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Novel Approaches in Determining Baseline Information on Annual Disposal Rates and Trace Element Content of U.S. Coal Combustion Residues: A Response to EPA's June 2010 Proposed Disposal Rule

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Novel Approaches in Determining Baseline Information on Annual Disposal Rates and Trace Element Content of U.S. Coal Combustion Residues: A Response to EPA's June 2010 Proposed Disposal Rule

by

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Thesis

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Dedication

To Mum and Dad, who have always been my strongest supporters.
I couldn’t have asked for more loving parents, nor truer friends.

And to my Dukies, for keeping me grounded throughout this journey.
Special thanks to Erica, roommate extraordinaire, for much encouragement, advice, and the occasional (well-deserved) kick in the pants; and to Caleb, for lending a patient ear to many long conversations about all things academia, and all things beyond it.
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Abstract

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The University of Texas at Austin, 2010

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Although products of coal combustion (PCCs) such as coal ash are currently exempted from classification as a hazardous waste in the United States under the 1976 Resource Conservation and Recovery Act (RCRA), the U.S. Environmental Protection Agency (EPA) is now revising a proposed rule to modify disposal practices for these materials in order to prevent contamination of ground- and surface water sources by leached trace elements.

This paper analyzes several aspects of EPA's scientific reasoning for instating the rule, with the intent of answering the following questions: 1) Are EPA's cited values for PCC production and disposal accurate estimates of annual totals?; 2) In what ways can EPA's leaching risk modeling assessment be improved?; 3) What is the total quantity of trace elements contained within all PCCs disposed annually?; and 4) What would be the potential costs and feasibility of reclassifying PCCs not under RCRA, but under existing NRC regulations as low-level radioactive waste (LLRW)?

Among the results of my calculations, I found that although EPA estimates for annual PCC disposal are 20% larger than industry statistics, these latter values appear to be closer to reality. Second, EPA appears to have significantly underestimated historical PCC disposal: my projections indicate that EPA's maximum estimate for the quantity of fly ash landfilled within the past 90 years was likely met by production in the last 30 years alone, if not less. Finally, my analysis indicates that while PCCs may potentially meet the criteria for reclassification as low-level radioactive waste by NRC, the cost of such regulation would be many times that of the EPA June proposed disposal rule ($220-302 billion for PCCs disposed in 2008 alone, versus $1.47 billion per year for the Subtitle C option and $236-587 million for Subtitle D regulatory options).

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Introduction

WHAT IS COAL ASH?

Coal generates about half of all electricity in the United States. In 2008, the combustion of over 1 billion short tons of coal produced nearly 2 million GWh of electricity, supplying 48.2% of national power consumption (EIA, 2010a, p. 8; EIA, 2010b, p. 12). In comparison, American nuclear power plants produced 0.8 million GWh of power during the same year, supplying 19.4% of domestic electricity consumption (EIA, 2010b, p. 12).

As with other fossil fuel sources, the conversion of coal to energy is not a fully-efficient process, and creates waste in the form of what I will here generally term PCCs, or "products of coal combustion."[1] Although many such materials are generated by the combustion process, the majority of PCC output is composed of three components: fly ash, bottom ash, and boiler slag. Collectively referred to here as "coal ash," these materials totaled nearly 93 million short tons in 2008 alone, or 68% of total PCC production (ACAA, 2010a).[2]

Each of the three constituents of coal ash is formed at a different stage of the combustion process in a coal-fired power plant, and therefore possesses unique physical and chemical properties. The most plentiful of the three ash types is fly ash, vitrified particles <20 μm in diameter which are carried along by hot flue gases before being trapped by emissions-control devices in a power plant's stacks. Depending on the efficiency of these filtration devices, approximately 95%-99.5% of the ash may be lost to

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[1] See Introduction, Endnote #1, p. 91.

[2] See Introduction, Endnote #2, p. 92.

the atmosphere, while the remainder is retained (Beck, 1989, p. 286). The next subcategory of coal ash, bottom ash, is composed of particles which settle to the bottom of the coal boiler due to being too large or too heavy to be entrained in flue gases. Finally, boiler slag is most often formed in coal-fired power plants utilizing older slag tap or cyclone furnaces; in this process, coal ash particles melt at high temperatures during combustion, forming dense, glass-like masses which collect at the bottom of the boiler and in stack exhaust filters.

ENVIRONMENTAL CONCERNS

There historically have been two general methods for disposal of coal ash and other PCCs. In "dry storage," the disposed material is simply dumped into an open-air landfill which may be at or below surface level. In "wet storage," the PCCs are placed into a surface impoundment, which may be generally defined as a topographic depression or man-made structure used to receive PCCs that have been flushed or mixed with water to facilitate movement. This water is often obtained from nearby water bodies such as creeks or rivers, and most active surface impoundments will undergo periodic exchanges of water and/or PCC "slurry" between the surface impoundment and the surface water source during their lifecycles, with the frequency of these events varying by disposal facility and company practices. As defined by EPA, examples of surface impoundments are "holding, storage, settling, and aeration pits, ponds, and lagoons" (*Federal Register*, 2010, p. 35130).

Today, increasingly, a third disposal option exists for PCCs: captured materials – particularly coal ash – may be sold by the producer and transported off-site for reuse in a variety of "beneficial use" applications. Coarse-grained bottom ash may be used in structural fill material for landscaping purposes or creation of road bases, for example, while fly ash is commonly used as a strengthening agent in production of Portland cement. Nevertheless, this trend towards beneficial use has only truly gained momentum in the last ten to fifteen years. Even today, the majority of coal ash produced in the United States is still disposed as waste in landfills and surface impoundments.

As will be discussed in greater detail in Chapter 2 below, the majority of such facilities in the U.S. have not been constructed with protections in place to minimize leaching of stored materials upon exposure to water. Surface impoundments, which contain thousands to millions of gallons of watery PCC "slurry," therefore may be particularly prone to both leaching of disposed materials and horizontal migration of this leachate into the local watertable; yet 62% of all national surface impoundments surveyed by EPA in 2004 lacked artificial or clay liners to restrict leachate movement, and 58% had no groundwater monitoring measures in place should such leaching occur (*Federal Register,* 2010, p. 35151).

The reason why some environmental groups challenge the above disposal practices is that, during the combustion process, a variety of trace elements contained naturally in whole coal become artificially-enriched on the particle surfaces of coal ash and other PCC materials. These trace elements may include heavy metals such as lead and mercury, as well as radionuclides; of particular concern in this latter category are radioactive isotopes ^{238}U, ^{235}U, ^{232}Th, and associated daughter products such as radium and radon (Van Hook, 1979, p. 228). Thus, there is concern that when PCCs are stored in open-air disposal units that lack covers or liners to protect contents from exposure to rain or to isolate them from underlying groundwater sources, trace elements could be leached from particle surfaces and subsequently contaminate ground- and surface water supplies, posing a hazard to public health and the environment.

REGULATORY HISTORY

While there is a wide body of literature and regulations addressing the potential environmental and public health impacts of atmospheric releases of flue gas-entrained coal ash produced by coal-fired power plants (*cf.* EPA, 1998; Van Hook, 1979; McBride *et al.*, 1978), historically there has been much less attention focused on any such impacts associated with the *captured* coal ash. This apparent dichotomy can be explained quite simply by differences in respective federal regulation. Under the U.S. Clean Air Act, any coal ash which escapes a power plant's filters and is emitted to the atmosphere constitutes a form of air pollution, and therefore must be managed under EPA guidelines to comply with emissions limits for airborne particulate matter. Yet for the much greater 95-99% of coal ash (particularly fly ash) that is produced by coal combustion but captured by a power plant's filtration systems prior to atmospheric release, no equivalent regulation currently exists to clarify how this captured ash can be used or stored, or whether any monitoring of ash after disposal is necessary.

The reason for this regulatory gap can be traced back to 1980, when Congress passed the Solid Waste Disposal Act Amendments (also known as Public Law 96-482, or the "Bevill Amendment") modifying the 1976 Resource Conservation and Recovery Act (RCRA). As the principal federal law governing the disposal of both hazardous and non-hazardous waste in the United States, RCRA established not only a comprehensive set of criteria to determine whether a waste was hazardous, but also a "cradle-to-grave" monitoring system to track hazardous waste from production to disposal. Prior to the Bevill Amendment, coal ash and other PCCs were treated as solid wastes under RCRA regulations, and thus technically also were able to be classified as hazardous wastes if certain requirements were met:

> (2) The following materials are solid wastes, even if the recycling involves use, reuse, or return to the original process […]:
> (i) Materials used in a manner constituting disposal, or used to produce products that are applied to the land;
> (ii) Materials burned for energy recovery, used to produce a fuel, or contained in fuels.
>
> (40 CFR 261.2(e).2)

Under the 1980 amendments, however, a special exception was made under RCRA for PCCs, preventing them from being classified as hazardous wastes unless already handled at facilities managing other hazardous wastes:

> (b) *Solid wastes which are not hazardous wastes.* The following solid wastes are not hazardous wastes:
> […] (4) Fly ash waste, bottom ash waste, [boiler] slag waste, and flue gas emission control waste, generated primarily from the combustion of coal or other fossil fuels, except as provided […] for facilities that burn or process hazardous waste.
>
> (40 CFR 261.4(b).4)

In essence, then, this exemption allowed PCC producers to continue their existing disposal practices without fearing accusations of environmental contamination caused by the discarded materials. Thus, the practical result of the Bevill Amendment was that few disposal facilities bothered to undertake the extra expense of installing clay or plastic liners to prevent possible leaching from disposed PCCs at existing landfill and surface impoundment sites, nor to incorporate these barriers into the designs of new disposal sites (again, a topic which is discussed in greater detail in Chapter 2 below).

Nevertheless, even while issuing its "nonhazardous" exemption, the Bevill Amendment also directed EPA to perform further research related to PCC disposal practices and any associated environmental or public risks (*i.e.,* the potential consequences of leaching from materials placed into landfills or surface impoundments for disposal, and/or repurposed by industry for “beneficial” uses such as cement manufacturing). Over the next two decades, this order would result in several Congressional reports by EPA, as well as the publication of two major regulatory determinations in 1993 and 2000.

It is worth noting that there is a common misconception that the 2000 EPA ruling was somehow the “final word,” incontrovertibly verifying the nonhazardous nature of PCCs. The truth is that EPA only firmly recognized the “low risk” nature of PCCs used for beneficial purposes, particularly when in stable, “encapsulated” forms such as cement and other construction materials (*Federal Register*, 2000, p. 32229). For the much larger volume of PCCs annually disposed in landfills and surface impoundments, however, EPA made it quite clear that the risk profile was quite different:

> [W]hile significant improvements are being made in waste improvement practices due to increasing state oversight, gaps in the current regulatory regime remain. We have determined that the establishment of national regulations is warranted for coal combustion wastes when they are disposed in landfills and surface impoundments, [in part] because […] the composition of these wastes has the potential to present danger to human health and the environment under some circumstances.
>
> (*Federal Register*, 2000, p. 32230)

Furthermore, the document makes repeated mentions of a forthcoming rulemaking which would incorporate new information currently missing from or thought to be inaccurate in the 2000 ruling, particularly regarding the groundwater model used for analyzing the potential risks from leaching of metals contained in PCCs (*cf. Federal Register*, 2000, pp. 32227, 32230). Thus, the 2000 ruling was not so much a “final word” on the matter, but rather more a tentative acknowledgement that risk potential was indeed present and that this issue would be addressed more clearly by EPA in the future.

THE KINGSTON SPILL

When no PCC disposal legislation was forthcoming eight years after the 2000 EPA ruling, it appeared that the measure had stalled, perhaps permanently: whereas evidence of legitimate threats of public health might have spurred disposal legislation forward, EPA studies of 11 proven damage cases related to PCC disposal instead had found no evidence of human exposures to leached trace elements (*Federal Register*, 2000, p. 32230). In December 2008, however, conceptions about the relative "harmlessness" of PCCs were abruptly reversed, as the Kingston fly ash spill became the United States' "largest environmental disaster" since the 1989 *Exxon Valdez* oil spill (Dewan, 2008a; Dewan, 2008b).

Located approximately 40 miles west of Knoxville, Tennessee, the Kingston Fossil Plant is situated at the confluence of the Clinch and Emory Rivers, both tributaries of the Tennessee River. It is one of the largest fossil-fuel fired facilities owned and operated by the Tennessee Valley Authority (TVA), at full capacity burning about 14,000 tons of coal daily; prior to the spill, the plant provided power to over 670,000 households in the region, and generated an estimated 1,000 tons of fly ash daily (U.S. House, 2009a, p. vi-vii). This ash and other PCCs had been placed into three on-site, above-ground surface impoundments for disposal; the unlined retaining pond involved in the December 2008 spill spanned 84 acres, and had contained about 9.4 million cubic yards of PCC material, primarily fly ash (U.S. House, 2009a, p. ix).

