by the consumer, and the resulting receptor exposures.

Available Data

Of the fly ashes generated through coal combustion, only a subset is suitable for beneficial use in concrete. There are requirements for silica content, loss of ignition, and other characteristics that must be met before a fly ash is considered appropriate for use (ASTM Standard C618). This type of detailed information is not available for the majority of fly ash samples. Therefore, the current evaluation considered all available fly ash data. It is unknown what portion of this dataset reflects these beneficial use specifications. However, through the use of high-end concentrations from a data set that is representative of the full range of fly ashes generated across the United States, this evaluation ensures that it also captures the subset of suitable fly ashes for beneficial use. Therefore, the fly ash dataset used in this evaluation may overestimate COPC releases. However, the magnitude of this overestimation is unknown.

Treatment of Non-detect Data

Non-detect data are concentrations that are present at levels below the capacity of an analytical instrument to differentiate from background noise. The presence of non-detects in any dataset introduces some amount of uncertainty because the concentration of a COPC is not known with certainty. The quantity of non-detects present in the available fly ash data varies by COPC. To calculate 90^{th} percentile concentrations for comparison with screening benchmarks, the current evaluation replaced non-detect values with half of the reported detection limit according to the recommendations in *Risk Assessment Guidance for Superfund (RAGS) Part A* (US EPA, 1989) and *EPA Region 3 Guidance on Handling Chemical Concentration Data near the Detection Limit in Risk Assessments* (US EPA, 1991). Because the evaluation relied on a value halfway between zero and the detection limit, the true value is equally likely to be higher or lower than the value assigned. To ascertain the impact this approach had on non-detect values, the evaluation compared the calculated 90^{th} percentile concentrations when non-detect values are set to zero, half the detection limit, and the detection limit. **Table 5-1** presents the results of this analysis.

Table 5-1: 90th Percentile Fly Ash Concentration with Different Non-Detect Treatments (mg/kg)

Sample Type	Detection Frequency	Non-Detect Treatment		
		Zeroed Non-Detects	Half Detection Limit	Detection Limit
Aluminum	22 / 22	126,139	126,139	126,139
Antimony	38 / 42	17.4	17.4	17.5
Arsenic	97 / 100	171	171	171
Barium	61 / 61	6,800	6,800	6,800
Beryllium	24 / 32	21.8	21.8	21.8
Boron	34 / 34	1,065	1,065	1,065
Cadmium	70 / 91	11.9	11.9	11.9
Chromium	91 / 91	170	170	170
Cobalt	49 / 49	89.2	89.2	89.2
Copper	47 / 48	302	302	302
Iron	23 / 23	61,425	61,425	61,425
Lead	98 / 99	165	165	165
Mercury	73 / 87	0.65	0.65	0.65
Molybdenum	44 / 46	187	187	187
Nickel	75 / 76	227	227	227
Selenium	71 / 79	20.3	20.3	20.3
Strontium	22 / 22	2,250	2,250	2,250
Thallium	19 / 27	5.3	6.5	11.6
Uranium	10 / 19	14.7	14.7	14.7
Vanadium	43 / 43	461	461	461
Zinc	51 / 51	583	583	583

mg/kg = milligrams per kilogram

In most cases, there is no difference between the various 90th percentile values, regardless of the method used to address non-detect values. None of the differences identified were great enough to alter the results of the evaluation. This is because the upper percentile exposures are predominately associated with higher, detectable concentrations in the distribution rather than the lower concentrations associated with non-detect values. This comparison shows that, this uncertainty is unlikely to impact the results of the evaluation.

Constituents Not Evaluated

The evaluation selected toxicity values for each constituent identified as a COPC according to the selection hierarchy detailed in the *Office of Solid Waste and Emergency Response 2003 Directive 9285.7-53* (US EPA, 2003b). However, several constituents lack both human health and ecological toxicity values (i.e., calcium, chloride, magnesium, phosphate, potassium, sodium, silicon, sulfate, and sulfur). The absence of toxicity values is not necessarily equivalent to the absence of toxicity. However, in the absence of other compelling information to indicate potential adverse effects from these constituents, the evaluation did not retain these constituents as COPCs for further consideration. The lack of toxicity values for these constituents may result in an underestimation of chronic risk to some

receptors. However, the magnitude of this underestimation is likely to be small because many of the constituents that do not have toxicity values are also known to be nutrients essential for life.

This evaluation did not address dust exposure for a few of the constituents that US EPA (2010a) identified as potentially present in CCRs (i.e., cyanide, fluoride, nitrate/nitrite) because the available data for these constituents were either from CCRs other than fly ash or fly ash mixed with other CCRs. US EPA (2010a) eliminated these constituents based on the results of a screening. A review of the data used in that screening found concentrations of these constituents in CCRs to be at least an order of magnitude below relevant screening benchmarks. While there is the potential for somewhat higher concentrations of constituents in pure fly ash, the current evaluation demonstrates that the dilution of fly ash into concrete combined with the dilution of concrete dust into surface soil reduces constituent concentrations present in the original fly ash by at least two orders of magnitude. This reduction, together with the low concentrations reported in US EPA (2010a), make the uncertainty introduced through the exclusion of these constituents small.

Portland Cement Use and Replacement Rates in Concrete

This evaluation modeled fly ash concretes as having a fly ash replacement rate between 5 percent and 40 percent of the portland cement used, based on the upper limit specified in current ASTM standards for blended cements (ASTM Standard C595). In addition, this evaluation modeled concretes as containing between 7 percent and 15 percent portland cement by mass, based on typical rates reported by the Portland Cement Association (PCA, No Date a,b). These ranges may not represent the complete range of theoretical concrete mixes. However, the current evaluation focused on the range typically used in practice.

When calculating the potential fly ash contribution to concrete, the evaluation assumed that each fly ash replacement rate and cement use rate is equally likely. Furthermore, the evaluation assumed that the fly ash replacement rate selected was independent of the cement use rate. Weighting all values equally is anticipated to bias calculated results high, because studies report fly ash replacement rates around 15 percent to be more common in practice (US EPA, 2012d). This assumption is considered appropriate in the absence of detailed information on the frequency at which different replacement rates occur in practice. These conservative assumptions are likely to overestimate potential releases. However, the magnitude of this overestimation is unknown.

Comparison to Analogous Products

The statistical comparison to analogous products conducted in Step 2 (Comparison of Available Data) indicated that concentrations of manganese and silver in fly ash are either less than or equal to those in portland cement. The evaluation considered the possibility that the statistical tests may be unduly influenced by a small number of extreme values present in the datasets. However, removal of the high-end values from either the fly ash or portland cement datasets did not change the results of the statistical tests. The evaluation also considered whether the data available from Eckert and Guo (1998) may overestimate COPC concentrations in portland cement because the data are from kilns co-fired with hazardous waste-derived fuels. While these data represent actual cements generated in the United States, it is unknown if the manganese concentrations measured in this relatively small number of samples are

any higher than the national distribution of portland cement. However, while there is some uncertainty associated with these data, the impact on the conclusions of the evaluation is negligible. Even if manganese had been retained through Step 2 (Comparison of Available Data), this COPC would have been screened out by a wide margin in Step 4 (Screening Assessment), when the 90th percentile exposure concentration of 1.5 mg/kg was compared to an HBN of 2,794 mg/kg and an Eco-SSL of 220 mg/kg (US EPA, 2007b). Silver concentrations were not reported in Eckert and Guo (1998).

The evaluation retained all other COPCs for further consideration. Although only a small amount of data on concentrations of some COPCs in portland cement was available for comparison (e.g., aluminum, boron, molybdenum), the evaluation found these COPCs to be higher in fly ash and retained them for further consideration. The current evaluation considers the consequences of incorrectly accepting the null hypothesis (H_0) and removing a COPC from consideration (i.e., Type II Error) more severe than incorrectly rejecting H_0 and retaining the COPC for further consideration in subsequent stages of the evaluation (i.e., Type I Error). Therefore, the uncertainty associated with the potential error due to incorrectly retaining these constituents at this stage of the evaluation is considered acceptable.

Dust Generation Rate

Studies show that generation of particulate matter from concrete is possible in high abrasion environments, such as roadways exposed to studded tires. However, the evaluation did not identify any studies that evaluated the rate at which concrete dust is generated under these conditions. To address this uncertainty, the current evaluation assumed that 10 percent of the soil was composed of fly ash concrete dust. This value originates from the US EPA (2010a) screening assessment, which evaluated the potential for overland transport of fly ash from uncovered CCR landfills through wind dispersion and runoff. Levels this high are unlikely for encapsulated concrete because the tire studs on passing cars must grind the concrete for dust to be released, compared to granular fly ash, which is available for release at any time. The Washington State Department of Transportation estimated that about 0.25 mm of concrete pavement wear away per one million studded tire vehicle passes, with measured erosion rates between 0.04 and 0.5 mm/yr (WSDOT, 2010). Furthermore, the contributing area of even a moderate-sized CCR landfill is much greater than that of even the most heavily traveled roadways. Therefore, the assumptions used in the current evaluation are likely to overestimate releases. While the magnitude of this overestimation is unknown, it is anticipated to be considerable.

Bioavailability

Bioavailability is the fraction of the total contaminant mass that is available to interact with and potentially cause harm to a receptor's body. The remaining fraction of the compound that is not bioavailable will pass through or over the body with no effect to the receptor. A host of different factors, such as the pH, particle size, moisture, and redox potential of the environment influence bioavailability. In addition, the receptor's age, sex, nutritional state, and physiological state affect bioavailability (US EPA, 2007). Given the numerous variables involved in determining bioavailability, this remains a source of uncertainty. The current evaluation assumed that the total mass of each COPC ingested was entirely bioavailable. This assumption can only overestimate exposure. The magnitude of this overestimation will vary on a case-by-case basis, depending both on the characteristics of the contaminated media and the individual receptor.

Roadway Composition

The current evaluation assumes that the uppermost layer of the roadways that is exposed to high abrasion is composed of only concrete. In reality, many roads are either entirely composed of or overlain by other materials, such as asphalt. Therefore, as Zubek et al. (2004) points out, generated dust may contain no concrete at all. While the assumption of an exposed concrete surface used in the current evaluation may represent high-end exposure scenario, it likely overestimates typical exposures. The magnitude of this overestimation is unknown.