At approximately 1:00 a.m. on December 22, 2008, a retaining wall at this surface impoundment experienced a major structural failure, allowing the escape of approximately 5.4 million cubic yards of coal ash sludge and 327 million gallons of water in what EPA termed a "tidal wave of water and ash" through the breach (U.S. House, 2009a, p. 1). On land, the spill ultimately deposited ash over more than 300 acres, with sludge buildup in some areas exceeding six feet in height; although no individuals were hurt by the spill itself, three homes were destroyed and 42 other properties were damaged (U.S. House, 2009a, pp. viii, 1). More importantly, however, a large proportion of the spilled fly ash entered the nearby Clinch and Emory waterways, in some cases forming deposits on the riverbed more than 20 feet thick (*cf.* TVA, 2010a, pp. 5-9, 15; U.S. House, 2009a, p. vii).

The Kingston spill itself was eventually determined to have been caused by a specific, long-evolving, and complex set of at least four different conditions, some of which were not necessarily typical of other PCC wet-storage measures used by other power plants within the U.S. For example, a study performed for the Kingston facility in 2004 identified an "unusual foundation layer consisting of sensitive slimes and silt" with "potential for liquefaction" at the storage pond in question. This layer has been determined to have been a driving factor behind the initial impoundment wall failure (U.S. House, 2009b, pp. 26-27, 30). Nevertheless, the sheer volume of ash involved in the spill was enough to spark EPA concerns that other surface impoundments across the U.S. might be in similarly poor condition, and so could also pose a potential environmental hazard.

These concerns were further justified when the first environmental impact studies began to quantify the leaching threat posed by fly ash from the Kingston spill. In briefest terms, the primary finding was that although the high heavy metal content of the spilled fly ash and the high solubility of these compounds had resulted in at least some amount of leaching to the Emory and Clinch Rivers, final concentrations of these heavy metals in downstream waters had been diluted nearly to baseline trace levels due to the large volume of water involved in the rivers' convergence (Ruhl *et al.* 2009, p. 6330-6331). While this was good news for local communities concerned about quality of drinking water supplies, the researchers warned that the majority of heavy metals such as mercury remained entrained in spilled ash and sediments along the river bottoms, and expressed particular concern for potentially major ecological impacts related to uptake and bioaccumulation of heavy metals in local food webs (Ruhl *et al.* 2009, p. 6330; see also Chapter 2 for further discussion of impacts of heavy metal-containing leachate on organisms and ecosystems).

The issue of possible ecological effects from leaching of ash released by the Kingston spill remains a concern. Whereas TVA has made reasonable progress in dredging “time-critical” areas of the affected waterways, the cleanup efforts are ongoing and far from complete, both for coal ash deposited on land as well as in the river system. Until cleanup is complete, the spilled ash may continue to pose risk to both public health and that of local ecosystems, as trace elements are leached either from land-deposited ash by rainwater – particularly during the region’s “customarily rainy winter months” (TVA, 2009a, p. 1) – or from waterway-deposited ash by the movement of water or other chemical reactions in the riverbed environment.

As of mid-February 2010, over 2.6 million cubic yards of the ash slurry had been recovered from the Emory River and the immediate proximity of the breached surface impoundment, and an additional 50,000 cubic yards were said to be “on track” for removal by the end of April 2010 (TVA, 2010b, pp. 2, 4). No clear end-date has yet been established for removal of the 1.195 million cubic yards of ash still remaining in the Emory River (TVA, 2010b, p. 4), but as per TVA’s recent “non-time critical” removal plan, full cleanup of the total 2.5 million cubic yards of all remaining spilled coal ash (i.e., for deposits both on land and in local waterways) should be complete by Q2 2014 at latest (TVA, 2010c, pp. 3, 16).

Thus far, TVA has spent an estimated $3 million to cover oversight costs by the Tennessee Department of Environment and Conservation (TDEC), $40.2 million to purchase property from 142 individuals living in or near the region affected by the Kingston ash spill, and $43 million to facilitate cleanup efforts via grants to Roane County (Stambaugh, 2010; Dewan, 2009). Although final estimates of cleanup costs will not be available until operations are concluded, some recent reports indicate this may be close to $1.2 billion (Dewan, 2009). In comparison, initial post-spill estimates ranged from $525-825 million, not including the costs of long-term environmental remediation (TVA, 2009b, p. 2). Nevertheless, even this $1.2 billion amount may be an underestimate of the total costs incurred to TVA by the Kingston fly ash spill, as the corporation now faces not only several federal lawsuits, but also has recently been penalized $11.5 million by TDEC for violations of the Tennessee Water Quality Act and the Tennessee Solid Waste Disposal Act (Poovey, 2010).

ASH DISPOSAL REGULATION TODAY

Although federal and state agencies' greatest fears about drinking water contamination from the Kingston ash spill were never realized (see again Ruhl *et al.* 2009; U.S. House, 2009a), the scale of the disaster nevertheless gave environmental organizations' continued push for stricter PCC disposal regulations some much-needed political support. Thus, when EPA presented its proposed ruling in June 2010, along with the results of numerous new studies and models assessing risk assessment for PCC disposal, it unsurprisingly placed heavy emphasis on prevention of ash spills at other U.S. disposal facilities.

Although the ruling as recorded in *Federal Register* (2010) is important for the obvious reasons, in that it suggests regulatory alternatives for coal ash disposal as a follow-up to the 2000 EPA ruling, it is equally interesting to see how greatly EPA's stance on the matter has changed over the past decade. While it is an overgeneralization to state that this increased awareness of the potential public and environmental health risks posed by coal ash and other PCCs has been entirely a consequence of the Kingston ash spill, there can be no doubt that the proposed rule has been designed in recognition that the contamination levels due to leaching of released fly ash after the Kingston spill were fortuitously low in comparison to initial estimates, and that contamination from future release ash release events might be much worse.

In simplest terms, the new proposed rule solicits public comment on two potential methods for future regulation of PCC disposal and associated trace element leaching risks. In a RCRA Subtitle C approach, EPA would assume more comprehensive "top-down" control by enforcing a new federal "special waste" category created to regulate all U.S.-produced PCCs on a lengthier "cradle-to-grave" timescale. In the alternative Subtitle D approach, EPA would establish baseline minimum federal guidelines for PCC disposal, but enforcement largely would be left to the individual states. EPA believes that regulations under either of these co-proposed rules should create a strong incentive for disposers to phase out use of existing surface impoundments and slow construction rates for new surface impoundments, in favor of a transition to landfills for PCC disposal and storage, thus effectively removing a major potential contamination pathway (*i.e.*, preventing PCCs from leaching contaminants into ground- or surface water sources from unlined surface impoundments).

However, it should be noted that EPA has also proposed another "more cost-effective" variation of the Subtitle D approach, called "D prime," in which it would not phase out surface impoundments, but would instead establish and fund an oversight program to regularly monitor structural stability of impoundments with a "High" or

"Significant" hazard potential rating, as listed in the National Inventory of Dams (NID) and determined by criteria developed by the U.S. Army Corps of Engineers (*Federal Register*, 2010, p. 35210). Under this approach, then, passive leaching of contaminants could still occur from PCCs stored in unlined but structurally-stable surface impoundments, but the more dramatic threat of surface impoundment collapse and PCC release, as in the Kingston spill, hopefully would be avoided.

Table A1 of Appendix A presents some of the key differences between the three potential regulatory approaches, including a comparison of estimated annual costs under each subtitle proposal.

PURPOSE OF THIS PAPER

In many ways the 2008 Kingston ash spill was a wake-up call – not simply in highlighting the need for more comprehensive disposal regulations to ensure impoundment integrity and public safety, but also in revealing how thoroughly the Bevill Amendment has been internalized by industry, and how many gaps are present in our knowledge of PCC creation, disposal, and leaching behavior. (To name one example, see Chapter 1 for a discussion of how wildly incorrect even "official" estimated values for historical U.S. coal ash disposal may be.) While EPA has made admirable progress in filling in some of these knowledge gaps by assembling comprehensive records for active PCC disposal sites and by modeling potential leaching rates and related human and environmental risks from these sites, the fact remains that EPA is in many cases the only holder of this information. Thus, the Agency's greatest challenge in relation to the June proposed disposal rule is ensuring that the public comment periods do not become a virtual "echo chamber," in which the datasets underlying its models can neither be verified nor contested due to simple lack of "outside" data to use.

This paper has been written with the intent of contributing precisely this sort of "outsider's perspective" on the current debate about the June proposed rule. By creating my own, simplified calculations to estimate annual PCC production and disposal, and total trace element concentrations within these materials, I hope not only to provide a critical perspective on some of the scientific evidence presented by EPA in support of its ruling, but also to provide "baseline" information which may be useful for other groups wishing to create alternate PCC leaching and risk assessment models to more directly assess the accuracy of EPA conclusions.

These analyses are necessary for several reasons. First, as will be discussed in much greater depth in Chapter 1, there still exist some extremely basic gaps in our understanding of PCCs – in particular, the answer to the fundamental question: *How much coal ash is produced and placed in disposal annually?* Currently, EPA's most recent estimate for PCC disposal is over 20% larger than both commonly-accepted values provided by the Energy Information Administration (EIA) and the American Coal Ash Association (ACAA) (*Federal Register*, 2010, p. 35212; ACAA, 2009a). It is therefore vital to determine with which side the discrepancy may lie, as this will have important implications for ultimate policy decisions. It is my hope that the calculations presented here, basic as they may be, will provide a useful new perspective in resolving this issue.

After a general discussion of the environmental and public health dangers associated with leaching of trace elements from PCCs (particularly heavy metals and radionuclides), Chapter 2 will attempt to "demystify" the primary evidence utilized by EPA in support of the June proposed rule: the leaching risk assessment model, and the demographic model. These are critical portions of the proposed rule which must be thoroughly evaluated during the public comment period, but the complexity of both models (and the amount of technical jargon utilized in "explaining" their methodology and conclusions in *Federal Register*, 2010) will likely discourage non-scientific individuals from understanding them fully. By clarifying both the general conclusions of the models and those areas which require improvement, I hope to make these sections slightly more accessible to this segment of the public.

As a follow-up to these points, whereas EPA risk models evaluate the migration time of contaminants leached from disposed PCCs, it should be noted that these conclusions cannot be verified or challenged by rival models without knowledge of “baseline” trace element amounts contained in PCCs. In Chapter 3 I will calculate these values for coal ash, both in a “worst-case” scenario in which 100% of all trace elements present in whole coal are transferred to ash upon combustion, and in a 1-to-1 correspondence scenario where trace elements are held proportional as whole coal is converted to ash. Particular attention in this chapter will be paid to coal ash disposed in surface impoundments, as these storage areas pose the greatest risk of leaching trace elements into ground- and surface water supplies, and so are under greatest scrutiny by EPA.

Chapter 4 will take these concerns to their most extreme conclusion. There I have attempted to simulate the “worst-case response” to my “worst-case scenario” from Chapter 3 – namely, whether the estimated radioactivity levels present in coal ash placed into disposal annually are high enough to warrant reclassification of coal ash not under Subtitles C or D of RCRA regulations, but under the more stringent NRC regulations, to be treated as a low-level radioactive waste (LLRW). In estimating the potential costs of this “overkill” regulatory scenario, particularly in comparison to the existing costs of NRC LLRW regulation for nuclear power utilities, I hope to place cost estimates from the 2010 EPA proposed rule in a more balanced perspective.

Finally, Chapter 5 will assess what might be next in PCC disposal regulation, briefly surveying some public and industry concerns regarding potential impacts of the June proposed rule on total annual PCC production, usage, and disposal in light of the information discussed in previous sections.

Chapter 1: How Much Coal Ash is Produced by Electrical Generation Activities and Disposed Annually?

INTRODUCTION

It is perhaps a testament to how thoroughly the Bevill Amendment has been embraced by the industry that while the Energy Information Administration (EIA) keeps meticulous records on the mining, distribution, and combustion of whole coal by electric utilities within the United States (*cf.* EIA, 2010a; EIA, 2010c), remarkably little current, accurate information is available for the total annual volume of coal ash produced on a national level by the combustion of this whole coal.

Although the most widely-cited sources for annual production and usage of coal ash and other PCCs are the American Coal Ash Association (ACAA)'s "Coal Combustion Product Production and Use Survey" report series (*cf.* ACAA, 2010b), these reported values cover only a fraction of the industry at best, and thus are not necessarily representative of total PCC production and use in the United States.[3] As noted by EPA, it is likely that annual reports by both ACAA and EIA (2010c) also may have regularly underestimated total national PCC production values because they utilized survey methodologies that excluded smaller-sized electric utility plants with nameplate capacity below 100 MW (*Federal Register*, 2010, p. 35212). When EPA adjusted its estimates to account for these excluded facilities, it found that the amount of PCCs being placed into disposal in 2008 alone increased by 19 million tons, or approximately 20% over previous estimates![4,5] Thus we can see that the answer to even this most basic question varies

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[3] See Chapter 1, Endnote #1, p. 92.

[4] See Chapter 1, Endnote #2, p. 92.

widely according to the source – with obvious implications as the EPA now revises its proposed PCC disposal rule.

This finding led me to believe that while the values provided in the ACAA, EIA, and EPA reports were valuable starting points in understanding the scale of annual PCC production and disposal, there was additional value in performing a back-end "sanity check" to verify their ultimate plausibility, and particularly in gaining another perspective on the possible origins of the 20% discrepancy between EPA-estimated and ACAA-reported 2008 PCC disposal values. My methodologies and results for each of the major calculations performed are described in greater detail below.

METHODOLOGY

General Ash Tonnage Calculations

Approximately 93% of all whole coal mined in the United States in 2008 was utilized for electrical generation purposes (EIA, 2010e, p. 66). However, not all whole coal produces equal amounts of ash upon combustion; this value varies depending on furnace type and combustion temperature, as well as the quality of the whole coal itself (Beck, 1989, p. 286). Furthermore, the precise mix of the whole coal feedstock utilized by coal-fired utilities may vary widely over the course of a year as facilities respond to changes in supply and price. For these reasons, I deemed that a back-end, user-centric approach to calculating annual coal ash production by determining the exact mix of whole coal utilized by each coal-fired power plant in every state over as fine a timescale as possible would be unfeasibly difficult.

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[5] See Chapter 1, Endnote #3, p. 93.

Instead, I adopted a front-end approach: rather than determining the ultimate destination of the coal, I utilized the EIA annual report series on “Domestic Distribution of U.S. Coal by Origin State, Consumer, Destination, and Method of Transportation” (EIA, 2010f) to compile a table of the amount of domestically-produced whole coal that was also domestically-consumed, both by all industries and by the electrical-generation sector alone. By multiplying these values by the average percentage ash content present in coal produced by each respective state, as listed in the EIA annual report series on “Cost and Quality of Fuels for Electric Plants” (EIA, 2010g), I was able to calculate the “unadjusted” amount of ash that could be theoretically generated by the combustion of coal annually in the United States, both by all industries and by the electrical-generation sector alone.

Tonnage of Ash Types Disposed

As expected, because of the datasets I utilized in performing my calculations and because of my own assumptions when making these and subsequent calculations in this section (see “Key Assumptions” and “An Important Note on Datasets” sections below), the “unadjusted” annual ash production values I calculated using the above steps differed from those listed in ACAA annual reports for the 2001-2008 time period (ACAA, 2010b). However, by assuming that the relative proportions of each ash type composing total annual coal ash production in the ACAA surveys were equally applicable to my own calculated totals, I was able to further subdivide my calculations into fly ash, bottom ash, and boiler slag “unadjusted” annual production values. Equally, I assumed that the proportion of production to disposal for each of these three ash types as provided in the ACAA reports were similarly applicable to my own calculations, and so was able to calculate the “unadjusted” amounts of each coal ash type which theoretically could have been produced and disposed in either landfills or surface impoundments annually.

“Adjusting” Tonnages for Realism

I next tried to refine further these “unadjusted” production and disposal values in order to better represent the *actual* amount of each coal ash type captured and placed in disposal annually, in recognition that not all coal-burning electrical generation facilities within the U.S. have equal ash capture rates. Namely, the quantity of ash captured by a power plant is determined not only by the pollution-control technologies it must install to comply with relevant state and federal environmental regulations, but also by the age of the facility itself (*i.e.*, whether it may qualify for more lenient environmental regulations due to the difficulty and expense of retrofitting newer pollution-control technologies).

Thus, we can broadly classify power plants into two main groups. Older “Grandfathered” facilities that are permitted to operate under more lenient environmental regulations have less-effective flue gas filtering technologies, and therefore tend to emit greater amounts of fly ash into the atmosphere. In contrast, the newer “Modern” facilities that are required to abide by more stringent regulatory measures have more advanced emissions-control technologies would emit a smaller proportion of fly ash annually (and so would capture, and dispose of, a larger proportion of produced coal ash annually than their older counterparts).

After establishing an adjustable ratio of “Grandfathered” to “Modern” facilities to represent the overall makeup of the coal-burning power generation sector, and setting similarly-adjustable percentages for annual capture rates of fly ash, bottom ash, and boiler slag at both facility categories, I was thus able to multiply my “unadjusted” theoretical ash type production and disposal values calculated above by these “refining” variables to produce a new “adjusted” dataset which would more accurately reflect true U.S. annual coal ash production and disposal values.

Ash Tonnage per Disposal Method

It is a notable omission in many discussions of PCC safety – particularly those regarding the potential for trace element leaching from coal ash and other PCC materials either stored in or spilled from surface impoundments – that the values most often cited are ACAA-derived statistics for *total* annual PCC disposal, with no mention as to the particular breakdown of this amount into landfill versus surface impoundment categories. Because such information is naturally critical for full assessment of leaching risks, I have made an attempt to determine these values among my own “adjusted” calculations here by assuming that the only available reference point for these values from 2001 (ACAA, 2003a) remained equally-applicable to landfill and surface impoundment disposal ratios for subsequent years. Again, the “Key Assumptions” and “An Important Note on Datasets” sections below provide more complete explanations for my assumptions and the reasoning underlying these choices.

Extrapolating Ash Data to All PCCs

As the final step in comparing EPA’s PCC disposal estimate from *Federal Register* (2010, p. 35212) with that of ACAA (2009a), I first utilized the latter organization’s “CCP Production and Use” report series to determine what proportion of total annual national PCC production and disposal, respectively, was composed of coal ash. By multiplying my existing “adjusted” coal ash calculations by these ratios, then, I was able to “backwards-extrapolate” what total amount of PCCs might have been produced and placed into disposal each year.

KEY ASSUMPTIONS

As previously mentioned, several assumptions needed to be made during these initial steps in order to facilitate both these and future calculations. Because each of these assumptions represents a potential source of error in my calculations, I have listed these below, along with an explanation of my reasoning in each instance. Please note that an additional explanation regarding my choice of certain key datasets also follows in the "An Important Note on Datasets" subsection below.

1. *Despite individual differences in terminology, ACAA, EIA, and EPA production and disposal statistics all refer to the same group of byproduct materials created by coal combustion, and can therefore be compared without need for further qualifiers.*

   As I have described in Chapter 1, Endnote #2, p. 92, it appears that the ACAA and EPA definitions for "coal combustion products" (CCPs) and/or "coal combustion residuals" (CCRs) are strongly similar, and that EIA values as stated in *Federal Register* (2010, p. 35212) are likely to share the same definition of CCPs as provided by ACAA (2009a). Because of this, I here have assumed that the annual production and disposal statistics cited by all three groups do in fact refer to the same byproducts of coal combustion (*i.e.*, what I have termed "PCCs"), and therefore can be considered to be on equal footing for comparative purposes. Although to the best of my knowledge this assumption is correct, due to the complexity of the EIA (2010c) datasets and the limited timeframe in which this project was completed, it nevertheless is possible that I have missed some subtle but important distinction among the three groups' definitions which would invalidate this point.

2. *All coal produced within a given year is also consumed domestically; no coal is exported to foreign countries.*

According to EIA (2010h, p. 2), the United States exported approximately 66 million short tons of coal in 2008; EIA (2010e, p. 12) draws upon a different – and perhaps more accurate – dataset to place the 2008 export amount at more than 81 million short tons. If we add these values to the more than 1 billion short tons of domestically-produced coal which remained in the U.S. for national consumption and distribution in the same year (EIA, 2010e, p. 66), we can see that these values would represent approximately 6.2% and 7.5% of total national production, respectively.

Because I was unable to find a more comprehensive breakdown regarding what quantity of coal was exported to foreign countries from each coal-producing state annually – and because of the lack of consensus regarding more general values, as seen above – for the sake of simplicity I here have assumed that no exports of domestically-produced coal to foreign countries occurred at all during the 2001-2008 timeframe in which my calculations take place.

3. *Conversely, all coal consumed within a given year is domestically-produced; no coal is imported from other countries.*

According to EIA (2010i, p. 12), the United States imported more than 34 million short tons of coal in 2008. EIA (2010d, p. 48) places this value at a lower 27 million short tons, but notes that the origin of an additional 14 million short tons delivered to electric power plants in the same year could not be ascertained. In both cases, these values are still only a small proportion (3.4% and 2.7-4.1%, respectively) of the more than 1 billion short tons of coal produced within the U.S. and distributed for domestic consumption in 2008 (EIA, 2010e, p. 66).

As detailed in Chapter 3 below, some of my later calculations require detailed knowledge of trace element concentrations within coal used for electrical generation purposes annually. Unfortunately, I was unable to find any similar breakdown of trace element concentrations in coal annual deliveries (and, indeed, I doubt that recent and comprehensive studies of this exist at all; hence my reason for trying to extrapolate general state values to answer this question in Chapter 3 below). To preserve internal consistency of my own calculations, I therefore have assumed that all coal burned by U.S. power plants during the 2001-2008 timeframe of my analysis was domestically-produced, and did not include any quantities of coal produced by and imported from foreign countries.

4. *All coal received by power-producers within a given year is also consumed within that year; there is no carry-over of coal between calendar years.*

Although I do not doubt that some small quantities of coal delivered for electrical generation purposes must be carried-over in this fashion (*e.g.*, a large quantity of coal mined and delivered to a power plant on December 31st may not be fully consumed by that facility until January 1st of the new year), this amount cannot be determined from publicly-available EIA reports, which report weekly coal production values but only quarterly consumption values. Nevertheless, this is not likely a significant amount when viewed on the larger scale of total annual

U.S. coal production and consumption, and thus feel safe in making this assumption.

5. *All coal produced by a state produces a flat percentage of ash when burned; this value does not vary by the region within a state in which the coal was produced, or by any technologies or treatment methods utilized by the recipient during processing and combustion of the delivered coal.*

   My reasons for this particular assumption are discussed in greater detail in the section titled "An Important Note on Datasets" below. The brief summary is simply that I was unable to find more detailed values for ash content in domestically-produced coal without restricting my basic dataset in a way that would be detrimental to my efforts to assess the previously-discussed 20% disposal discrepancy between EPA and EIA/ACAA data.

6. *All coal-burning power generation facilities belonging to "Modern" and "Grandfathered" categories, respectively, have equal capture rates for coal ash types – i.e., capture rates vary between the two categories, but not among facilities within the same category.*

   The calculations performed to transform my "unadjusted" production and disposal values per ash type into more realistic "adjusted" values are meant for conceptual purposes only, as any more-comprehensive attempt to represent the precise capture rates for each of the more than 400 coal-burning power facilities in the United States simply was not possible in the limited timeframe in which this paper was being written.

The “default” values I have used for this adjustment process are summarized in Table B3 of Appendix B. For users wishing to experiment further, I have also constructed my spreadsheets to allow fine-scale adjustment of both the Modern/Grandfathered facility ratio as well as the capture rates for each ash type produced by each of the two facility categories.

7. *Proportions of each ash type (i.e., fly ash, bottom ash, and boiler slag) produced and utilized annually by facilities included in ACAA surveys can be extrapolated to all annual coal-burning electrical-generation activities within the U.S.*

As noted in Chapter 1, Endnote #1 on p. 92, PCC production and usage statistics for power-generation facilities surveyed in ACAA annual reports are at best representative of only about half of the total coal-burning power-generation facilities within the United States. For the purposes of these calculations, I have assumed that the number of facilities included in the ACAA data is sufficiently large as to be representative of the sector as a whole, and thus that ACAA annual values are applicable to all annual coal ash type production.

Note, then, that this assumption refers to two broad conditions: first, that the annual percentage of coal ash composed by each of the three coal ash types is representative of the national average over the same time period; and second, that the annual percentage of PCCs composed of all coal ash types is representative o the national average over the same time period.

8. *Proportions of each ash type (i.e., fly ash, bottom ash, and boiler slag) placed in either landfills or surface impoundments in the 2001 ACAA survey can be*

extrapolated to all annual coal-burning electrical-generation activities within the U.S., and have not changed in subsequent years.

Just as for the above point, I have assumed that ACAA data available for the amount of each coal ash type placed into either landfills or surface impoundments for disposal in 2001 (ACAA, 2003a) are equally applicable to all other unsurveyed coal-burning power-generation facilities in the U.S. for that year. However, because 2001 was the only year for which this disposal-type breakdown per coal ash type was recorded in ACAA annual surveys, I have also made the more general assumption that these proportions have not significantly changed within the 2002-2008 time period in which the remainder of my calculations take place.

9. The only storage methods available for coal ash types placed into disposal (i.e., not utilized for a "beneficial" purpose) are landfills and surface impoundments.