Receptor Habitats near Roadways

It is unknown to what extent receptors may be exposed to any dust that accumulates near roadways and other concrete surfaces subjected to high levels of abrasion. Both child and adult residents are unlikely to spend extended periods of time alongside major roadways. Even under the theoretical scenario that an abandoned highway is converted to residential property, the construction would disturb the surface soil and either dilute the concrete dust present with subsurface soils or remove it entirely. As a result, the high concentrations evaluated likely overestimate exposures. However, the magnitude of this overestimation is unknown.

Wildlife is more likely to spend appreciable time near roadways. The level of exposure depends on a number of factors, including the foraging range of the species, the quality of available food sources, and the season, as well as intra-species and inter-species competition. Furthermore, fragmented land near a roadway is often poor habitat for permanent ecological populations. Consequently, this uncertainty is likely to overestimate exposures. However, the magnitude of this overestimation is unknown.

5.2.2 Uncertainties for Ground and Surface Water Exposures

The uncertainties addressed in this subsection pertain to the evaluation of leachate released from fly ash concrete during use, the transport to ground and surface waters, and the resulting receptor exposures. This document does not discuss uncertainties introduced through the use of the Industrial Waste Evaluation Model (IWEM). IWEM is a peer-reviewed model, and *IWEM Technical Documentation and User's Guide* (US EPA, 2010d) discusses the various uncertainties associated with this model.

Selection of COPCs

The current evaluation selected COPCs based on the findings of the 2010 *CCR Risk Assessment* (US EPA, 2010a). This risk assessment aggregated modeling results for all types of CCRs when drawing conclusions. It is possible that reliance on these findings to select COPCs may result in retaining COPCs related to CCRs other than fly ash. Conversely, it is possible that consideration of data on CCRs other than fly ash may dilute modeling results enough to eliminate COPCs that would have otherwise been retained. However, Kosson et al. (2013) demonstrated that fly ash concrete leach at rates lower than pure fly ash. As a result, any constituents that may have been screened out in US EPA (2010a) as a result of the dilution of fly ash data are likely to have also been screened out in the current evaluation due to the lower leaching rates. Therefore, the magnitude of this uncertainty is considered small.

Available Data

Based on the available studies of the leaching from fly ash alone and fly ash in cement materials, the leaching of COPCs from fly ash was not increased by incorporation of the fly ash in cement materials (US EPA, 2012b; Kosson et al., 2013). Therefore, the current evaluation assumed that, if the leaching behavior of fly ash was adequately captured in the available samples, so was the contribution of fly ash to leaching from fly ash concrete. Kosson et al. (2013) analyzed the leaching behavior of the pure fly ash prior to mixing in concrete. One or more of the fly ashes used in this study exhibited arsenic, antimony, boron, chromium, lead, and molybdenum leaching close to the upper bounds identified as part of a broader sampling effort for US EPA (2009a). However, these fly ashes exhibited cadmium, selenium, and thallium leaching closer to the median identified in US EPA (2009a). Therefore, the current evaluation may underestimate high-end leaching of these three COPCs from fly ash concrete.

The fly ashes used in Kosson et al. (2013) and Garrabrants et al. (2013) are from the subset of ashes known to meet the specifications for use in concrete. It is unknown whether this subset has the same upper bounds of cadmium, selenium, and thallium leaching as the fly ashes in US EPA (2009a). Yet, even if these higher leaching fly ashes are appropriate for use in concrete, the potential increase in fly ash concrete leachate concentrations is unlikely to alter the conclusions of the evaluation. The current evaluation identified cadmium, selenium, and thallium as COPCs based on the findings of the 2010 *CCR Risk Assessment* (US EPA, 2010a), which identified the 90th percentile hazard quotients of 5.0 for cadmium (ecological sediment exposure), 2.0 for selenium (ecological surface water exposure), and 3.0 for thallium (human ground water exposure). Kosson et al. (2013) showed that leaching from fly ash concrete was consistently lower than leaching from pure fly ash by at least a factor of two for selenium and an order of magnitude for both cadmium and thallium. The magnitude of these decreases alone would be sufficient to eliminate these constituents as COPCs in US EPA (2010a). Furthermore, the LEAF Method 1313 data used in Kosson et al. (2013) represent saturated leachate from ground up concrete. Consideration of time-dependent leaching of intact concrete would further reduce leachate concentrations. Based on these different lines of evidence, the magnitude of this uncertainty is likely to be small.

Non-Detect Values

The current evaluation eliminated arsenic, cadmium, lead, molybdenum, and thallium as COPCs at Step 2 (Comparison of Available Data) because Garrabrants et al. (2013) did not detect these COPCs in leachate collected from fly ash concrete or portland cement concrete with EPA Method 1315. Based on the available data, the evaluation concluded that the rate at which these COPCs leach from fly ash concrete is comparable to that of portland cement concrete. These constituents may still be present in the samples at concentrations below the MDL. However, they are unlikely to be present near the MDL. The concrete matrix is known to become denser as it cures (Garboczi, 1995). As the internal pore spaces decrease in size, the rate at which COPCs can leach out decreases. Selenium provides the best example for this: it was not detected in any of the leachate from three month cure samples, but this evaluation retained selenium for further consideration because this COPC was detected in leachate from a few samples of fly ash concrete cured for only 28 days. Therefore, the other COPCs that were not detected even at 28 days are likely to be even lower after a three month cure time. In addition, Garrabrants et al. (2013) did not detect arsenic, cadmium, lead, molybdenum, and thallium during the leaching time step

of 14 days. Therefore, the cumulative mass leached for a duration of fewer than 2 days will be even further below the MDL. The potential for these constituents to be released at higher levels from fly ash concrete than from portland cement concrete remains. However, the incremental increase in exposures at these already low levels is unlikely to result in appreciable risk to downgradient receptors. Therefore, the impact of this uncertainty on the findings of the current evaluation is likely to be minimal.

Incremental Releases

The current evaluation does not consider the incremental fly ash contribution to releases from concrete. Leaching is not necessarily an additive process, and the available dataset makes it difficult to empirically parse out the exact contribution of fly ash to releases from concrete. Therefore, to remain conservative, the evaluation used total releases from fly ash concrete to estimate potential exposures. As a result, the COPC releases that result from incorporation of fly ash into concrete may be lower than those presented in Step 4 (Screening Assessment). This assumption will likely overestimate the COPC exposures associated with the beneficial use of fly ash in concrete. In particular, chromium was shown in Step 2 to be present in and released from portland cement concrete at levels approaching those of fly ash concrete. The magnitude of this overestimation is not known.

Chromium Speciation

Chromium was the one COPC in this evaluation that warranted fate and transport modeling. The speciation of chromium is an important consideration because its toxicity is dependent on speciation. The evaluation did not identify any studies evaluating the speciation of constituents leaching from concrete. Even if these data were available, receptors are unlikely to ingest leachate directly from the concrete. Instead, the leachate will migrate through the soil and ground water table prior to contact with any receptors. During transport, the leachate will change in pH and be exposed to a different set of redox conditions, which can dramatically change speciation. Therefore, even when one species leaches from concrete, the amount released and the amount that reaches the receptor may not be the same. Because of the high uncertainty surrounding the exact speciation of chromium at the point of exposure, the evaluation conducted fate and transport modeling under the assumption that chromium was released in the form most mobile in the environment. Therefore, this assumption may overestimate potential exposures. However, the magnitude of this overestimation is unknown.

Roadway Dimensions

The width and thickness of concrete roadways used to model concrete leaching in the current evaluation are based on the upper bound of specifications from the American Association of State Highway and Transportation Officials (AASHTO, 1993; 2004). The use of the upper bound values for these variables likely overestimates potential exposures. The length of the concrete roadways is uncertain. The needs of the individual project, rather than best design practices define this variable. However, the exact length of the concrete is unlikely to have an impact on the current evaluation of HEIs. The evaluation assumes that human and ecological receptors are located midway along the length of the concrete source where they are exposed to the highest ground water concentrations. Extending the ends of the concrete out even farther will not change the concentration at the centerline of the plume.

Therefore, the assumed roadway dimensions may overestimate releases and subsequent exposures. However, the magnitude of this overestimation is unknown.

Infiltration Rate

Water movement through cracked concrete is a complicated phenomenon that is difficult to model accurately. The actual amount of water that can pass through concrete is determined by a number of variables, including, but not limited to, the size, number, and orientation of cracks present on the road surface (Apul et al., 2002). To avoid compounding uncertainty by specifying ranges for each of these variables, the evaluation only considered a high-end scenario where infiltration through concrete was not a limiting factor. In this scenario, the amount of water that can pass into the ground water table is bounded by the infiltration rate of the soil underlying the concrete road, base, and sub-base. In practice, the use of drainage pipes beneath the roadway may reduce the amount of infiltrating water that can reach the ground water table. However, not all roadways include this design feature. Furthermore, without proper maintenance, these pipes can become clogged and ineffective at rerouting water. Therefore, the evaluation does not account for the potential effects of drainage pipes on infiltration rates in this evaluation. These assumptions likely overestimate releases and subsequent exposures, but the magnitude of this overestimation is unknown.

Smaller cracks may exist in the concrete matrix that can retain infiltrating water for longer durations than those modeled, and soils may retain water in contact with the concrete for some time after a precipitation event has ended. Both scenarios may allow greater mobilization and accumulation of COPCs than considered in this evaluation. However, in both cases, capillary forces will act against gravity and impede the flow of water, resulting in much of the water evaporating before it can migrate to the subsurface. Instead, the current evaluation focuses on high infiltration rates through the concrete. The much larger volume of water assumed to pass through these larger and more numerous cracks, together with the high-end COPC concentrations used in the evaluation, will result in a higher mass flux of COPCs into the subsurface environment. Even if a smaller crack size exists that could result in higher mass fluxes than those considered in this evaluation, these cracks will propagate and expand, becoming larger with time. Thus, small cracks are less representative of a long-term, high-end leaching scenario. These assumptions likely overestimate releases and subsequent exposures, but the magnitude of this overestimation is unknown.