In this project, I have utilized only the two general ACAA (2003a) definitions of "wet storage" (what I here refer to as "surface impoundments") and "dry storage" (what I here refer to as "landfills"). Theoretically, two additional categories could be introduced for a more detailed final analysis – one to record the amount of PCCs annually placed into mines for disposal; and another for the use of PCCs on land as unencapsulated fill material, an activity which is recognized by ACAA as "beneficial use" but due to leaching concerns is believed by EPA to be "disposal" (*cf. Federal Register*, 2010, p. 35155). However, due to the limited timeframe of this project, I have been unable to determine what annual quantity of coal ash and other PCCs are utilized or disposed in these two manners annually, and thus have chosen to conform to these two general categories rather than attempting to add extra categories to encompass these activities.

AN IMPORTANT NOTE ON DATASETS

As may be evident from the “Key Assumptions” section above, I unfortunately had to make some fairly large assumptions in the process of calculating annual ash production values; and, similarly, any margins of error made here are equally present in nearly all subsequent calculations seen below, as these truly are the “foundational” values on which the rest of this paper is built. Nevertheless, I concluded that these assumptions were the “lesser evil” in the tradeoff between the level of detail represented and level of reliability present in the various datasets I used to compile my baseline information sets prior to beginning calculations – specifically in relation to my choice of a primary data source for both annual coal production by state and annual coal consumption by the electrical generation industry.

I previously have discussed EPA’s concern that EIA’s exclusion from reporting statistics of small electrical generation facilities with generation capacity at or below 100 MW may have led to underreporting of total national annual PCC production by as much as 20% (*cf. Federal Register*, 2010, p. 35212; and further discussion at Chapter 1, Endnote #3, p. 93). Specifically, the 5-10 million tons of PCCs produced by these excluded facilities may account for as much as half of the 20% discrepancy between reported annual national PCC production values by EPA and EIA, and are themselves a small but still-significant percentage of the total 131-141 million tons of PCCs produced in the U.S. in 2008 (*Federal Register*, 2010, p. 35212; ACAA, 2009a). Again, because my primary goal for conducting the ash tonnage calculations in this chapter was to perform a “sanity check” on EPA’s estimated values for annual PCC disposal from the

2010 proposed rule, I tried to preserve this excluded group of power generation facilities in my own data so as to avoid EIA's "threshold errors" and create a more accurate comparison to EPA's own estimates.

By these criteria, then, the information for annual coal production and consumption presented in EIA (2010f) was the immediately-preferable choice to that of EIA (2010g), simply because the latter succumbed to the "threshold error" by reporting only coal usage by power plants with nameplate capacity equal to or greater than 50 MW (*cf.* EIA, 2010d, p. ii). Although the alternate EIA (2010f) dataset admittedly had its own quirks[6], on the whole this appeared to be more reliable of the two data sources, as to the best of my knowledge it did not appear to exclude any distribution data from its final reporting.

RESULTS AND DISCUSSION

Overview of Tables

A demonstration of the derivation of theoretical (*i.e.*, "unadjusted") annual coal ash production by electrical generation activities was calculated from initial EIA data is shown in Table B1 of Appendix B, with all "unadjusted" annual state ash production totals for the entire 2001-2008 period compiled in Table B2. The variables I have utilized to adjust these numbers to account for varying levels of pollution control (and thus ash production) by industry are documented in Table B3, and the resultant "adjusted" calculations for national coal ash production and disposal values are again compiled in Tables B4 and B5, respectively. Lastly, Tables B6 and B7 display the results of the final

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[6] See Chapter 1, Endnote #4, p. 94.

"backwards-calculation," by which national PCC production and disposal values have been derived from ACAA-documented ratios of coal ash to total PCCs.

Comparisons with EPA and ACAA Data

In all instances, my calculations have returned lower values for production and disposal of both coal ash and PCCs in 2008 than either those which have been reported by ACAA, or those estimated by the EPA in the June proposed rule. Specifically, I have found annual PCC production to be between 122 million short tons ("adjusted" values) and 130 million short tons ("unadjusted" values) for this year; these results are therefore fairly close to the EIA estimate of 131 million short tons and the ACAA reported value of 136 million short tons for the same year, but significantly lower than the 141 million short tons of estimated production given by EPA. Similarly, my calculations for national PCC disposal in 2008 range between 68 and 69 million short tons ("unadjusted" and "adjusted" values, respectively); again, these are fairly close to the ACAA reported value of 75 million short tons, but significantly lower than the 94 million short ton estimate provided by EPA in the June 2010 proposed rule.

As I have mentioned previously, my dataset choice was based on the unwillingness to exclude the coal quantities utilized by small generators, a variable pointedly included in EPA's production and disposal calculations for PCCs in 2008 (*Federal Register*, 2010, p. 35212) which might be at least partially-responsible for the observed 20% divergence between EPA and ACAA disposal values for that year (see again Chapter 1, Endnote #3, p. 93). As I have also recorded in the above sections, however, in the course of my calculations I was forced to make several initial assumptions. Barring for the moment the possibility of error in the EIA dataset itself, I must conclude that these assumptions must be the reason why my resultant calculations trend towards the lower side of the national PCC production and disposal spectrum.

In reviewing the eight assumptions I have provided above, I count two which are irrelevant to the above statistics (*i.e.*, #7 and #8, being both concerned with disposal formats), two which should have no impact on production or disposal results as calculated above (#1 and #3), one which has likely resulted in an underestimate of national disposal statistics (#2), one which has influenced only "adjusted" values (#5), and a final two with uncertain effect (#4 and #6). These last four are explained below:

- *Assumption #2: Imported coal not included in consumption statistics.*

  As mentioned above, the exclusion of imported coal from total U.S. coal consumption could result in a 3.3% "loss" of consumption by EIA (2010i, p. 12) statistics, or between a 2.6% and 3.9% "loss" according to EIA (2010d, p. 48) statistics, depending whether or not coal of "unknown origin" is included in the excluded totals. Although these percentages would not represent an increase of more than about 5 million short tons in my estimates of national PCC production for both "adjusted" and "unadjusted" values in 2008, and about 3 million short tons for PCC disposal, this in both cases would nevertheless account for some of the observed gap between my values and the ACAA estimates (though still far short of the disputed EPA disposal estimates).

- *Assumption #4: Every state produces coal with fixed ash percentage by weight.*

  Clearly, this was a large assumption to make, and could certainly have been a source of error – possibly even a major one. However, without more detailed information about location-specific coal quality in a format which does not exclude small consumers (this last being my reason for rejecting the second dataset I considered using for these baseline calculations), I cannot tell whether any given state's "flat" percentage of ash by whole coal weight was an under- or overestimate, thereby resulting in either lower- or higher-than-actual annual coal ash production values.

- *Assumption #5: "Adjusted" ash statistics accurately represent industry.*

  As described above and detailed in Table B3 of Appendix B, the values I utilized to "adjust" my final calculations were simply placeholders to demonstrate how utilization of various emissions-control technologies might affect coal ash production and disposal values on an industry-level scale; being user-adjustable, these were not necessarily meant to be realistic. The fact that the "adjusted" PCC production values differ so greatly from the "unadjusted" values may indicate that I was too aggressive in some of my initial variable assignments – but again, this is easily-correctible via the user-defined fields for "default" adjustment values in my spreadsheets (as noted above in the "Key Assumptions" section, p. 24-25).

- *Assumption #6: ACAA ratios of coal ash to PCCs, and coal ash types to total coal ash, accurately represent annual production and usage statistics.*

  Finally, I must note that although many of my calculations were based on the relationships among PCC categories listed in the annual ACAA reports (namely, the percentage of coal ash composed by each coal ash type, and the more general percentage of all PCCs composed by all three coal ash types), the fact that these reports themselves represent only about half of the total coal-consuming electricity generators in the U.S. means that they are not necessarily representative of national statistics. As I have no other data source for comparison, however, I do not know with certainty whether ACAA reported data is necessarily accurate in this respect.

Historical Ash Disposal Values

The results of my calculations pertaining to the three coal ash types proved additionally informative from a different angle. To quote EPA (2010a, p. 3):

> The American Coal Ash Association (ACAA) estimates that between 100 million and 500 million tons of fly ash have accumulated in U.S. landfills since the 1920s, when the disposal of fly ash in landfills began. It is unclear, however, how much of this fly ash is available for beneficial use.

Here, “availability” refers to fly ash that is easily-recoverable – that is, materials which are stored in dry format in landfill disposal sites, rather than in wet storage at surface impoundment disposal sites, for which the costs of dewatering in preparation for reuse are not currently economically-feasible. However, unless the above-cited values are the result of a particularly-egregious typo on the part of either EPA or ACAA, these values are almost certainly significantly lower than reality. My calculations for the 2001-2008 period alone estimated national landfill disposal of fly ash to be more than 199 million short tons, and as demonstrated in Chapter 1, Endnote #5 on p. 95, even under extremely conservative initial conditions, it is in fact quite possible that annual fly ash disposal in landfills could easily have exceeded this estimate even within the past forty years, rather than over the last ninety, as EPA (2010a) suggests above.

Thus, it appears that there are not one but two “elephants in the room”: not only does EPA’s 2008 PCC disposal estimate continue to appear questionably-high in comparison to ACAA values and my own calculations (which, admittedly, appear to be on the conservative end of national production and disposal estimates for coal ash); but EPA currently does not appear to possess any reliable information on what quantity of coal ash and other PCCs may be present in closed landfills or surface impoundments (see again the EPA (2010a) example above). Considering that these older disposal units are often unlined, and thus present a higher leaching risk than newer, lined units (see Chapters 2 and 3 below), it would be advisable for EPA to look more closely into this issue – in particular, to determine a) what quantity of these older disposed materials are stored at facilities in which other disposal units are still-operational (and thus theoretically able to be monitored or otherwise modified under the June proposed rule), and b) whether leaching rates (and associated threats to public and environmental health) vary according to the age of the PCCs themselves, an area which to my knowledge has not yet been studied.

Chapter 2: What are the Potential Problems Posed by Trace Elements in Disposed Coal Ash?

INTRODUCTION

One primary issue lies at the heart of the EPA proposed rule: the potential leaching of trace elements from coal ash and other PCCs that are placed in disposal – particularly those in surface impoundments. This chapter will discuss the reasons for this concern. After providing a general overview of the scientific evidence for leaching and known ecological consequences, I will summarize EPA's specific concerns regarding the number of unlined and structurally-unsound PCC disposal units in the United States, as well as the demographic reasons for particular concern in the field of environmental justice.

TOXICOLOGY OF COAL ASH: AN OVERVIEW

As mentioned previously, whole coal naturally contains a variety of trace elements; these include heavy metals such as arsenic and mercury, as well as radionuclides and associated decay products. The ultimate concentration of these trace elements in coal ash and other PCCs is therefore initially dependent on the depositional history of the coal bed from which the fuel-coal was originally mined. However, the trace element levels may be further affected by both the combustion and end-point ash removal technologies utilized by the power plant. Although this latter process largely remains poorly understood (see my discussion regarding "concentration factors" in Chapter 3 and associated endnotes below), trace elements contained in coal typically exhibit one of four general behaviors during combustion activities: they may 1) become incorporated into vitreous slag particles, 2) volatilize and subsequently condense or adsorb on the surfaces of fly ash particles, 3) concentrate approximately equally in both fly ash and slag, or 4) remain almost completely in the gas phase, binding rarely to either material (Klein *et al.*, 1975, p. 975-976).

In the subsections below, I will briefly explore the role that these potential end-states may play in the ultimate availability of PCC-contained trace elements for leaching. After beginning with EPA's primary focus – the potential environmental and public health impacts associated with leaching of several elements known to be enriched in fly ash, particularly heavy metals – I will examine the more rarely-studied question of erosion rates for vitreous particles and the potential for release of the encapsulated radionuclides and other trace elements.

Causes of Leaching and Reasons for Concern

Defined most broadly, two potential scenarios may result in leaching of trace elements from disposed PCCs. The first scenario is that of passive *in situ* leaching, in which trace elements are gradually leached by exposure to either rainwater or the aqueous environment of a surface impoundment; if the disposal unit is not lined with an impermeable membrane, this leachate will over time migrate downwards and enter the water table. The second, more dramatic scenario is a situation like the 2008 Kingston ash spill – an uncontrolled release of PCC materials into local waterways occurs, exposing ash particles to a more complex set of physical and chemical interactions which may accelerate the pace of leaching beyond the *in situ* "baseline" rates.

Because of the unpredictable nature of PCC spills and other uncontrolled releases, "*ex situ*" studies of leaching behavior of the spilled materials will almost always be imperfect and incomplete, due both to the inevitable lag in response time between the occurrence of a spill and the earliest sediment and water quality studies, and to the frequent lack of any pre-spill measurements in the affected areas to which to compare results after sampling. Thus, the majority of our current knowledge about the leaching behavior of PCCs necessarily comes from *in situ* studies.