Leaching Evaluation Assessment Framework Data

The current evaluation used the LEAF Method 1313 data from Kosson et al. (2013) to bound the COPC concentrations that can be present in leachate. This sampling method requires the sample to be ground up prior to leaching. Grinding a material increases the surface area available for contact with the leachant. The higher surface area exposed to the leachant results in higher leachate concentrations for those COPCs than are likely to occur during use. The evaluation used LEAF Method 1315 data from Garrabrants et al. (2013) to calculate COPC concentrations in leachate passing over the concrete surface. This sampling method retains the sample in monolith form. The concrete monolith is submerged in a tank containing deionized water for a set time, and then is transferred to another tank with fresh water. Then, the resulting leachate in each tank is measured. Although the leachate from this method is more representative of water exposed to an encapsulated material, the contact time between the water and

concrete is much higher than would generally occur during a precipitation event. Therefore, the current application of both Method 1313 and 1315 data is likely to overestimate COPC releases, but the magnitude of this overestimation is unknown.

The *Cementitious Materials Report* (US EPA, 2012b) compared the pH-dependent leachate data with single pH data collected in other studies (Cheng et al., 2008 and Zhang et al., 2001) to determine whether the pH-dependent data had accurately captured the range of potential leaching behaviors. The single pH leaching tests were conducted with either the Synthetic Precipitation Leaching Procedure (SPLP, EPA Method 1312) or the Standard Test Method for Shake Extraction of Solid Waste with Water (ASTM D3987-85). The report did not consider data from the Toxicity Characteristic Leaching Procedure (TCLP, EPA Method 1311) because the pH of the leaching solution was below the lowest theoretical pore water pH of 7.0 for concrete. The comparison showed that the single extraction leaching tests produced results that were generally consistent with the pH-dependent leaching tests at the same pH. However, single extraction leaching tests do not provide an indication of the changes in material leaching with changes in pH that occur as a consequence of material aging. The few single extraction point measurements that deviated from this trend exhibited lower leachate concentrations. These results may be a consequence of partial carbonation that occurred during preparation and testing of the ground up samples (Garrabrants et al., 2004). Based on these results, US EPA (2012b) concluded that the LEAF leachate data agree well with other single-pH leachate tests. Therefore, reliance on LEAF data, rather than single-pH data, is likely to reduce the amount of uncertainty in the evaluation.

Additional Leach Test Parameters

The current evaluation of leaching relies on data from Kosson et al. (2013) and Garrabrants et al. (2013) to evaluate leaching from fly ash concrete. The primary focus of these studies was to evaluate the effects of pH, liquid to solid ratio, and physical form (e.g., ground, monolithic) on leaching from concrete. All of these parameters are known to have a major impact on the leaching of inorganic COPCs. There may be other factors, such as light and heavy fractionation; mineral phases; trace metal speciation; solution composition; and background electrolyte and ionic strength, which influence measured concentrations to some degree. However, these factors are not anticipated to be major sources of uncertainty in the current evaluation because:

- The current leaching evaluation is based on use of the entire material that was subjected to leach testing, rather than some fraction of the material. Therefore, no distinction between light and heavy fractionation needed to be made for this evaluation.
- Because the COPCs identified in this evaluation are trace inorganics for which mineral phases are below typical instrument detection limits, COPC concentrations were measured directly through leaching tests rather than inferred from major mineral determinations.
- The speciation of certain COPCs, such as chromium, in leachate was considered as part of previous evaluations of CCR leaching (US EPA, 2009c). The potential effects of reducing conditions on COPC speciation in concrete leachate were not considered. However, this is not anticipated to be a source of uncertainty, as oxidizing conditions are anticipated to be prevalent around the uses of concrete discussed in this document. It is important to note that this is distinct from reducing

conditions that may occur in subsurface soils and ground water, which are addressed through fate and transport modeling.

- Other parameters were considered through previous evaluations of fly ash leaching because of the potential to affect COPC leaching on a case-by-case basis. However, none of these parameters had a consistently strong impact on leaching. Some of these less significant parameters were discussed in US EPA (2009c). Leachate concentrations for a number of minor analytes in the samples tested [e.g., dissolved organic and inorganic carbon, conductivity (which can be converted to ionic strength), copper, iron, manganese, nickel, silicon, sodium, zinc, and others] were evaluated, but not presented in US EPA (2009c). These data are available from the authors, and are included in Leach XS Lite.[11]

A more extensive analysis of specific fly ash concrete samples may reveal individual cases where one of these additional parameters is important. However, for an evaluation that is intended to estimate a national bounding of releases from fly ash concrete, it is believed that the current focus on the parameters known to consistently have the greatest effect on leaching remains appropriate. Therefore, while the data used in this evaluation may over- or underestimate concrete leaching on a case-by-case basis, the overall magnitude of these uncertainties is expected to be small.

Concrete Aging

As concrete ages, physical and chemical changes that occur in the concrete matrix may alter the rate at which the concrete releases COPCs. Carbonation is the primary mechanism that drives concrete aging. Carbonation is the reaction of carbon dioxide [CO_2] with the various alkaline constituents in the concrete matrix. The most important of these reactions is with calcium hydroxide [$Ca(OH)_2$], which ultimately generates calcium carbonate [$CaCO_3$]. This shift from calcium hydroxide to calcium carbonate can alter leaching in several ways:

- The first way that aging may alter leaching is through changes to the concrete pore water pH. An initial pH of roughly 12.4 is common for newly-poured concrete based on the dissolution chemistry of calcium hydroxide. Complete carbonation of the concrete matrix may result in a pH as low as 7.0, based on the dissolution chemistry of calcium carbonate (Garrabrants et al., 2004). Changes in this pH will alter the leaching behavior of constituents with pH-dependent solubility.
- The second way that aging may alter leaching is through changes to the composition of the concrete. Carbonation of certain minerals in the concrete matrix may result in desorption of COPCs that otherwise would have remained bound within the concrete matrix (Garrabrants et al., 2004). Studies have shown that this desorption acts in concert with changing pH to alter leaching rates. Müllauer et al. (2012) demonstrated higher cumulative leaching of chromium from highly-carbonated ground concrete. Sanchez et al. (2002) demonstrated higher cumulative leaching of arsenic, but lower cumulative leaching of lead, from highly-carbonated ground concrete.

[11] Leach XS Lite is a tool that allows users to evaluate and characterize the release of constituents based on comparisons derived from leaching test results for a wide range of materials and waste types This tool is available on-line at: http://www.vanderbilt.edu/leaching/downloads/leachxs-lite/

- The final way that aging may alter leaching is through physical changes to the porosity of the concrete matrix. Van Gerven et al. (2006) compared the total porosity of concrete samples carbonated for 60 days with total porosity of similar samples with relatively little carbonation. The results showed a 12 percent reduction in the porosity of the carbonated samples, compared to the relatively uncarbonated samples. Van Gerven (2006) noted that this decrease in porosity resulted in decreased leaching for sodium and potassium. Sanchez et al. (2002) also noted that the permeability of an intact concrete matrix decreased with carbonation and, as a result, the retention of arsenic, chromium, and lead all increased (Lange, 1996 as cited in Sanchez et al., 2002).

The findings of Van Gerven (2006) and Lange (1996) (as cited in Sanchez et al., 2002) contrast with those of Sanchez et al. (2002), Garrabrants et al. (2004), and Müllauer et al. (2012), but do not contradict them. The latter three studies evaluated samples of ground concrete, which eliminated the physical concrete matrix and found higher leaching with increased carbonation, while Van Gerven et al. (2006) evaluated samples of intact concrete and found lower leaching with increased carbonation. Grounding concrete allows samples to become highly and uniformly carbonated by breaking down the dense concrete matrix. However, it is unlikely that such a high degree of carbonation will occur during the useful life of most intact concretes. This results in uncertainty regarding the ultimate effects of carbonation on leaching behavior of carbonated concrete, and how such effects impact releases of inorganic constituents. As a result, the current evaluation may underestimate or overestimate long-term leaching on a case-by-case basis, but the magnitude of this misestimate is unknown.

Fish Bioconcentration Factors

US EPA (2003a) recommends the use of bioconcentration factors (BCFs) to assess exposure to inorganic metals. Therefore, this evaluation calculated the potential exposure from fish ingestion using BCFs for trophic level 3 (TL3) and trophic level 4 (TL4) fish (i.e., fish at the higher levels of the food chain where bioconcentration is greater) to estimate the transfer of pollutants from environmental media into fish. In the current evaluation, the evaluation used only BCFs from laboratory or field studies of TL3 and TL4 fish, rather than values estimated from physical or chemical properties [e.g., octanol-water partition coefficient (K_{ow})]. Aquatic BCFs are developed by dividing measured concentrations in aquatic biota by total surface water concentrations. There are several sources of uncertainty associated with the models used to estimate BCFs for aquatic biota.

One source of uncertainty is experimental error that may affect the true value of the BCF. Error may originate from the relatively short exposure timeframe used in a study compared to the lifetime of exposure in the field; some laboratory BCF studies may not have attained steady-state concentrations in the fish due to short exposure durations (Arnot and Gobas, 2006; and CalEPA, 2012). Other sources of error may be laboratory-prepared water with concentrations that are not representative of field conditions (e.g., constant and unrealistically high concentrations), the use of radio-labeled compounds without adequate correction for the parent signal, or the use of a less precise analytical method when analyzing samples (Arnot and Gobas, 2006; and CalEPA, 2012). These laboratory errors may bias the resulting measurements to be either high or low; the overall magnitude of this uncertainty is unknown.

Another source of uncertainty stems from the method used to calculate BCFs. BCFs are based on whole body concentrations for fish. However, the amount of bioconcentration often varies among

different organs and muscle tissues in the fish (du Preez et al., 1993). For example, fish muscle tissues often have the lowest accumulation of metals. Because muscle is the bulk of the edible part of the fish, a whole body concentration that includes other organs (with higher concentrations of contaminants) may overstate potential human exposures from fish ingestion (CalEPA, 2012). In addition, BCFs may be measured from juvenile fish. Studies have shown an inverse relationship between metal accumulation and weight or size of the fish, with the metal concentrations in tissues decreasing as fish size or weight increases (Liao et al., 2003, as cited by CalEPA, 2012). Scientists have attributed this effect to a number of factors, including growth dilution, increased metabolic rate in juvenile fish, and increased ability to depurate the metals as the fish matures. As a result, metal uptake studies in fingerlings or juvenile fish may overestimate BCFs in mature fish (CalEPA, 2012). Finally, BCFs are based only on the bioconcentration of metals from environmental media. Ingestion of sediment and sediment-dwelling invertebrates by bottom-dwelling fish species may also contribute to metal uptake by these fish (CalEPA, 2012). Therefore, reliance on only BCFs to estimate fish exposure may overestimate or underestimate the actual accumulation of a metal in fish. However, the overall magnitude of this uncertainty is unknown.

The final source of uncertainty is the application of the BCFs. In most cases, the evaluation used BCF data for a single fish species to represent all fish at that trophic level. This may or may not accurately represent the fish species commonly caught and consumed, introducing some uncertainty into the analysis. Overall, it is unknown whether use of BCFs overestimates or underestimates concentrations present in fish tissue.

5.2.3 Uncertainties for Air Exposures

The uncertainties addressed in this section pertain to the evaluation of mercury releases from fly ash concrete and FGD gypsum wallboard during consumer use, and the resulting receptor exposure.

Linear Increase of Mercury Emanation

The current evaluation assumes that the rate at which fly ash concrete and FGD gypsum wallboard emit mercury is a linear function of the amount of mercury present in the CCR products. Vaporized mercury moves through porous solids (e.g., concrete and wallboard) by diffusion, which is controlled by the physical characteristics of the material (e.g., porosity) and ambient environmental conditions (e.g., temperature and pressure). These physical characteristics and environmental conditions may change with time, independent of the mercury concentration present, and may have non-linear impacts on the rate of mercury diffusion. However, the available literature identifies emanation rates measured under high temperatures, negative pressure conditions, and short cure times (i.e., higher concrete porosity). Therefore, the assumption that these extreme physical characteristics and environmental conditions remain constant likely overestimates mercury releases under typical room conditions.

Elemental mercury is the form of mercury that is available to vaporize from concrete and wallboard. The remainder of the mercury is generally present in various oxidized compounds (e.g., mercuric chloride) that do not vaporize under standard environmental conditions. A linear increase in mercury emanation assumes that the ratio of elemental mercury to total mercury remains constant with increasing mercury concentrations. Elemental mercury is difficult to capture during coal combustion and is more

likely to escape into the atmosphere, while oxidized mercury is more water soluble and is more effectively captured by pollution control devices (Wilcox et al., 2012). As a result, the ratio of elemental and oxidized mercury present in fly ash is likely to decrease as total mercury concentrations increase. The presence of higher carbon content does not alter this fact. Most of the retention and oxidation of mercury associated with fly ash involves carbon content. Studies have shown that organic carbon with the highest oxidation capacity results in some of the highest mercury concentrations in fly ash (Abad-Valle et al., 2011). Therefore, the assumption of a constant ratio of elemental and oxidized mercury in fly ashes and FGD gypsum when extrapolating emanation rates will likely overestimate releases. This agrees with the finding of Golightly et al. (2009) that fly ashes with higher organic carbon content result in fly ash concrete with lower mercury emanation rates relative to total mercury content.

Air Exchange Rates

Air exchange rates can vary considerably based on geography due to the different climates across the United States, and based on the season due to different building heating and cooling requirements. When conducting the survey in 1995 that forms the basis for these air exchange rates, the authors examined air exchange rates in homes across the country during all four seasons. However, in recent years, attention has been focused on reducing heat loss and, consequently, air loss from buildings. Several building construction codes currently require a minimal air exchange rate of 0.35 ACH for newly constructed habitable structures.[12] However, the actual air exchange rate of a completed building is rarely measured and is still subject to seasonal changes (US EPA, 2010b). Therefore, the measured air exchange rates presented in Koontz and Rector (1995) still represent the best available estimates of average and high-end exposures for the country. Use of this data may overestimate exposures, but the magnitude of this overestimation is unknown.

Organic Carbon Content

Golightly et al. (2009) found that fly ash with higher organic carbon content results in fly ash concrete with lower mercury emanation rates relative to total mercury content. As a result, the linear increase of mercury emanation assumed in this evaluation causes the fly ash concrete with the lowest organic content to have the highest adjusted emanation rates. Yet, a low organic content fly ash is unlikely to contain the highest mercury concentrations, because a strong positive correlation exists between the mercury concentration and organic content of fly ash (Wilcox et al., 2012 and Abad-Valle et al., 2011). Data on the organic carbon content was not available for the majority of fly ash samples. Therefore, to remain conservative, all available samples were incorporated into the evaluation. As a result, the fly ashes with the highest mercury concentrations likely represent organic content higher than the limit of 6 percent for concrete production (Golightly et al., 2009). Extrapolating mercury emanation from fly ash concrete using an upper bound mercury concentration that does not consider the limit on fly ash organic carbon content is likely to overestimate exposures, but the magnitude of this overestimation is unknown.

[12] Examples of these building codes include American Society of Heating, Refrigerating, and Air-Conditioning Engineers Standard 90.1 (ASHRAE, 2007), International Residential Code (ICC, 2006a), and International Mechanical Code (ICC, 2009).

FGD gypsum is composed predominantly of calcium sulfate. Fly ash is considered a contaminant in FGD gypsum because it decreases the quality of the wallboard produced. As a result, industry standards are in place limiting the amount of fly ash allowed in raw FGD gypsum intended for wallboard production (Henkels and Gaynor, 1996). However, the trace amounts of fly ash that are present may contribute to mercury emanation from the FGD gypsum wallboard. The FGD gypsum samples available did not report fly ash content. However, the examination of numerous FGD gypsum and FGD gypsum wallboard samples produced across the United States provides a high degree of confidence that the evaluation captured the range of fly ash and mercury concentrations.

Chemical Admixtures in Fly Ash Concrete

The majority of fly ash concrete evaluated in the literature consists of cement, fly ash, water, and some form of aggregate. However, Golightly et al. (2007; 2009) also included an air entrainment admixture (AEA). AEAs are chemicals added to concrete to control the size and spacing of air pockets (i.e., voids) within the concrete matrix. These voids connect smaller capillary pores, which are the spaces within the concrete matrix filled with unreacted water. The primary purpose of introducing air voids in the concrete matrix is to provide the unreacted water present in capillary pores a place to expand when exposed to freezing temperatures. This reduces the amount of strain placed on the concrete matrix and reduces internal cracking. Once the ice thaws, the water in the voids is drawn back into the narrower pore network through capillary forces. Although AEAs alter the number and spacing of air voids, they do not directly impact the size and spacing of capillary pores and smaller gel pores. This network of capillary and gel pores throughout the concrete matrix serves as the primary pathway through which gases can diffuse through the concrete and into indoor air. This evaluation found no indication that inclusion of AEAs in concrete will reduce the rate of mercury emanation from concretes during use. Instead, AEAs may actually promote transport of gases through the concrete by providing a direct connection for the capillary pores within the concrete and increasing the volume of internal air spaces. This greater air space and interconnectivity could allow easier migration of mercury through the concrete, resulting in an overestimation of releases from concrete without chemical additives.

Surface Covering

Both concrete and wallboard are typically coated by some combination of paint, glue, carpet, laminate, or other substance prior to use. Concrete surfaces that remain uncovered are often polished to a smooth finish. All of these coatings will impede the migration of mercury into the indoor air by either directly covering interstitial pore spaces or minimizing the surface area that can emit mercury. However, it is unknown to what extent different combinations of surface coatings inhibit mercury emanation rates. All available literature on mercury emanation from fly ash concrete and FGD gypsum wallboard measured releases from uncoated products. Consistent with the available data, the current evaluation assumed that the building materials were placed bare in a building. The assumption of no surface covering prior to use overestimates the calculated exposures, but the magnitude of the overestimation is unknown.

Combined Use of Fly Ash Concrete and FGD Gypsum Wallboard

The current evaluation did not consider a scenario where fly ash concrete and FGD gypsum wallboard were used in the same residence. While it is possible that both CCR products could be used in the same building, wallboard and concrete are unlikely to be used in the same wall. Because FGD gypsum wallboard emits mercury at a lower rate than concrete, even after adjustment, use of wallboard would only reduce the mercury concentrations estimated for a building with all floors, ceilings, and walls made with concrete. Therefore, this uncertainty is small and unlikely to affect the results of the evaluation.

Sorption of Mercury to Building Surfaces

Mercury vapor has the potential to sorb to various surfaces within a building. This sorption will reduce the concentration of mercury present in the indoor air, but it will introduce an additional potential exposure pathway. The sorption of mercury onto indoor surfaces may result in higher exposures than those predicted through consideration of indoor air concentrations alone as a result of hand to mouth contact with dust and with various household surfaces. However, the calculated 90^{th} percentile raw fly ash mercury concentration of 0.65 mg/kg is over an order of magnitude below the incidental ingestion screening benchmark of 14.4 mg/kg recalculated for an infant. Therefore, the impact of this uncertainty on the findings of the current evaluation is likely to be minimal.

Steady-State Mercury Concentration

"Steady state" is the condition of equilibrium when the rates of mercury entering and leaving a building are equal and the resulting mercury concentration inside the building is constant. The current evaluation makes the conservative assumption that mercury concentrations in the air are at a steady state for the duration of receptor exposure. In reality, mercury concentrations will fluctuate with time. The opening and closing of doors and windows will alter the air exchange rate of the building, disrupt steady-state conditions by allowing mercury to leave the building faster than it enters, and reduce mercury concentrations in the air. On the whole, the assumption of steady state will overestimate exposures, but the magnitude of the overestimation is unknown.