To date, EPA has recognized 27 cases of proven damages to ground- or surface water as a result of leaching and/or improper PCC disposal, and has identified an additional 40 potential damage cases (*Federal Register,* 2010, p. 35233-35239). On review of the 27 proven damage cases, I would classify only 17 as being true instances of *in situ* leaching as defined above (*i.e.,* those entries listed at *Federal Register,* 2010, p. 35234-35236), with the remaining 10 cases classified as varying degrees of "*ex situ*" scenarios, as they primarily involve improper disposal or discharge of PCC materials into local lakes or waterways (*Federal Register,* 2010, p. 35237-35239). Taken together, these damage cases demonstrate that *in situ* leaching is indeed occurring in at least some areas across the country, and that the accumulation of trace elements in groundwater can occur quite rapidly. Perhaps the most notable example of this is in Gambrills, Maryland, where fly ash and bottom ash disposed in unlined sand and gravel quarries in 1995 had already resulted in observed MCL exceedances for various trace elements in numerous off-site drinking water wells by 2006-2007 (*Federal Register,* 2010, p. 35236).

As EPA correctly notes, however, even a lined surface impoundment with little to no leakage can contain sufficiently high levels of dissolved trace elements in its waters to impact the health, behavior, and development of organisms living in its immediate vicinity (*e.g.,* amphibians, fish, plants). If more mobile predators exist to prey upon these populations (*e.g.,* wading or fishing birds, or roaming herbivores), these trace elements may then bioaccumulate among multiple levels of the trophic pyramid, potentially affecting not only local but also migratory populations of multiple species.

Potential Environmental and Public Health Impacts of Heavy Metal Leaching

As listed on Table 2.1 on the next page, “Class II” elements are those which, upon combustion of the whole coal in which they are contained, tend to bind to and concentrate on the surfaces of fly ash particles. Because they are bound to the surface rather than encapsulated within the particles, they are therefore at particular risk of being leached readily upon exposure to water, with associated toxicological ramifications which can affect ecosystem health on both short- and long-term timescales. Particularly as several of these Class II elements are also heavy metals known to cause a variety of human health problems when found at elevated concentrations, EPA in the June proposed rule is quite clear in its assessment of leaching of these elements as a true threat, and its resolve to reduce their release from PCCs in disposal sites.

Much of the evidence EPA utilizes to support its stance in the June proposed rule comes from the conclusions of its comprehensive risk assessment modeling, which estimated national leaching rates from PCC disposal sites and the potential severity of human and ecological impacts caused by exposure to each of 41 constituents which might leach from these disposed materials (EPA, 2010b). Both for reasons of space and because of the complexity of the EPA analysis itself, I will not attempt to discuss the study in full here, but will instead note a few highlights. In modeling estimates for both 50th percentile nationwide exposure risks to leached contaminants (also called “central tendency estimates”) and 90th percentile risks (“high-end risk estimates”) over a 10,000-year simulated leaching duration, EPA found multiple instances in which modeled constituents exceeded the risk criteria for listing as a hazardous waste.[7] These results are summarized in Table A2 of Appendix A; note particularly that modeled leaching rates

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[7] See Chapter 2, Endnote #1, p. 96.

Table 2.1: General classification scheme describing trace element behavior upon combustion of whole coal, as derived by Klein et al. (1975) from PCC sample analysis at a TVA power plant with cyclone furnaces.

Behavioral Classification	**Elements in Each Category**
Class I	Al, Ba, Ca, Ce, Co, Eu, Fe, Hf, K, La, Mg, Mn, Rb, Sc, Si, Sm, Sr, Ta, Th, Ti
Class II	As, Cd, Cu, Ga, Pb, Sb, Se, Zn
Class III	Br, Cl, Hg
Intermediate between Classes I and II	Cr, Cs, Na, Ni, U, V

Notes:

As per Klein *et al.* (1975, p. 975):

- **Class I** elements are those which are readily incorporated into slag (*i.e.,* are primarily found encapsulated in dense, vitreous slag particles rather than bound to the surfaces of fly ash particles).
- **Class II** elements are those which are not found in significantly elevated levels in either slag or fly ash, but rather appear in approximately equal concentrations in both materials.
- **Class III** elements are those which “remain essentially completely in the gas phase,” volatilizing upon combustion and generally binding neither to slag nor to fly ash particles.

All behavioral classifications from Klein *et al.* (1975) above were based on the analysis of whole coal, fly ash, slag, and combustion gases in January 1972 and August 1973 at the T.A. Allen Steam Plant in Memphis, Tennessee. More information on the power plant itself and a detailed description of the sampling methodology can be found in Bolton *et al.* (1973) and Bolton *et al.* (1974).

Source:

Klein *et al.* (1975)

from all types of surface impoundments have been predicted to result in extremely high hazard quotient (HQ) values for organisms in direct contact with contaminated water and sediment.

The potential for heavy metals to cause moderate to severe developmental, neurological, and reproductive problems in a variety of organisms is well-documented in the scientific literature. A fairly comprehensive review of field studies on the effects of aquatic disposal of PCCs (*i.e.*, leaching from surface impoundments, or from materials spilled or disposed directly into surface water bodies) may be found in Rowe *et al.* (2002). For a brief representative selection of studies on heavy metal exposures in specific groups of organisms, see also Hopkins *et al.* (2004) on benthic fish, Bryan *et al.* (2003) on birds, Cherry *et al.* (1979) on invertebrates, Hopkins *et al.* (2006) on toads, and Nagle *et al.* (2001) on turtles.

It should be noted that such species-specific analyses of PCC leaching hazards may risk losing sight of an important indirect exposure pathway: bioaccumulation. In this manner, even predators with little to no daily exposure to leachate-containing surface waters or sediments could accumulate high tissue levels of heavy metals after preying on organisms with much larger daily exposures, and thus with higher "baseline" tissue levels of these contaminants (*e.g.*, an osprey feeding regularly from a fish population living within a surface impoundment, or a nearby lake contaminated by leachate from a PCC disposal unit). Thus, effects of heavy metal accumulation by exposure to leachate could extend not only to local predators, but also to transient populations, such as migratory waterfowl.

Perhaps one of the most comprehensive studies in this area is that of Cherry *et al.* (1984), an eight-year survey of the ecosystem effects of exposure to leachate generated by regular discharge of coal ash "slurry" from surface impoundments directly into a stream/swamp confluence (these surface impoundments essentially functioning as "staging areas" rather than permanent disposal units, as is most common today). The researchers found that aquatic pH of the surface impoundment would become dramatically more acidic upon addition of new coal ash to the unit, with levels reaching as low as pH 3.5. Although pH of the impoundment waters would return to equilibrium (*i.e.,* the prevailing "neutral" pH of 7.4) after a period of recovery, exposure to this acidic "slurry" prior to pH stabilization often appeared to promote tissue uptake of heavy metals in a variety of organisms (and thus also the bioaccumulation rate of these contaminants) at higher rates than those observed for a neutral-pH mixture. Thus, there are actually four factors affecting trace element uptake and bioaccumulation rates: the leaching rates of trace elements as affected by pH; the physiological and behavioral differences among organisms exposed to the leachate; the rate at which PCC disposal generates pH "spikes" in the leachate produced; and the chemical characteristics of the PCC material itself upon exposure to water, regarding release of the H^+ ions necessary to create this initial "spike" in pH.

More research is needed at a finer timescale than the Cherry *et al.* (1984) analysis to determine how long- or short-lived might be this initial pH "spike" upon PCC disposal, and the extent of any associated "accelerating" effects of pH in bioaccumulatory processes. To the extent of my knowledge, no subsequent studies have ever elaborated on this observed relationship between PCC disposal and heavy metal content in leachate, but this could prove a particularly valuable supplement to the EPA (2010b) report, which focuses on long-term impacts of leaching only.

Radionuclides: A Neglected Variable?

Historically, the majority of scientific research performed on the environmental and public health effects of exposure to trace elements leached from PCCs has focused primarily on heavy metals. In contrast, comparatively little research has been conducted on similar effects for leached radionuclides, despite the fact that elemental uranium and thorium are often found in whole-coal concentrations that are similar to those of several heavy metals of greatest concern to EPA (see next chapter, and particularly Table C5 of Appendix C). Furthermore, the few studies that do exist in this area have primarily focused on the public health risks posed by inhalation of particulate matter – that is, the relatively small percentage of fly ash (<1-5% of total annual production) which is not retained by pollution-control technologies at power plants, but instead are emitted to the atmosphere (*cf.* Van Hook, 1979; McBride *et al.*, 1978). As such, these analyses necessarily ignore the much larger amount of fly ash and other PCCs that are captured and placed into disposal annually, and, by association, the potential impacts of the radionuclides contained therein.

The lack of research interest in this area may be in part attributable to studies that have disputed the leaching capability of some radioactive trace elements. As per Klein *et al.* (1975), thorium was originally classified as a Class I element (*i.e.*, an element which was incorporated primarily into slag rather than fly ash), whereas uranium was of "intermediate" classification, concentrating in both fly ash and slag. Subsequent studies have found some evidence of uranium binding to surfaces of fly ash particles, particularly those with diameters <1 μm (USGS, 1997; Beck, 1989), but determined that the majority of elemental uranium in fly ash remain sequestered in the hollow centers of these glassy particles (Zielinski and Budahn, 1998; USGS, 1997). Because the vitreous composition of fly ash and slag particles generally is thought to make them relatively resistant – at least over the short-term scale – to erosive activities that would release encapsulated trace elements (*cf.* Tracy and Prantl, 1985), it is perhaps no surprise that researchers have chosen to focus on the leaching dangers of surface-bound Class II elements instead.

The question remains, however, whether this assumption regarding the resistance of vitreous particles to erosion over the short-term is necessarily accurate. Although USGS (1997, p. 4) acknowledges that "extremes of either acidity (pH<4) or alkalinity (pH>8) can enhance solubility of radioactive elements," to my knowledge no formal studies have yet been performed to determine how the variations in pH over the lifetime of an active PCC disposal site (as per Cherry *et al.*, 1984) might affect leaching rates of trace elements. Namely, it is unclear whether the highly-acidic conditions created by chemical reaction of PCCs in the presence of water can result in significant dissolution of vitreous ash and slag particles, thereby causing release of encapsulated trace elements. In the absence of field or laboratory studies designed to investigate long-term leachability of radionuclides from PCCs under environmental conditions, no conclusions regarding the environmental and public health impacts of such erosion-caused leaching can be drawn. Equally, however, the presence of such effects cannot be discounted until further research is conducted (Hassett, 1994; Carlson and Adriano, 1993; Dudas, 1981).

LINERS AND STRUCTURAL CONCERNS

As described in *Federal Register* (2010) and summarized in Table A1 of Appendix A, in its June proposed disposal rule EPA is pushing strongly for a nationwide "phase out" of surface impoundments, noting that the general lack of synthetic liners and questionable structural stability of many existing impoundments together pose a potentially-significant leaching risk, and are therefore a hazard to public health and the environment. Only in dry form rather than an aqueous solution, EPA believes, can PCCs be effectively managed in a manner that reduces these leaching risks.

Particularly in areas where the underlying soils are very permeable, EPA recommends that landfills be lined with synthetic liners to prevent leaching of trace elements into ground- and surface water supplies. The presence of a liner is even more vital for surface impoundments, where the total immersion of PCCs in water is likely to increase the rate at which trace elements are leached from particle surfaces in comparison to dry-storage landfill disposal facilities. In the absence of an impermeable barrier to movement, the water in the storage ponds will naturally interact with the underlying rock formation, potentially forming a contaminant plume as leached trace elements are dissolved and drawn along.

Although EPA and the U.S. Department of Energy (DOE) have noted that a larger proportion of "new" disposal facilities constructed or expanded between 1994 and 2004 possess liners and groundwater monitoring programs than in decades past, leaching remains an important concern for both new and old facilities. Even among the 56 "new" units constructed in this period, a significant proportion (29% and 17% among landfills and surface impoundments, respectively) possess clay liners, which have been shown to be less-effective at halting leachate movement than synthetic liners (DOE/EPA, 2006, p. 33). Furthermore, while these new units indeed demonstrate high rates of liner adoption, they together represent only a small fraction of the total estimated 300 landfills and 584 surface impoundments currently in use nationally – not to mention the unknown number of "legacy" PCC disposal sites no longer active (*Federal Register*, 2010, p. 35151). Of the operational disposal facilities surveyed by EPA in 2004, it was estimated that 31% of landfills and 62% of surface impoundments lacked any manner of liners, and that 10% of landfills and 58% of surface impoundments did not have groundwater monitoring programs (*Federal Register*, 2010, p. 35151).

In addition to this general leaching risk from unlined landfills and surface impoundments, EPA is also concerned about the structural stability of these units, and particularly how many surface impoundments might be prone to a similar catastrophic failure similar to the Kingston fly ash spill. To this end, EPA has requested information from all known active PCC disposal facilities in the United States. As of mid-August 2010, it had received responses from 228 facilities containing 629 disposal units. Thus far, hazard potential ratings have been assigned to 200 of these units according to criteria developed for the National Inventory of Dams (NID), providing, if not a complete picture of the industry, at least a revealing snapshot of its status. EPA (2010d) rated 71 of the 200 units (36%) as having “significant hazard potential,” indicating that dam failure would not likely result in loss of human life, but could result in economic losses and environmental damage, in addition to other impacts. An additional 50 units (25%) were assigned a “high hazard potential” rating, indicating that dam failure would “probably cause loss of human life,” as well as economic, environmental, and other impacts. In a series of followup assessments by contractors to evaluate the respective stabilities of these 121 most-problematic sites, EPA did not find any sites of “unsatisfactory” quality (*i.e.*, a safety threat requiring of immediate attention), but did determine that 16 (13%) were of “poor” structural quality, and meriting of further follow-up attention (EPA, 2010e).