Complete Mixing of Indoor Air

The current evaluation conducted the evaluation of mercury emanation under the simplifying assumption that indoor air is completely mixed. The 1997 *Exposure Factors Handbook* (US EPA, 1997b) supports this assumption with the following statement:

> *"…for an instantaneous release from a point source in a room, fairly complete mixing is achieved within 10 minutes when convective flow is induced by solar radiation. However, up to 100 minutes may be required for complete mixing under quiescent (nearly isothermal) conditions."*

The studies relied upon in the 1997 *Exposure Factors Handbook* (US EPA, 1997b) used a low air exchange rates of less than 0.1 hr^{-1}. This is lower than the conservative 10^{th} percentile estimate of 0.18 hr^{-1}, and much lower than the central tendency estimate of 0.45 hr^{-1} recommended by US EPA (1997b). Thus, these studies seem to support the assumption of complete mixing. Other studies summarized by

US EPA (1997b) were conducted at more typical room ventilation rates, which found that "the ratio of source-proximate to slightly-removed concentration was on the order of 2:1." This indicates that mercury concentrations may increase directly adjacent to the ceiling, wall, or floor that emits mercury vapor. However, most building occupants are not stationary and move around a building throughout the day. Therefore, while the assumption of complete mixing may underestimate some short-term exposures, it is a reasonable representation of typical long-term, chronic exposures.

Room Temperature

Temperature may alter the rate of mercury emanation from building materials. Mercury undergoes some volatilization at room temperature; higher temperatures can increase the kinetic energy of gases and may result in faster migration through the concrete matrix. The American Society of Heating, Refrigerating and Air Conditioning Engineers defines comfortable room temperature as being between 20 and 26ºC (68ºF and 79ºF), depending on the season (ASHRAE Standard 55). This range represents the temperatures that provide thermal comfort for approximately 80 percent of the population. Therefore, the majority of habitable buildings will have temperatures somewhere within this range. At present, the evaluation did not identify any research directly evaluating the effects of temperature on mercury emanation from concrete. Therefore, this remains an uncertainty in the current evaluation.

The available studies on fly ash concrete (Golightly et al., 2006; 2009) evaluated mercury emanation at a constant ambient temperature of 40 ºC (104 ºF). A temperature of 40 ºC represents an upper bound of realistic outdoor temperatures in most of the United States. However, the building materials placed indoors will be cooled by shading, air conditioning, air flow, and other factors that likely prevent indoor temperatures from reaching levels this high. Based on these findings, the temperatures present in these studies likely overestimate mercury releases from fly ash concrete, but the magnitude of the overestimation is unknown.

The one available study on FGD gypsum wallboard (Shock et al., 2009) evaluated mercury emanation at a temperature between 22 and 24 ºC (71.6 and 75.2 ºF). These data represent a range of comfortable room temperatures. Although higher temperatures than these are possible, it is considered unlikely that they would occur simultaneously with low-end air turnover rates. Use of open windows, fans, or other means to cool the area would increase the air turnover rate. Based on these findings, the temperatures present in this study provide an estimate of typical mercury releases from FGD gypsum wallboard. The potential for higher indoor air temperatures may result in an underestimation of the upper bound of mercury releases. However, the impact of this underestimation on resulting exposure estimates is likely to be small.

Pressure Gradient

The presence of a negative pressure gradient in a building may alter the rate of mercury emanation from building materials. Several factors can generate negative pressure gradients, including: temperature differences between the indoors and outdoors, changes in the outdoor wind or barometric pressure, and operation of mechanical fans or vents. Studies have shown that the measured negative pressures in homes range between 1 and 50 pascals [0.00015 and 0.007 pounds per square inch (psi) (MassDEP, 1995)]. It is well known that the presence of a pressure gradient can greatly influence the movement of

gases into a building through cracks in building walls and foundations (US EPA, 2008a). However, the evaluation did not identify any research that evaluates the degree to which change in pressure affects mercury emanation from intact concrete. Therefore, this remains an uncertainty in the current evaluation.

Of the available literature, only Golightly et al. (2009) reported the negative pressure applied during sample collection. However, based on the similar experimental setups used among the three available studies, the evaluation assumes that the negative pressures generated are on the same order of magnitude. Golightly et al. (2009) maintained a constant pressure drop of 3 psi across the concrete throughout the experiment with a mechanical fan that pulled air across the concrete and through the air sampler. This negative pressure is over two orders of magnitude higher than the pressure commonly found in buildings. Based on these findings, the pressure gradients present in these studies will overestimate mercury releases. However, the magnitude of this overestimation is unknown.

Relative Humidity

The relative humidity may alter the rate of mercury emanation from concrete. As the relative humidity in a home decreases, the rate of movement of water from the concrete matrix may begin to increase. Lower water retention within the concrete matrix may act to remove barriers to the transport of mercury vapor. The evaluation did not identify any research directly evaluating the effects of indoor ambient humidity on mercury emanation from concrete.

Shock et al. (2009) reported the relative humidity during collection of air samples near wallboard. The relative humidity ranged between 25 percent and 40 percent during sample collection. The two studies on concrete did not report the relative humidity (Golightly et al., 2005; 2009). However, because no reported attempts were made to alter the humidity level in the buildings, the evaluation assumes that the relative humidities fall somewhere in the standard indoor range of 30 percent to 60 percent (US EPA, 2010e). Furthermore, because of the higher air temperature maintained around the concrete during these evaluations, the relative humidity in the sampling containers in which the concrete was enclosed would be lower than that of the surrounding room. Based on these findings, the relative humidities present in these studies may over- or underestimate the amount of mercury released from concrete and wallboard on a case-by-case basis, but the magnitude of the effect on the results of the evaluation is likely to be small.

5.2.4 General Uncertainties

This section discusses the uncertainties that are present throughout the evaluation and affect multiple exposure pathways.

Pollution Control Technologies

The current evaluation contains information on fly ash from coal-fired power plants that use air pollution control technologies necessary to span the range of coal types and established air pollution control technology configurations addressed in US EPA (2010a). In recent years, new pollution control technologies, such as those with activated carbon or halogen additions, have been proposed and implemented in response to new regulations on coal combustion facilities. The intent of these pollution

control technologies is to capture mercury and other pollutants from flue gases before they can escape into the atmosphere. Limited data are available in the CCR Constituent Database to evaluate the effects of these pollution control technologies on COPC releases. Because of the limited available data, this evaluation draws no conclusions about the beneficial use of CCRs generated with these or any other future pollution control technologies.

Hexavalent Chromium Mode of Action

At present, the US EPA Integrated Risk Information System (IRIS) program is in the process of developing an oral slope factor for ingested hexavalent chromium [chromium (VI)]. In the absence of a finalized Tier I or Tier II toxicity value, as described in *Office of Solid Waste and Emergency Response 2003 Directive 9285.7-33* (US EPA, 2003b), the evaluation relied upon the Tier III value finalized by the State of New Jersey Department of Environmental Protection (NJDEP) (NJDEP, 2009; Stern 2010). The NJDEP drew no conclusions about the mode of action for chromium (VI). Based on existing laboratory research, there is evidence that chromium (VI) may have a mutagenic mode of action (NTP, 2012; McCarroll et al., 2010). On the other hand, there is laboratory research indicating that the mutagenic mode of action occurs only at high exposures associated with cell death and is, therefore, not relevant to human environmental exposures (Stern, 2010). At present, neither NJDEP nor IRIS has finalized a decision on the relevance of a mutagenic mode of action for human ingestion of chromium (VI). Therefore, the lack of consensus on the interpretation of the current state of the science adds uncertainty to the quantitative estimation of the oral cancer risk from ingestion of chromium (VI). If the evaluation were to quantify a mutagenic mode of action, it would result in lower (i.e., more stringent) screening benchmarks than those currently used for chromium (VI).

Human Exposure Factors

Exposure modeling relies heavily on data pertaining to population activity patterns, mobility, dietary habits, body weight, and other factors. The physical characteristics, activities, and behavior of individual receptors can vary considerably. Therefore, the single set of often high-end exposure factors used in the current evaluation is likely to overestimate potential exposures. The magnitude of this overestimation will vary on a case-by-case basis, depending on the characteristics of the individual receptor.

In instances where information on exposure factors was not available for a given receptor, this evaluation used data on similar receptors. For example, this evaluation drew child fish-consumption rates from data on adult recreational anglers. This extrapolation likely overestimates actual exposures, as the amount of food consumed by a small child is anticipated to be less than that consumed by a full-grown adult. In instances where age-specific child exposure factors were not available, this evaluation used available child exposure data for all age cohorts. The current evaluation used soil ingestion rates reported for children between the ages of three and six for all child age cohorts. This extrapolation may under or overestimate exposures on a case-by-case basis.

This evaluation drew exposure factors from the 1997 *Exposure Factors Handbook* (US EPA, 1997b) and the 2008 *Child-Specific Exposure Factors Handbook* (US EPA, 2008b). This evaluation carefully reviewed and evaluated both documents for quality. The evaluation criteria included peer review, reproducibility, pertinence to the United States, adequacy of the data collection period, validity of the

approach, representativeness of the population, characterization of the variability, lack of bias in study design, and measurement error. EPA has also recently released the 2011 *Exposure Factors Handbook* (US EPA, 2011). This document has undergone the same level of peer review as the previous handbooks. However, OSWER is still assessing how best to incorporate the updated recommendations into Agency evaluations. Therefore, this evaluation only drew data from this document where they were not available in other iterations of the *Exposure Factors Handbook*. A review of the relevant exposure factors contained in the 2011 *Exposure Factors Handbook* (US EPA, 2011) found that adult water consumption rates are lower and that adult body weights are higher than those found in the 1997 *Exposure Factors Handbook* (US EPA, 1997b). All other exposure factors relevant to this evaluation remained the same between the 1997 and 2011 editions.