DEMOGRAPHIC CONCERNS

Initial Findings of EPA Analysis

In addition to the above concerns regarding potential water supply contamination from uncontrolled PCC releases, EPA notes that, in a more general context, regulatory attempts to prevent trace element leaching from PCC disposal sites nationwide must also consider demographic traits of those communities living in closest proximity to existing storage sites, which make them perhaps uniquely vulnerable to leaching occurrences.

In the June proposed rule, EPA reports the results of its demographic analysis, performed for 464 of the 495 U.S. power plants that utilize coal for electrical generation purposes (*Federal Register*, 2010, p. 35227-35228).[8] By comparing demographic statistics from the 2000 U.S. Census for the zip codes in which each power plant was located (totaling approximately 6.08 million people across 47 states) to the averaged "baseline" statistics of each respective state, EPA was able to broadly determine which power facilities might have most "at-risk" populations, with either high percentages of children (who are particularly sensitive to chemical exposure, especially during early development), or low-income and minority groups (who likely would be least able to afford healthcare costs, and/or might be unable to relocate in the event of an ash spill or other leaching event resulting in water contamination). The results of the demographic analysis may be summarized as follows:

- *Distribution of child populations near coal-burning power plants:*

  Nationally, the "benchmark" proportion of each state's population composed of children (*i.e.,* any individual under the age of 18) ranged from

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[8] See Chapter 2, Endnote #2, p. 97.

21.5% (Maine) to 30.9% (Utah). Nationally, the percentage of total U.S. population composed of children was 24.7%.

Approximately 1.54 million children lived within the ZIP codes of all 495 coal-burning power plants identified by EPA, comprising 25.4% of the total population of these ZIP code areas – slightly higher than the national average.

Comparing the population distributions within the ZIP code of each power plant to its respective state benchmark (a "plant-by-plant" comparison), EPA found that 310 facilities (63%) were located in areas with child populations exceeding the statewide percentages, while 185 facilities (37%) were located in areas with child populations below the statewide benchmarks.

Comparing the population distributions within the ZIP code areas of all power plants within a single state to the respective state's benchmark distributions (a "state-by-state" comparison), EPA similarly found that populations living near power plants in 27 of the 47 states (57%) contained a disproportionately high percentage of children.

- *Distribution of minority populations near coal-burning power plants:*

Statewide benchmark percentages for percentage of population composed of minority groups ranged from 3.1% (Maine) to 75.7% (Hawaii). Nationally, the percentage of total U.S. population composed of minorities was 24.9%.

Approximately 1.32 million minorities lived within the ZIP codes of all 495 coal-burning power plants identified by EPA, comprising 21.7% of the total population of these ZIP code areas – lower than the national average.

In a "plant-by-plant" comparison, EPA found that 138 facilities (28%) were located in areas with minority populations exceeding the statewide benchmark percentages, while 357 facilities (72%) were located in areas with minority populations below the statewide benchmarks. This indicates that relatively few facilities in the U.S. have a disproportionately high percentage of minorities in the surrounding populations.

Although the "state-by-state" comparison found that populations living near power plants in 24 of the 47 states (57%) contained a slightly disproportionately higher percentage of minorities, EPA notes that this approach does not include "numerically weighing of state plant counts or state surrounding populations" as in the other numerical comparisons, and therefore results in a slightly different value (*Federal Register*, 2010, p. 35230).

- *Distribution of low-income populations near coal-burning power plants:*

Statewide benchmark percentages for percentage of population composed of low-income individuals ranged from 7.3% (Maryland) to 19.3% (New Mexico). Nationally, the percentage of total U.S. population composed of low-income individuals was 11.9%.

Approximately 0.8 million low-income individuals lived within the ZIP codes of all 495 coal-burning power plants identified by EPA, comprising 12.9% of the total population of these ZIP code areas – slightly higher than the national average.

In a "plant-by-plant" comparison, EPA found that 256 facilities (52%) were located in areas with low-income populations exceeding the statewide benchmark percentages, while 239 facilities (48%) were located in areas with low-income populations below the statewide benchmarks. This indicates that a slightly disproportionately higher distribution of low-income individuals may be found in close proximity to coal-burning power plants than the national average.

In a “state-by-state” comparison, EPA similarly found that populations living near power plants in 29 of the 47 states (62%) contained a disproportionately high percentage of low-income individuals.

Concerns Regarding Accuracy of EPA Analysis

The above analyses are interesting public health micro-studies unto themselves, and it is encouraging to see EPA’s research into and inclusion of these important – but rarely discussed – matters of environmental justice in the June proposed rule. Nevertheless, I must express my concern as to the methodology EPA has chosen to perform this set of demographic analyses.

As clearly shown in Appendices M and N to the June proposed rule (EPA, 2009a, p. 285-315), EPA has based its analysis on the locations of the PCC *generating* facilities (*i.e.,* the coal-fired power plants) rather than the far more relevant matter at hand: the location of the storage and disposal facilities. While the majority of PCCs are indeed stored in on-site facilities by the power plants which produced these materials, EPA’s analysis ignores the 17% of facilities (84 of the 495 coal-burning power plants tracked by the agency) which store PCCs in off-site commercial facilities (*Federal Register*, 2010, p. 35227-35228). Furthermore, by incorrectly assuming that all PCC material is disposed in the same state in which it was produced (*cf.* TVA, 2010c, p. 4 for one example of cross-state waste transport), this analysis does not always provide an accurate representation of the materials’ final resting place, and thus misrepresents the actual potential leaching threat for local populations.

Additionally, it should be noted that EPA's demographic analysis also includes 28 power plants (6% of the national total) which do not store PCCs at all, but are able to fully utilize all annual PCC production for beneficial purposes such as creation of Portland cement (*Federal Register,* 2010, p. 35227-35228). Thus, while inclusion of these facilities in the demographic analysis may be useful in the context of a scenario in which the final EPA disposal ruling results in such irreversible stigma that beneficial use of PCCs by industry is highly discouraged, and perhaps halts entirely (*i.e.*, category "B" for cost estimates in Table A1 of Appendix A), this again is not an accurate portrayal of the *current* disposal situation, and similarly may be skewing EPA's state demographic analysis.

Finally, while EPA's demographic analysis clearly relies on ZIP codes for ease and speed of analysis, this may not necessarily be an accurate representation of population distribution near the PCC-generating facilities: a power plant near the boundary of two ZIP code areas, for example, may be in closer proximity to populations in the neighboring ZIP code than those in the one in which the plant is formally situated. A more accurate and informative method of analysis instead would be to retabulate demographic distributions based on radial distance from PCC disposal sites, with particular attention paid to what proportion of total population within each "ring" segment lives in close proximity to waterways into which surface impoundments could potentially spill, and what proportion of the population relies on private wells for drinking water, and is thus at greatest exposure risk in cases of trace element leaching.

Unfortunately, while the material presented in the EPA (2009a) appendices supplementing the June proposed rule is extremely detailed in many other respects, EPA either does not have information on or has not made publicly available any details regarding ash storage sites. Thus, while it is not currently possible to provide a "revised" demographic analysis, I very much believe that access to such a dataset is necessary to more clearly understand the disposal controversy – and, in the process, to “fact-check” EPA’s proposed rule, should any out-of-state disposal facilities prove to significantly impact aspects of the demographics presented in *Federal Register* (2010).

Chapter 3: What Volume of Trace Elements are Present in Coal Ash Produced by Electrical Generation Activities?

INTRODUCTION

From the statistics cited in Chapter 2, it should be quite clear why EPA believes that leaching (either in the case of natural percolation from unlined or clay-lined disposal units into ground- and surface water supplies; or in the case of PCC release during a spill event resulting from containment wall failure) is a legitimate environmental and public health concern, particularly for surface impoundments versus the less failure-prone dry landfills. Nevertheless, what is less-clearly understood in the scope of the current debate is precisely *how much* of a risk leaching may currently pose – that is, what quantity of trace elements is contained within PCC material placed in disposal annually and is therefore theoretically available to leaching activities.

To the best of my knowledge, this particular variety of analysis has not yet been performed, or at least the relevant results have not yet been disclosed clearly and publicly. Perhaps the best information currently available is the dataset utilized by EPA (2010b) to model national risk assessment from leaching rates, but even this may not necessarily be complete. While EPA states that the approximately 140 disposal sites from which samples were taken "do represent the national variability in CCW [coal combustion waste] characteristics" for PCCs disposed at these facilities annually (EPA, 2010b, p. 2-2), this nevertheless is only a fraction of the 629 total disposal sites known to EPA at this time. Particularly considering that market forces may result in a coal-fired power plant having a different "mix" of fuel coal from year to year, with all resultant implications for trace element levels present in this fuel coal, the matter of *temporal* relevance of samples taken from the 140 disposal facilities should also receive some critical consideration and discussion.

Along the same line of thought, while EPA's choice to collect samples from only active or recently-closed facilities is understandable in light of being most relevant to the immediate goals of its proposed disposal legislation, it is unclear whether this data on trace element content or the model's conclusions regarding leaching rate and risk potentials can be accurately extrapolated to older landfills and surface impoundments, for which these factors may be quite different (*e.g.*, if the majority of the trace elements once contained within the PCCs may have already leached out many years earlier. In this case, the EPA model would overestimate both trace element content and leaching rate for materials contained within this facility).

Without more sampling and research to confirm EPA's claim of a nationally-representative dataset – and particularly in the absence of scientific studies which would evaluate whether trace element content and *in situ* leaching rates for PCCs vary by age of the disposed material – it is difficult to evaluate the accuracy of EPA's modeled risk assessment. Nevertheless, such verification by independent parties is an essential step during this initial public-comment period for the June proposed rule, particularly as EPA acknowledges that its own understanding of the factors influencing leachate generation (namely, the importance of considering pH and infiltration conditions in sampling and calculations) is still very much evolving, and far from complete (*cf. Federal Register*, 2010, p. 35141-35142).[9]

Thus, due to the difficulty of attempting to determine what is an "average" trace element content in PCC when concentrations can vary so widely among individual power

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[9] See Chapter 3, Endnote #1, p. 97.

facilities – and likely among PCC batches produced by the *same* facility, as output demand and fuel-coal sources vary over time – I decided to take an alternate approach in my own calculations. Rather than repeating EPA's modeling to assess plausible leaching rates and downstream risk, I instead work from upstream, first by calculating how trace elements from fuel-coal might be concentrated in ash during combustion, and then by utilizing the baseline calculations from Chapter 1 to find the amount of certain trace elements of concern present in PCC disposal units nationwide. During the second step, data was processed under two different scenarios: first, a "worst-case scenario," in which all of trace elements present in whole coal are concentrated in coal ash, with no losses during or after combustion; and second, a "zero concentration factor" (ZCF) scenario, in which produced coal ash retains the same overall proportion of trace elements as was originally present in whole coal (*i.e.*, a 1:1 correspondence scenario between whole coal and coal ash).

It is precisely because the EPA leaching risk modeling assessment is such a critical portion of the rationale behind the June proposed rule that it cannot be allowed to go unverified. By building a more complete picture of the relative toxicity of PCC materials present in national disposal sites, I hope that this information will aid public understanding of EPA's public- and environmental health-based reasons for advancing this rule, as the concept of trace element tonnages should be more easily accessible than the intricacies of EPA's complex leaching risk modeling assessment. More importantly, however, I hope that having such "baseline" information available will encourage debate and develop alternate points of view on EPA's risk model through construction of new leaching risk simulations in the near future.

METHODOLOGY

Adapting Regional Data to Individual States

As I have mentioned previously, the geologic history of a coal formation may produce significant variations in trace element concentrations of whole coal from one region to the next; these differences, in turn, are reflected in the chemical composition of coal ash and other PCCs produced during combustion. Therefore, in this next step I utilized state-specific annual averages for trace element concentrations in coal produced by each state, in order to ensure that a "baseline" value existed prior to the facility-specific variations in the coal combustion and emissions-capture process which might result in PCCs of differing "quality" even among power plants consuming equal amounts of fuel-coal.

Although now somewhat dated, the 1976 U.S. Geological Survey (USGS) report by Swanson *et al.* (1976) contains a wealth of information about minimum, maximum, and geometric mean concentrations for various elements in coal formations across the United States. As described in the notes to Table C1 of Appendix C, because not all coal-producing states utilized in my calculations from Chapter 1 had been surveyed in the Swanson *et al.* report, I assigned the "missing" states to the nearest "province" category which appeared to best approximate the geologic history of the coalbeds. For two other instances in which states were assigned to more than one "province" by Swanson *et al.*, I averaged mean trace element concentrations from both of the assigned provinces to produce a single value per element. Thus, with the single exception of Washington state, for which the number of available samples was too small to confidently use in national trace element calculations, I was able to match all coal-producing states which had been used in the PCC calculations in Chapter 1 with corresponding average trace element values for the whole coal each state produced.

Trace Element Tonnages per Ash and Disposal Type

After the above manipulations, it was a simple matter to multiply the annual coal production values per state by the region-appropriate geometric means (in ppm) for 28 trace elements documented by Swanson *et al.* (1976) to calculate trace element concentrations for all domestically-produced whole coal. Multiplying these values in turn by the annual coal ash production and disposal statistics derived in Chapter 1, I was thus able to obtain tonnage values for these trace elements at state and national scales, both for all whole coal domestically-produced and consumed by electrical generation activities annually (*i.e.*, the "worst-case" leaching scenario, assuming that 100% of these trace elements were transferred to coal ash during combustion), as well as for the three coal ash types (under a scenario in which trace element levels are neither lost nor concentrated in the conversion of whole coal to coal ash – see section on "Trace Element Concentration Factors" below). The coal ash type values then were further subcategorized by production year, coal ash type, and – most importantly – by disposal method (*i.e.*, landfills versus surface impoundments).

Trace Element Concentration Factors

Recall that in the previous chapter, original coal ash production and disposal values were "adjusted" by noting that older, less-efficient ("Grandfathered") coal-burning power plants would have lower ash capture rates – and thus comparatively greater annual coal ash production rates – than newer ("Modern") coal facilities following more stringent pollution-abatement regimes. In this chapter, I created an additional set of user-adjustable variables, such that one could specify the precise percentages of each element retained in each coal ash type by both Grandfathered and Modern power facilities, and thus create more accurate state and national trace element calculations.

The “default” state of the above calculations assumes a strict proportional transfer of trace elements from whole coal to coal ash, a state which I have here termed “ZCF,” or “zero concentration factor”: although the *volume* of material decreases as approximately 10-15% of whole coal by weight is retained as coal ash, with the remainder either lost to combustion processes or otherwise transformed, I assume that the proportion of trace elements in either the initial or produced materials remains constant throughout (*i.e.,* there is neither loss nor enrichment of trace elements from one step to the next). Chapter 3, Endnote #2 on p. 98 provides two completely-fabricated examples of how such variables might function in controlling for loss or enrichment at an element-specific level, but as I have discussed there, due to general lack of rigorous, large-survey-sample surveys of trends in trace element concentration in relation to whole coal geographic origin and facility technology, I did not feel it was appropriate at this time to impose my own values on this process.

KEY ASSUMPTIONS

Because the majority of values for calculations in this chapter were taken from Chapter 1, the same set of caveats applies here as well. Two additional assumptions were made to process the Swanson *et al.* (1976) data, however, both of which are documented below.

1. *For any given state, all coal produced has the same trace element content, regardless of year or mining location.*

Trace element and ash content of whole coal differ by geologic history of the formation, a factor which may vary as widely within a single state as between adjacent states. Although with more time I likely would have been able to determine from EIA records annual, location-specific data for coal production at sites within states, due to the age of the Swanson *et al.* (1976) data set, I did not feel comfortable in attempting to extrapolate the trace element values in a similar manner. Thus, I assumed that trace element values are fixed both in space and time, being a static average for an entire state's production at any given time.

2. *All trace elements present in whole coal are transferred to coal ash; no elements are removed by pre-treatment, combustion, or pollution-control technologies.*

Klein *et al.* (1975) and others have identified the "partitioning" behavior of trace elements upon combustion of whole coal, in which some elements become surface-bound to ash particles, encapsulated within the particles themselves, and/or are otherwise entrained in flue gases and emitted to the atmosphere (see again Table 2.1 on p. 38 above). However, as I have described elsewhere in this chapter and in Chapter 3, Endnote #2 on p. 98, several aspects of this process remain poorly-understood. In particular, it is unclear whether Class II or "intermediate" trace elements that are found in both fly ash and slag always exhibit the same binding behavior (*i.e.*, whether the fly ash : slag binding ratio remains constant for a given trace element regardless of whole coal properties and/or power plant combustion or pollution control technology).

Because I did not feel sufficiently confident in the meager data available on this subject to hazard a guess as to what these ratios (if any) might be, I have assumed for the purposes of the "ZCF" calculations in this chapter that all trace elements listed in the Swanson *et al.* (1976) dataset are fully transferred from whole coal to PCCs, with zero losses during the combustion process. However, in the trust that more reliable information on this topic will become available in the future, I have also constructed my spreadsheets in such a manner that the transfer ratios of trace elements between whole coal and PCCs can be manually adjusted on an element-by-element basis, and to a very fine degree of detail.

RESULTS AND DISCUSSION

Overview of Tables

Table C1 of Appendix C summarizes the process by which I assigned individual states to the general geographic regions of the United States documented in Swanson *et al.* (1976), such that all states within a given region would be assigned identical ppm values for trace elements. Utilizing arsenic as an example element, Table C2 then demonstrates variation in trace element content of whole coal produced per state annually for use in the electrical generation sector. Table C3 in turn summarizes the final, national-level results for all 28 trace elements of Swanson *et al.* (1976), providing tonnages present in domestically-produced whole coal supplied to the electricity generation sector over all years of the 2001-2008 period.

As mentioned previously, Table C3 represents the “worst-case” scenario for trace element content in coal ash – that is, a scenario in which there is full, lossless transfer of all trace elements present in whole coal during the combustion process, and where all of the trace elements thus concentrated in coal ash are considered to be fully-available to leaching processes. In contrast, the calculations in Tables C4 through C7 were performed assuming an experimental “zero concentration factor” (“ZCF”) scenario, in which trace element content of coal ash remains *proportional* to the natural baseline levels of whole coal, with no concentration or losses among any of the three ash types. Specifically, Table C4 summarizes the total arsenic content present in "adjusted" volumes of coal ash (*i.e.*, modified by the user-adjustable variables from Table B3 in Appendix B) produced by combustion of coal by electrical generation activities in the United States and placed into disposal annually over the 2001-2008 period; and as before, Table C5 repeats the same calculations for the other 27 trace elements present in "adjusted" coal ash. The next two tables further break these values into subcategories, both by ash type and by disposal format (*i.e.*, amount of material placed in either landfills or surface impoundments); Table C6 displays results for 2008 only, and Table C7 for the entire 2001-2008 period. Selected results from some of the above calculations are also summarized in Table 3.1 on the following page for ease of comparison.

Finally, Table C8 provides a comparison of ppm values for ten trace elements present in whole coal from Swanson *et al.* (1976) to those concentrations measured in coal combustion residues from EPA's June proposed disposal rule in *Federal Register* (2010), a matter which will be discussed in greater detail below.

Significance of Findings

As stated at the beginning of this chapter, my goal in performing these calculations was not to supplant EPA's much more intricate leaching and toxicity analyses, but simply to provide information that would help place these values in the proper perspective – namely, by providing basic tonnage calculations for the amounts of various trace elements *potentially* present in coal ash and thus available to leaching activities. This concept should be much more easily understandable by the general public than EPA's scientifically-thorough but less-intuitive system of hazard quotients (HQs)

Table 3.1: Summary comparisons of trace element content of PCCs produced by electrical generation activities and placed into disposal annually, 2001-2008, as calculated for both “worst-case” (100% transfer) and “zero concentration factor” (proportional transfer) scenarios.

Trace Elements		**Theoretical Trace Element Content of PCCs Produced Annually (in short tons)**		**Theoretical Trace Element Content of PCCs Placed into Disposal Annually (in short tons)**		**ZCF “Adjusted” Trace Element Content of PCCs Captured and Placed into Disposal Annually, by Disposal Method (in short tons)**			
						Landfill		**Surface Impoundment**	
		2008	**2001-2008**	**2008**	**2001-2008**	**2008**	**2001-2008**	**2008**	**2001-2008**
Class I	Ba	162,856	1,198,690	90,308	721,836	4,509	34,351	2,010	15,294
	Co	3,043	22,773	1,687	13,730	115	873	51	389
	Mn	83,008	615,846	46,030	371,489	3,304	24,423	1,473	10,877
	Sc	2,214	16,607	1,228	10,006	81	622	36	277
	Sr	80,547	599,447	44,665	360,998	2,604	20,004	1,161	8,907
	Th	2,389	17,775	1,325	10,711	80	608	36	271
Class II	As	5,121	38,136	2,840	23,005	203	1,511	90	673
	Cd	257	1,906	143	1,149	8	64	4	29
	Cu	10,953	82,001	6,074	49,420	401	3,056	179	1,361
	Ga	3,813	28,552	2,114	17,209	147	1,112	65	495
	Pb	7,030	52,093	3,899	32,425	250	1,861	111	829

Table 3.1 (cont.)

Class II (cont.)	Sb	528	3,935	293	2,372	19	144	8	64
	Se	1,950	14,741	1,081	8,882	79	609	35	271
	Zn	14,254	106,443	7,904	64,160	494	3,772	220	1,680
Class III	Hg	84	626	46	377	3	23	1	10
Intermediate between Classes I and II	Ni	7,774	58,422	4,311	35,227	312	2,364	139	1,053
	U	947	7,113	525	4,284	34	260	15	116
	V	13,730	103,084	7,614	62,119	513	3,929	229	1,749

and risk percentiles.

When viewed in comparison to annual U.S. whole coal production levels, the initial results presented in Tables C2 and C3 were not surprising: given that the 28 trace elements under examination were indeed at very low mean concentrations in the whole coal samples studied by Swanson *et al.* (1976), it is indeed only expected to see elements such as mercury (84 short tons) or arsenic (5,121 short tons) forming such comparatively small percentages of the more than 1 billion short tons of domestically-produced whole coal consumed by electrical generation activities in 2008. Similarly, it is unsurprising to see that under the proportional-transfer scenario of the subsequent "ZCF" calculations, trace elements continue to form only a small fraction of the produced material (4 short tons Hg and 293 short tons As placed into disposal in 2008, respectively; for other trace elements, see side-by-side comparison of both transfer scenarios in Table 3.1 above).

I suggest, however, that the true significance of these calculations is not necessarily the results which have been documented in Tables C2 through C7, but rather in the information they reveal to be *missing* in our understanding of coal ash characteristics – not only regarding how trace elements are leached from coal ash particles, but how and why they become bound to or encapsulated within these particles during the combustion process. To this end, I found Table C8 to be a particularly striking summary, simply because there has been so little study on "concentration factors" which influence accumulation of trace elements in coal ash during formation. In comparing concentrations of ten trace elements of concern present in both whole coal (Swanson *et al.*, 1976) and in coal combustion residues (*Federal Register*, 2010, p. 35169), we can see that many of the EPA mean values for these trace elements fall either within the uppermost range for geometric mean concentrations cited in the Swanson study, or beyond it entirely.

Although advancements in sensitivity of sampling instruments likely accounts for at least some of the more dramatic discrepancies between EPA values and the upper-limit ranges of the older Swanson dataset (*e.g.,* Cr, Sb), the divergence of values even in the mean ranges (*e.g.,* As, Pb) is worth further follow-up study to determine whether differences are due to variations in sampling methodologies among the two datasets (*e.g.,* limited sample sizes, uneven geographic distribution of sampled areas, etc.), or are in fact the result of a concentration effect as whole coal is reduced to coal ash. Needless to say, if this latter alternative is reinforced by further research, the ZCF values calculated in this chapter therefore will be underestimates of the total amount of these trace elements present in PCCs produced and placed into disposal annually; the true values will likely lie somewhere between the ZCF the full, lossless transfer scenarios described above.

While the results of studies such as EPA (2006) have been valuable in demonstrating the effects of pH on leaching, the initial factors which lead to concentration of trace elements in coal ash should merit just as much attention. Particularly if technological controls can be instituted to decrease the amount of trace elements present in coal ash and other PCCs that are placed in disposal annually (*e.g.,* via chemical treatment with chelating agents to remove heavy metals such as mercury, as described in EPA, 2006), this could potentially be a more cost-effective regulatory method to prevent leaching from newly-disposed PCCs than EPA's current proposal. Under this regulatory method, then, regulations on disposers could be relaxed in favor of stricter point-source contamination-reducing controls at power plants themselves, similar to the myriad emissions-control mechanisms already in place under existing environmental laws. Also notable is that such a setup would also avoid much of the feared “stigma” associated with classifying PCCs as hazardous waste (see Chapter 5 for further discussion regarding this concern).

At the moment this is clearly only speculation, as more research into this area is necessary before this type of technology-forcing proposal could be fully compared with the existing EPA rule. Nevertheless, given the already-considerable share of the U.S. electricity-generation sector fueled by coal, this is an important question which deserves further study, particularly if current growth patterns continue into the future.

Chapter 4: Should Coal Ash be Regulated as a Low-Level Radioactive Waste?