Cumulative Exposures

In the current evaluation, exposures to different COPCs were considered independently. In reality, receptors are exposed to multiple constituents simultaneously. An individual COPC may interact with other constituents present, resulting in synergistic or antagonistic effects that exacerbate or diminish the health impacts predicted by evaluating each COPC independently. For this screening assessment, it is considered inappropriate to consider additive exposures. The calculations in this document represent exposures at or above a reasonable high-end. Due to the natural variability of CCRs and human behavior, it is considered unlikely that a single receptor would be simultaneously exposed to high end concentrations of every COPC. Furthermore, individual receptors are unlikely to be exposed to COPCs from multiple exposure pathways within the same timeframe. COPCs are released and transported to different media at different rates. For example, a receptor may receive ground water from an impacted well and fish from an impacted surface water body; however, the well and water body will not be the same distance away from the contaminant source. As a result, the concentrations in the well and the water body will be different. Based on these considerations, the Agency chose not to consider cumulative exposures in this evaluation. This approach may underestimate or overestimate potential exposures on a case-by-case basis.

Exposure Pathways

In Step 4 (Screening Assessment), this evaluation considered the single pathway for each release that is most likely to result in the highest chronic exposures during use of fly ash concrete and FGD gypsum wallboard. Because these CCR products can be used in a variety of places and in a variety of ways, receptors can be exposed through pathways other than those evaluated in this document. However, consideration of these additional pathways is unlikely to result in a chronic time-averaged exposure any higher than those presented in this document. Some exposures, such as those arising from home renovation, may be higher in the short term, but will quickly decrease with time. The current evaluation assumes that all of the media a receptor encounters are contaminated with high-end COPC concentrations. If a receptor were to at any point leave this theoretical high-exposure environment for another, the receptor's exposures will only decrease. Therefore, while the existence of other exposure scenarios introduces some uncertainty into the evaluation, it is unlikely to affect the results of the evaluation.

5.3 Final Conclusions

The *Methodology for Evaluating Encapsulated Beneficial Uses of Coal Combustion Residuals* (US EPA, 2013a) is a resource to aid states, tribes, local governments, the general public, and the regulated community in evaluating the beneficial use of any encapsulated CCR. The current evaluation applied this methodology to two of the largest encapsulated beneficial uses of CCRs, fly ash used as a direct substitute for portland cement in concrete and FGD gypsum used as a replacement for mined gypsum in wallboard. The conclusions of this evaluation are applicable to the specific conditions considered in this evaluation, such as products conforming to relevant physical and performance standards established by voluntary consensus standard-setting bodies.

All COPCs were eliminated prior to Step 4 (Screening Assessment), or were found to be at or below all relevant regulatory and health-based screening benchmarks identified for this evaluation. Based on these findings, the evaluation did not proceed to Step 5 (Risk Assessment). This evaluation characterized the uncertainties present, and minimized the impact of these uncertainties to the extent practicable by biasing the evaluation in a conservative direction. The review of uncertainties conducted showed that, while the magnitude of the uncertainties is difficult to quantify, it is likely that the conservative direction of the uncertainties causes the evaluation to overestimate potential exposures. As a result, while the exposure concentrations calculated and reported in this document are considered sufficient to draw conclusions regarding the beneficial uses under evaluation, these concentrations should not be cited for purposes outside of the context of this evaluation.

Based on the analysis set forth in this document, the evaluation concludes that environmental releases of COPCs from CCR fly ash concrete and FGD gypsum wallboard during use by the consumer are comparable to or lower than those from analogous non-CCR products, or are at or below relevant regulatory and health-based benchmarks for human and ecological receptors. Thus, EPA supports the continued beneficial use of coal fly ash in concrete and FGD gypsum in wallboard. Furthermore, the Agency believes that these beneficial uses provide significant environmental and economic benefits, and opportunities to advance Sustainable Materials Management (SMM).

References

American Association of State Highway and Transportation Officials (AASHTO). 1993. *Guide for Design of Pavement Structures*. Washington, DC.

AASHTO. 2004. *Geometric Design of Highways and Streets, Fifth Edition*. Washington, DC.

Abad-Valle, P., M. A. Lopez-Anton, M. Diaz-Somoano, and M.R. Martinez-Tarazona. 2011. The Role of Unburned Carbon Concentrates from Fly Ashes in the Oxidation and Retention of Mercury. *Chemical Engineering Journal*. 174(1): 86-92.

ACAA. 2013. *2012 Coal Combustion Product (CCP) Production & Use Survey Report.* Accessed on November 25, 2013. Available at: http://www.acaa-usa.org/Publications/ProductionUseReports.aspx

American National Standard Institute and Health Physics Society (ANSI and HPS). 2009. *Control and Release of Technologically Enhanced Naturally Occurring Radioactive Material (TENORM)*. ANSI/HPS N13.53-2009. Health Physics Society. Mclean, VA.

Apul, D.S., K. Gardner, T. Eighmy, J. Benoit, and L. Brannaka. 2002. *A Review of Water Movement in the Highway Environment: Implications for Recycled Materials Use*. Recycled Resource Center, University of New Hampshire, Durham, NH.

Arnot, J.A., and F.A.P.C. Gobas. 2006. A Review of Bioconcentration Factor (BCF) and Bioaccumulation Factor (BAF) Assessments for Organic Chemicals in Aquatic Organisms. *Environmental Reviews*. 14: 257-297.

California Environmental Protection Agency (CalEPA). 2012. *Air Toxics "Hot Spots" Program Risk Assessment Guidelines Technical Support Document for Exposure Assessment and Stochastic Analysis. Appendix I – Fish Bioaccumulation Factors. Scientific Review Panel Draft.* Office of Environmental Health Hazard Assessment.

Cheng, C., P. Taerakul, W. Tu, B. Zand, T. Butalia, W. Wolfe, and H. Walker. 2008. Surface Runoff from Full-Scale Coal Combustion Product Pavements during Accelerated Loading. *Journal of Environmental Engineering*. 134(8): 591-599.

Church, D. A., L. Radd, and M. Tumeo. 1995. Experimental Study of Leaching of Fly Ash. *Transportation Research Record*. 1486: 3-12.

Consumer Product Safety Commission (CPSC). 2010a. *Final Report on an Indoor Environmental Quality Assessment of Residences Containing Chinese Drywall*. EH&E Report 16512. Prepared for the U. S. Consumer Product Safety Commission by Environmental Health & Engineering, Inc. under CPSC Contract No. CPSC-S-09-0027. Bethesda, M.D.

CPSC. 2010b. *CPSC Staff Preliminary Evaluation of Drywall Chamber Test Results: Reactive Sulfur Gases*.

US Department of Energy. 2008. *Fate of Mercury in Synthetic Gypsum Used for Wallboard Production.* Prepared for the National Energy Technology Laboratory by J. Sanderson of the USG Corporation

and G.M. Blythe, and M. Richardson of the URS Corporation under DOE Contract No. DE-FC26-04NT42080. Pittsburgh, PA.

du Preez, H.H., E. van Rensburg, and J.H.J. van Vuren. 1993. Preliminary Laboratory Investigation of the Bioconcentration of Zinc and Iron in Selected Tissues of the Banded Tilapia, Tilapia Sparrmanii (Cichlidea). *Bulletin of Environmental Contamination and Toxicology*. 50: 674-681.

Eckert, J. O., and Q. Guo. 1998. Heavy Metals in Cement and Cement Kiln Dust from Kilns Co-Fired with Hazardous Waste-Derived Fuel: Application of EPA Leaching and Acid-Digestion Procedures. *Journal of Hazardous Materials*. 59: 55-93.

Electric Power Research Institute (EPRI). 2010. *Public Comments to Hazardous and Solid Management System; Identification and Listing of Special Wastes; Disposal of Coal Combustion Residuals from Electric Utilities*. Available in EPA docket EPA-HQ-RCRA-2009-0640 as document number EPA-HQ-RCRA-2009-0640-9765**.** Palo Alto, CA.

European Commission (EC). 1999. *Protection 112: Radiological Protection Principles Concerning the Natural Radioactivity of Building Materials.* Office for Official Publications of the European Communities. Available online at: http://ec.europa.eu/energy/nuclear/radiation_protection/doc/publication/112.pdf

EC. 2001. *Radiation Protection 122: Practical Use of the Concepts of Clearance and Exemption: Part II: Application of the Concepts of Exemption and Clearance to Natural Radiation Sources.* Office for Official Publications of the European Communities. Available online at: http://ec.europa.eu/energy/nuclear/radiation_protection/doc/publication/122_part2.pdf

Garboczi, E. J. 1995. *Microstructure and Transport Properties of Concrete.* In J. Kropp, and H.K. Hilsdorf (Eds.). *Performance Criteria for Concrete Durability*. 198-212. London, U.K., E & FN Spon.

Garrabrants, A.C., F. Sanchez, and D.S. Kosson. 2004. Changes in Constituent Equilibrium Leaching and Pore Water Characteristics of a Portland Cement Mortar as a Result of Carbonation. *Waste Management*. 24: 19-36.

Garrabrants, A.C., D.S. Kosson, R. Delapp, and H.A. Van der Sloot. 2013 (In Press). Effect of Coal Combustion Fly Ash Use in Concrete on the Mass Transport Release Constituents of Potential Concern. *Chemosphere*.

Giergiczny, Z., and A. Krol. 2008. Immobilization of Heavy Metals (Pb, Cu, Cr, Zn, Cd, Mn) in the Mineral Additions Containing Concrete Composites. *Journal of Hazardous Materials*. 160: 247-255.

Golightly, D.W., P. Sun, C.M. Cheng, P. Taerakul, H.W. Walker, L.K. Weavers, and D.M. Golden. 2005. Gaseous Mercury from Curing Concretes that Contain Fly Ash: Laboratory Measurements. *Environmental Science and Technology*. 39: 5689-5693.

Golightly, D.W., C.M. Cheng, L.K. Weavers, H.W. Walker, and W.E. Wolfe. 2009. Fly Ash Properties and Mercury Sorbent Affect Mercury Release from Curing Concrete. *Energy and Fuels*. 22: 3089-3095.

Gypsum Association. 2010. *Public Comments to Hazardous and Solid Management System; Identification and Listing of Special Wastes; Disposal of Coal Combustion Residuals from Electric Utilities*. Available in EPA docket EPA-HQ-RCRA-2009-0640 as document number EPA-HQ-RCRA-2009-0640-8227.

Henkels, P. J. and J. C. Gaynor. 1996. Characterizing Synthetic Gypsum for Wallboard Manufacture. *Preprints of Papers, American Chemical Society, Division of Fuel Chemistry*, 41(2). Spring National Meeting of the American Chemical Society, New Orleans, LA.

Ingersoll, J.G. 1983. A Survey of Radionuclide Contents and Radon Emanation Rates in Building Materials Used in the US. Ingersoll. *Health Physics*. 45(2): 363-368

International Atomic Energy Agency (IAEA). 2003. *Technical Report Series No. 419: Extent of Contamination by Naturally Occurring Radioactive Material (NORM) and Technological Options for Mitigation*. Available at: http://www-pub.iaea.org/MTCD/Publications/PDF/TRS419_web.pdf

Kairies, C.L., K.T. Schroeder, and C.R. Cardone. 2006. Mercury in Gypsum Produced from Flue Gas Desulfurization. *Fuel*. 85(2): 530-2,536.

Kanare, H. M. and P.B. West. 1993. Leachability of Selected Elements from Concrete. *Proceedings from Portland Cement Association Symposium on Cement and Concrete in the Global Environment*. O'Hare Marriott Hotel, Chicago, IL.

Kolver, K., A. Perevalov, V. Steiner, and L.A. Metzger. 2005a. Radon Exhalation of Cementitious Materials Made with Coal Ash: Part 1 – Scientific Background and Testing of the Cement and Fly Ash Emanation. *Journal of Environmental Radioactivity*. 82: 321-334.

Kolver, K., A. Perevalov, V. Steiner, and L.A. Metzger. 2005b. Radon Exhalation of Cementitious Materials Made with Coal Ash: Part 2 – Testing Hardened Cement Fly Ash Pastes. *Journal of Environmental Radioactivity*. 82: 335-350.

Kosson, D.S. and A.C. Garrabrants. 2012. Personal communication, November 9, 2012.

Kosson, D.S., A.C. Garrabrants, R. Delapp, and H.A. Van der Sloot. 2013 (In Press). pH-dependent Leaching of Constituents of Potential Concern from Concrete Materials Containing Coal Combustion Fly Ash. *Chemosphere*.

Lindeken, C.L. and D.G. Coles. 1977. The Radium-226 Content of Agricultural Gypsums (UCRL-78720). Lawrence Livermore Laboratory, University of California. Berkeley, CA.

Long, C.M., S.N. Sonja, and A.S. Lewis. 2012. Potential Indoor Air Exposures and Health Risks from Mercury Off-Gassing of Coal Combustion Products Used in Building Materials. *Coal Combustion and Gasification Product*. 4: 68-74.

Massachusetts Department of Environmental Protection (MassDEP). 1995. *Guidelines for the Design, Installation, and Operation of Sub-Slab Depressurization Systems*. Prepared for MassDEP by T. DiPersio and J. Fitzgerald of BWSC/NERO.

McCann, R., K. L. Hartung, J. Hedgecock, P. Schuler, and S. Putrich. 2007. Risk Evaluation of Leachable Mercury From Concrete Products Made Fly Ash. *World of Coal Ash.* Covington, KY.

McCarrol, N., N. Keshava, J. Chen, G. Akerman, A. Kligerman, and E. Rinde. 2010. Review Article: An Evaluation of the Mode of Action Framework for Mutagenic Carcinogens Case Study II: Chromium (VI). *Environmental and Molecular Mutagenesis*. 51: 89-111.

Meij, R., E. Kokmeijer, L. Tamboer, and H. T. Winkel. 2001. Field Leaching of Bricks and Concrete Containing Coal Fly Ash. *2001 International Ash Utilization Symposium*. Center for Applied Energy Research, University of Kentucky.

Müllauer, W., R. E. Beddoe, and D. Heinz. 2012. Effect of Carbonation, Chloride, and External Sulphates on the Leaching Behavior of Major and Trace Elements from Concrete. *Cement and Concrete Composites*. 34: 618-626.

National Commission for Radiological Protection and Measurements (NCRP). 2009. *Ionizing Radiation Exposure of the Population of the United States. Report 160*. National Council on Radiation Protection and Measurements. Bethesda, MD.

New Jersey Department of Environmental Protection (NJDEP). 2009. *Derivation of Ingestion-Based Soil Remediation Criterion for Cr+6 Based on the NTP Chronic Bioassay Data for Sodium Dichromate Dihydrate.* New Jersey Department of Environmental Protection. Trenton, NJ. Available at: http://www.state.nj.us/dep/dsr/chromium/soil-cleanup-derivation.pdf

Nuclear Regulatory Commission (NRC). 2001. *Systematic Radiological Assessment of Exemptions for Source and Byproduct Materials* (NUREG-1717). Prepared for the NRC by S. Schneider, D.C. Kocher, G.D. Kerr, P.A. Scofield, F.R. O'Donnell, C.R. Mattsen, S.J. Cotter, J.S. Bogard of the Oak Ridge National Laboratory, and J.S. Bland and C. Wiblin of J. Steward Bland Associates, Inc. Washington, DC. Available at: http://www.nrc.gov/reading-rm/doc-collections/nuregs/staff/sr1717/nureg-1717.pdf

National Toxicology Program (NTP). 2012. *NTP Study of the Hexavalent Chromium Compound Sodium Dichromate Dihydrate*. Available at: http://ntp.niehs.nih.gov/index.cfm?objectid=072F592F-D1EC-A321-819ABF09DC08EBE1 (accessed July 17, 2012).

Pflughoeft-Hassett, D.F., D.J. Hassett, and C.M. Lillemoen. 1993. Comparative Leaching of Midwestern Coal Fly Ash and Cements. *Proceedings of the Tenth ACAA International Ash Use Symposium*, Vol. 1, 30-1 to 30-14. Energy & Environmental Research Center, University of North Dakota, Grand Forks, ND.

Portland Cement Association (PCA). 1992. *An Analysis of Selected Trace Metals in Cement and Kiln Dust*. Portland Cement Association. Skokie, IL.

PCA. 2003. *Design and Control of Concrete Mixtures, 14th ed.* Portland Cement Association. Skokie, IL.

PCA. No Date, a. *Concrete Masonry Units*. Available at: http://www.cement.org/masonry/block.asp. (Accessed May 13, 2012).

PCA. No Date, b. *Technical Brief: Green Practices 102 – Concrete, Cement, and CO_2*. Available at: http://www.concretethinker.com/technicalbrief/Concrete-Cement-CO2.aspx. (Accessed May 13, 2012).

Rankers, R.H. and I. Hohberg. 1991. Leaching Tests for Concrete Containing Fly Ash – Evaluation and Mechanism. *Waste Materials in Construction*. 48: 275-292.

Roper, A.R., M.G. Stabin, R.C. Delapp, and D.S. Kosson. 2013. Analysis of Naturally-Occurring Radionuclides in Coal Combustion Fly Ash, Gypsum, and Scrubber Residue Samples. *Health Physics*. 104(3): 264-269.

Sanchez., F., C. Gervais, A. C. Garrabrants, R. Barna, and D.S. Kosson. 2002. Leaching of Inorganic Contaminants from Cement-Based Waste Materials as a Result of Carbonation during Intermittent Wetting. *Waste Management*. 22: 249-260.

Shock, S.S., J.J. Noggle, N. Bloom, and L.J. Yost. 2009. Evaluation of Potential for Mercury Volatilization from Natural and FGD Gypsum Products Using Flux-Chamber Tests. *Environmental Science and Technology*. 43: 2282-2287.

Stern, A.H. 2010. A Quantitative Assessment of the Carcinogenicity of Hexavalent Chromium by the Oral Route and its Relevance to Human Exposure. *Environmental Research*. 10: 798-807.

Treasury and General Government Appropriations Act for Fiscal Year 2001, Public Law No. 106-554, §515, H.R. 5658.

United Kingdom Department of the Environment, Transport and the Regions (UKDETR). 1998. *Improved Materials Testing: Leaching Tests on Ready-Mixed Concrete.*

United Kingdom Health Protection Agency (UKHPA). 2004. *A Radiological Study of Pulverized Fuel Ash (PFA) from UK Coal-Fired Power Stations*. Radiation Protection Division. Chilton, Didcot, Oxfordshire. Available online at: http://www.ukqaa.org.uk/index_htm_files/AshTechA14ARadiologicalStudyOfPFAGoodingEtAl.pdf

United Kingdom National Radiological Protection Board (UKNRPB). 2001. *Radiological Impact on the UK Population of Industries Which Use or Produce Materials Containing Enhanced Levels of Naturally Occurring Radionuclides: Part 1 - Coal-fired Electricity Generation*. (NRPB-R327). Chilton, Didcot, Oxfordshire.

United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR). 1982. *Ionizing Radiation: Sources and Biological Effects*. New York. Available online at: http://www.unscear.org/unscear/en/publications/1982.html

UNSCEAR. 1988. *Ionizing Radiation: Sources, Effects and Risks of Ionizing Radiation*. New York. Available online at: http://www.unscear.org/unscear/en/publications/1988.html

UNSCEAR. 1993. *Ionizing Radiation: Sources and Effects of Ionizing Radiation*. New York. Available online at: http://www.unscear.org/unscear/en/publications/1993.html

UNSCEAR. 2000. *Ionizing Radiation: Sources and Effects of Ionizing* Radiation - Volume 1: Sources. New York. Available online at: http://www.unscear.org/unscear/en/publications/2000_1.html

UNSCEAR. 2008. *Ionizing Radiation: Sources and Effects of Ionizing* Radiation - Volume 1. New York. Available online at: http://www.unscear.org/unscear/en/publications/2008_1.html

US Environmental Protection Agency (US EPA). 1988. *Report to Congress: Wastes from the Combustion of Coal by Electric Utility Power Plants* (EPA 530-SW-88-002). Office of Solid Waste and Emergency Response. Washington, DC.

US EPA. 1989. *Risk Assessment Guidance for Superfund (RAGS) Part A* (EPA 530-SW-88-002). Office of Solid Waste and Emergency Response. Washington, DC.

US EPA. 1991. *EPA Region 3 Guidance on Handling Chemical Concentration Data Near the Detection Limit in Risk Assessments* (EPA/903/8-91/001). Region 3. Philadelphia, PA. Available at: http://www.epa.gov/reg3hwmd/risk/human/info/guide3.htm.

US EPA. 1997a. *Mercury Study Report to Congress: Volume V: Health Effects of Mercury and Mercury Compounds* (EPA-452/R-97-007). Office of Air and Office of Research and Development Washington, DC. Available at: http://www.epa.gov/ttnatw01/112nmerc/volume5.pdf

US EPA. 1997b. *Exposure Factors Handbook, Volumes I, II, and III* (EPA/600/P-95/002Fa). Office of Research and Development. Washington, DC.

US EPA. 1998. *Supplemental Report to Congress on Remaining Wastes from Fossil Fuel Combustion Technical Background Document: Beneficial Use of Fossil Fuel Combustion (FCC) Wastes*. Office of Solid Waste. Washington, DC.

US EPA. 1999. *Report to Congress: Wastes from the Combustion of Fossil Fuels: Volume 2 – Methods, Findings, and Recommendations* (EPA 530-R-99-010). Office of Solid Waste and Emergency Response. Washington, DC.

US EPA. 2002. *Constituent Screening for Coal Combustion Wastes*. Draft Report prepared by Research Triangle Institute for Office of Solid Waste, under EPA Contract No. 68-W-98-085. Washington, DC.

US EPA. 2003a. *Summary of General Assessment Factors for Evaluating the Quality of Scientific and Technical Information* (EPA 100/B-03/001). Science Policy Council. Washington, DC.

US EPA. 2003b. *Human Health Toxicity Values in Superfund Risk Assessments* (Directive 9285.7-53). Office of Solid Waste and Emergency Response. Washington, DC.

US EPA. 2007a. *Framework for Metals Risk Assessment* (EPA 120/R-07/001). Risk Assessment Forum. Washington, DC.

US EPA. 2007b. *Ecological Soil Screening Levels for Manganese. Interim Final.* OSWER Directive 9285.7-71. Office of Solid Waste and Emergency Response. Washington, DC.

US EPA. 2008a. *Brownfields Technology Primer: Vapor Intrusion Considerations for Redevelopment* (EPA 542-R-08-001). Office of Solid Waste and Emergency Response. Washington, DC.

US EPA. 2008b. *Child-Specific Exposure Factors Handbook* (EPA/600/R-06-096F). Washington, DC: US EPA, Office of Research and Development.

US EPA. 2009a. *Characterization of Coal Combustion Residues from Electric Utilities - Leaching and Characterization Data* (EPA-600/R-09/151). Prepared for the Office of Research and Development by D. Kosson, F. Sanchez, R. Delapp of Vanderbilt University; L.H. Turner of Turner Technology, LLC; and P. Seignett of the Energy Research Centre of the Netherlands under EPA Contract No. EP-C-09-027. Washington, DC.

US EPA. 2009b. *Inhalation of Fugitive Dust: A Screening Assessment of the Risks Posed by Coal Combustion Waste Landfills (Draft)* (EP-W2-09-004). US EPA, Office of Solid Waste and Emergency Response. Washington, DC.

US EPA. 2009c. *Characterization of Coal Combustion Residues from Electric Utilities – Leaching and Characterization Data.* (EP-600/R-09/151). Prepared for the Office of Research and Development by D. Kosson, F. Sanchez, and R. Delapp of Vanderbilt University; P.Kariher of ARCADIS; L.H. Turner of Turner Technology, LLC; and P. Seignett of the Energy Research Centre of the Netherlands under EPA Contract No. EP-C-09-027. Washington, DC.

US EPA. 2010a. *Human and Ecological Risk Assessment of Coal Combustion Wastes* (RIN 2050-AE81; EP-W2-09-004). Office of Solid Waste and Emergency Response. Washington, DC.

US EPA. 2010b. *ProUCL Version 4.00.02 User Guide* (EPA/600/R-07/038). Prepared for the Office of Research and Development and the Technology Support Center of Region 4 by A. Singh, R. Maichle, and N. Armbya of Lockheed Martin Environmental Services. Washington, DC. Available at: http://www.epa.gov/osp/hstl/tsc/ProUCL_v4.00.02_user.pdf

US EPA. 2010c. *ProUCL Version 4.1 Technical Guide* (EPA/600/R-07/041). Prepared for the Office of Research and Development and the Technology Support Center of Region 4 by A. Singh and N. Armbya of Lockheed Martin Environmental Services and A.K. Singh of the University of Nevada Department of Hotel Management. Washington, DC. Available at: http://www.epa.gov/osp/hstl/tsc/ProUCL_v4.0_tech.pdf

US EPA. 2010d. *Industrial Waste Management Evaluation Model (IWEM) Version 2.0 with Roadway Module*: *Technical Documentation and User's Guide*. Prepared for the Office of Solid Waste and Emergency Response by M. Lowry of the Research Triangle Institute and T. Lillys and V. Guvanasen of HydroGeoLogic Inc. under EPA Contract No. EP-W-09-004. Washington, DC.

US EPA. 2010e. *A Brief Guide to Mold, Moisture, and Your Home*. Office of Air and Radiation. Washington, DC. Available at: http://www.epa.gov/mold/pdfs/moldguide.pdf

US EPA. 2011. *Exposure Factors Handbook: 2011 Edition,* (EPA/600/R-090/052F). Office of Research and Development. Washington, DC.

US EPA. 2012a. *Peer Review Summary Report: Independent External Peer Review of the Preliminary Draft Report Methodology for Evaluating Encapsulated Beneficial Uses of Coal Combustion Residuals*. Prepared for the Office of Solid Waste and Emergency Response by Versar under EPA Contract No. EP-C-07-025. Washington, DC.

US EPA. 2012b. *The Impact of Coal Combustion Fly Ash Used as a Supplemental Cementitious Material on the Leaching of Constituents from Cements and Concretes* (EPA-600/R-12/704).

Prepared for the Office of Research and Development by H.A. van der Sloot of Hans van der Sloot Consultancy and D.S. Kosson, A.C. Garrabrants, and J. Arnold of Vanderbilt University School of Civil and Environmental Engineering under EPA Contract No. EP-C-09-27. Washington, DC.

US EPA. 2013a. *Methodology for Evaluating Encapsulated Beneficial Uses of Coal Combustion Residuals.* Office of Solid Waste and Emergency Response. Washington, DC.

US EPA. 2013b. *Responses to External Peer Review Comments: Methodology for Evaluating Encapsulated Beneficial Uses of Coal Combustion Residuals*. Office of Solid Waste and Emergency Response. Washington, DC.

United States Geological Survey (USGS). 1997. *Radioactive Elements in Coal and Fly Ash: Abundance, Forms, and Environmental Significance*. Available online at: http://pubs.usgs.gov/fs/1997/fs163-97/FS-163-97.html

United States Office of Management and Budget (US OMB). 1998. *Circular No. A-119 – Federal Register (Federal Participation in the Development and Use of Voluntary Consensus Standards and in Conformity Assessment Activities).* Available at: http://www.whitehouse.gov/omb/circulars_a119_a119fr

United States Office of Management and Budget (US OMB). 2002. *Guidelines for Ensuring and Maximizing the Quality, Objectivity, Utility, and Integrity of Information Disseminated by Federal Agencies* (67 CFR 8452). Available at: http://www.whitehouse.gov/omb/fedreg_final_information_quality_guidelines

Van Gerven, T., G. Cornelis, E. Vandoren, C. Vandecasteele, A.C. Garrabrants, F. Sanchez, and D. S. Kosson. 2006. Effects of Progressive Carbonation on Heavy Metal Leaching from Cement-Bound Waste. *American Institute of Chemical Engineers*. 52(2): 826-837.

Wollenberg, H.A. and A.R. Smith. 1962. *Portland Cement for a Low-Background Counting Facility* (UCRL-10475). Lawrence Radiation Laboratory, University of California. Berkeley, CA.

Wilcox, J., E. Rupp, S. C. Ying, D. Lim, A. S. Negreira, A. Kirchofer, F. Feng, and K. Lee. 2012. Mercury Adsorption and Oxidation in Coal Combustion and Gasification Processes. *International Journal of Coal Geology*. 90-91: 4-20.

Washington State Department of Transportation (WSDOT). 2010. *Studded Tire Wear on Portland Cement Concrete Pavement in the Washington State Department of Transportation Route Network* (WA-RD 744.3). Available at: http://ntl.bts.gov/lib/41000/41100/41185/744.3.pdf

Yost, L.J., S.S. Shock, S.E. Holm, Y.W. Lowney, and J.J. Noggle. 2010. Lack of Complete Exposure Pathways for Metals in Natural and FGD Gypsum. *Human and Ecological Risk Assessment: An International Journal*. 16(2): 317-339.

Zhang, M. H., M.C. Blanchette, and V.M. Malhotra. 1985. Leachability of Trace Metal Elements from Fly Ash, and from Concrete Incorporating Fly Ashes. Volume 1: *7th CANMET/ACI International Conference on Fly Ash, Silica Fume, Slag, and Natural Pozzolans in Concrete.*

Zhang, M., M.C. Blanchette, and V.M. Malhotra. 2001. Leachability of Trace Metal Elements from Fly Ash Concrete: Results from Column-Leaching and Batch-Leaching Tests. *ACI Materials Journal*. 98(2): 126-136.

Zielinski, R.A., C.A. Rice, J.T. O'Connor, and R.H. Affolter. 1998. Uranium in Coal Fly Ash: Abundance, Forms, Leachability, and Environmental Significance. *Abstract Volume, Division of Environmental Geochemistry, American Chemical Society 216th National Meeting, August 23-27, 1998*. Botson, MA. 38(2):190-193.

Zikovsky, L. and G. Kennedy. (1992). Radioactivity of Building Materials Available in Canada. *Health Physics*. 63(4): 449-452.

Zubeck, H., L. Aleshire, S. Harvey, S. Porhola, and E. Larson. 2004. *Socio-Economic Effects of Studded Tire Use in Alaska*. Anchorage, Alaska.

EPA530-R-14-001