INTRODUCTION

As I have noted in previous chapters, there remain many significant gaps in our knowledge of PCCs. Not only is there no clear consensus on the amount of materials disposed annually, but the processes controlling the concentration of trace elements within this material are poorly understood, as is the leaching behavior of PCCs on both long- and short-term timescales. Taken together, these “blank spots” of information pose not only technical difficulties for creation of new PCC disposal regulations, but also political difficulties in securing approval for these proposals.

EPA’s June proposed disposal rule provides three different Subtitle alternatives modifying RCRA legislation. Yet is this necessarily the most cost-effective approach for regulating national PCC disposal, or are there other existing environmental regulations which could be modified in similar fashion? This chapter examines whether regulations established by the U.S. Nuclear Regulatory Commission (NRC) for disposal of nuclear waste with low levels of radioactivity are applicable to PCC disposal as well – and, in the process, what might be the associated costs of such an approach.

What follows here is partly a theoretical exercise, a “worst-case” regulatory response to accompany my exaggerated “worst-case” trace element leaching scenario; but I believe that the comparison between this regulatory approach and that outlined in EPA’s June proposed rule should prove instructive in evaluating the latter’s cited cost estimates – and so, too, the rule’s relative effectiveness in accomplishing its goal of reducing potential leaching pathways for trace elements contained in disposed PCCs.

WHAT IS LOW-LEVEL RADIOACTIVE WASTE?

LLRW Classification

As defined by NRC, nuclear waste may be generally classified into high-level radioactive waste (HLRW), low-level radioactive waste (LLRW), or waste with intermediate activity levels (most often referred to as transuranic waste, or TRU; this category is sometimes further subdivided into other classes). As summarized in Table D1 of Appendix D, LLRW may itself be subdivided into three classes: A, B, and C, in order of increasing radioactive content; TRU and HLRW therefore fall into the "greater than Class C" category.

Following the Chernobyl incident and more recently the U.S. debates on nuclear waste storage in Yucca Mountain, much public attention has understandably focused on health and safety concerns associated with the transport and storage of HLRW, highly-radioactive material primarily associated with spent nuclear reactor fuel. However, the potential public health impact posed by the much larger volume of LLRW that is produced and stored annually has largely been ignored.

On average, a typical 1 GW nuclear power plant produces 10-20 m^3 of HLRW annually, and 200-350 m^3 of LLRW (IEA, 2007, p. 3-4).[10] Low-level radioactive waste is defined by NRC (2002, p. 19) as items which have been contaminated with radioactive material or have become radioactive through exposure to neutron radiation. These may include tools, filters, and protective equipment from nuclear power plants, as well as test tubes, animal carcasses, and other materials exposed to radioactive isotopes in medical and research settings.

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[10] For reference, 1 m^3 = 35.315 ft^3. Thus, this 1 GW power plant would produce approximately 353-706 ft^3 of HLRW and 7,063-12,360 ft^3 of LLRW annually.

LLRW Storage

Because of the relatively low radioactivity, NRC has allowed burial of LLRW in near-surface shallow trenches. Shielded containers are not necessary, although a specialized engineered barrier may be used to prevent soil and groundwater contamination by radionuclides, particularly from class C waste (see 10 CFR 61 for more regulatory details). Particularly in the medical field, some waste generators choose to dispose of low-level radioactive waste in on-site containment facilities; however, this comprises only a small fraction of the total LLRW generated nationwide annually. More frequently, this material is shipped by generators to specialized radioactive waste disposal facilities for long-term storage. In all cases, LLRW disposal sites must comply with state and federal requirements for the siting, operation, and post-closure care of disposal facilities, and must both monitor the site and be prepared to take corrective action in the case that leaching results in a contaminant plume (NRC, 2002, pp. 27, 30).

Seven facilities have historically served as repositories for low-level wastes in the United States, although four of these sites – located in or near Sheffield, Illinois; Morehead, Kentucky; Beatty, Nevada; and West Valley, New York – are no longer in operation. Of the three remaining locations which currently continue to accept wastes, the sites in Barnwell, South Carolina (operated by Duratek/Chem Nuclear) and Richland, Washington (operated by U.S. Ecology) both accept Class A, B, and C wastes. The Clive, Utah facility operated by Energy Solutions (formerly Envirocare of Utah) is more specialized, primarily storing Class A bulk waste such as contaminated soil received in preparation for license termination from facilities which have permanently shut down (NRC, 2002, p. 26-27).

REASONS FOR POTENTIAL RECLASSIFICATION

The relationship between human cancer rates and low-dose exposure to radiation has been an extremely controversial topic in medical literature for years, due largely to the difficulty of observing a small number of additional cancers against very high background incidence rates and numerous confounding variables. Some comprehensive summaries of both debate and scientific findings on this subject have been presented by Wall *et al.* (2006) and Dendy (2005).

Although a causal link between low-dose exposure to radiation and cancer has not yet been established, many scientists advise a conservative approach perhaps best summarized by EPA (1989, p. 264; emphasis mine):

> For neither induction of cancer nor genetic effects […] is there any convincing evidence for a “threshold” (*i.e.*, some dose level below which the risk is zero). Hence, so far as is known, any dose of ionizing radiation [resulting from decay of radionuclides], **no matter how small,** might give rise to a cancer or to a genetic effect in future generations.

Clearly, similar concerns for public health and safety led the U.S. Nuclear Regulatory Commission to create and enforce the elaborate current disposal system for nuclear waste. The question therefore becomes: are levels of radioactivity resulting from the decay of trace radionuclides such as uranium and thorium sufficiently high in coal ash or other PCCs which are produced and placed into disposal annually to merit regulation of these materials under NRC definitions of low-level radioactive waste? And if so, are the costs of disposal under existing NRC methods necessarily higher than those presented by the EPA in the June proposed rule?

COMPARING COAL ASH AND NUCLEAR-SOURCED LLRW

Methodology

The Manifest Information Management System, or MIMS database (see DOE (2010a) and “A Cautionary Note on DOE Dataset Reliability” section below) provided activity levels of LLRW produced by electrical generation activities at nuclear power plants. In order to compare these activity levels to those of coal ash produced by electrical generation activities annually, I multiplied the “adjusted” coal ash type production and disposal values calculated in Chapter 1 above by the typical activity levels found in various coal ash types, as provided by EPA (2009c).

Because the MIMS database also provided statistics for volumes of LLRW placed into disposal annually, for comparison purposes I needed to convert my “adjusted” coal ash type annual production and disposal values from Chapter 1 from units of short tons to cubic feet. This was accomplished by multiplying these values by the minimum and maximum density values for each ash type, which in turn were derived from the specific gravity values provided by Majizadeh *et al.* (1978) and the U.S. Department of Transportation (DOT, 2002); the calculations used to perform this conversion are described in the notes to Table D6 of Appendix D.

With both of these initial calculations complete, I was then able by simple division to determine activity levels per cubic foot of nuclear-sourced LLRW and of coal ash produced annually, as well as activity levels in this material per kWh of electricity produced annually by both coal-fired and nuclear power plants (as derived from annual electrical generation statistics from EIA, 2010k).

Finally, projected costs for placement of all annually-disposed coal ash into commercial LLRW storage were calculated utilizing the $400/ft^3 estimate provided by GAO (2004).

Key Assumptions

1. *For each coal ash type, all ash has the same minimum, maximum, and average activity levels as provided by EPA (2009c), regardless of power plant technology or original geographic origin of the whole coal utilized in combustion.*

   Similar to my assumptions related to the Swanson *et al.* dataset in Chapter 3 above, I have here assumed that the activity level values reported by EPA (2009c) are "flat rates" applicable to each of the three coal ash types produced nationwide annually by electrical generation activities, respectively. These rates are not changed by technologies present at the power plants or the geographic origins (and thus the associated natural variations in radionuclide content) of the fuel-coal burned at those power plants. Additionally, activity level values for the ash types remain confined with the ranges specified by EPA, neither falling below the minimum nor above the maximum range boundaries as specified.

2. *Boiler slag shares the same activity levels as bottom ash.*

   As I have noted in Table D4 of Appendix D, EPA (2009c) provided minimum, maximum, and average activity levels only for fly ash and bottom ash, but not for boiler slag. In order to complete my calculations here, I here made the temporary assumption that boiler slag possessed activity levels identical to those of bottom ash, which may or may not be the case in reality. Should this data become available at some later point, I have constructed this particular table in my spreadsheets to allow easy manual adjustment of these values by the user.

3. *Annual electricity production from coal combustion as provided by EIA (2010k) is reconcilable with EIA(2010f) data for annual domestic fuel-coal production and consumption.*

As described above, the calculations in this chapter comparing activity levels per kWh of electricity generation for both coal ash and LLRW from nuclear power plants over the 2001-2008 period were performed utilizing annual electricity generation statistics from EIA (2010k, p. 14). However, while EIA statistics for electricity production from coal include combustion of "traditional" coal types (*i.e.*, anthracite, bituminous, subbituminous, and lignite coal) as well as other sources (*i.e.*, waste coal and coal synfuel), the limited timeframe of this project prevented me from obtaining more accurate information as to the electricity generation split among these two categories. As seen in the discussion at Chapter 1, Endnote #4 on p. 94, this "lump treatment" of fuel-coal types remains a persistent source of error in my calculations across multiple chapters of this project, although I hope that any discrepancies introduced by use of these non-specific values are relatively small.

4. *Estimated costs for commercial storage of LLRW are assumed to be flat rates, not dependent on quality or chemical composition of the ash, nor including any costs associated with ash recovery, transport, and handling.*

For simplicity, I have here assumed that the GAO (2004) quote of $400 per cubic foot as an average rate for commercial LLRW storage is applicable to all coal ash and other PCCs which would be placed into storage within a given year. In a real-world setting this is clearly unrealistic, as these costs would vary among shipments of these materials – a shipment coal ash containing particularly high radioisotope levels might require storage as Class B rather than Class A waste under NRC regulations, for example, and transportation costs would certainly increase in direct proportion to the distance of the coal-fired power plant to the disposal site. Equally, if the objective was to remove older PCC deposits from surface impoundments for permanent closure of these sites, there would be some cost associated with dewatering the material in preparation for transport.

There is also some general error in assuming off-site transport and storage of coal ash and other PCCs at all: although the U.S. Government Accountability Office cites several high-cost examples of on-site storage (GAO, 2004), these are largely associated with LLRW produced by research or medical activities rather than by nuclear power plants; it is possible that the much larger quantity of material associated with utility operations (not to mention the already-limited amount of space available at the three existing commercial LLRW disposal facilities, as discussed in the GAO report) would in fact make some form of on-site or local storage a less-expensive option for electric utilities required to store PCCs in compliance with NRC regulations.

Due to the limited timeframe of this project, I could not obtain more specific values or information regarding how these factors might affect ultimate storage costs, but I have for now constructed my spreadsheets in such a way that users can manually adjust the theoretical storage cost per cubic foot for coal ash which has already been placed into disposal into either surface impoundments or landfills annually over the 2001-2008 period.

A Cautionary Note on DOE Dataset Reliability

Developed by DOE in 1986, the Manifest Information Management System, or MIMS, is a publicly-accessible federal database created to monitor the management of commercial low-level radioactive waste within the United States. According to DOE (2010a), all information reported is derived from manifests for waste shipments to the closed Beatty site as well as the active Barnwell, Richland, and Clive sites.

The accuracy of the waste volumes presented by the MIMS database has been called into question by a GAO report which noted discrepancies between volumes of stored waste reported by disposal facility operators, as well as failure to report the large quantities of LLRW shipped by the DOE to the Clive disposal site (GAO, 2004). Although DOE has since responded to this latter criticism (*cf.* DOE, 2010b), to date it is not clear if DOE has yet taken any steps to improve its data-collection process prior to reporting of values in MIMS.

In short, it remains unclear whether values provided by MIMS for the volumes (and, by association, the radioactivity levels) of LLRW stored at these facilities are at all representative of actual values. Table D2 of Appendix D attempts to summarize 2008 statistics as provided by the MIMS database (DOE, 2010a) and the DOE list of information excluded from MIMS analysis (DOE, 2010b), but even from this rudimentary compilation, several missing pieces of information are immediately apparent.

DOE (2010b) provides no data regarding the radioactivity levels of DOE and non-DOE waste excluded from the MIMS database, nor does it explain whether any other LLRW disposal facilities (*i.e.*, Barnwell or Richland) were also recipients of waste from this excluded stream. No explanation is given for how MLLW (*i.e.*, “mixed low-level waste”) differs from “LLW” (*i.e.*, “low-level [radioactive] waste”) as reported, and the rationale for including byproduct and NORM (*i.e.*, naturally-occurring radioactive material) wastes in these statistics is similarly unclear.

Finally, the waste volumes provided by DOE (2010b) are puzzling in light of the fact that the MIMS database already reports waste received by the Clive facility in 2007 (DOE, 2010a). While this might have been understandable if DOE only excluded its own waste from the MIMS database, the fact that non-DOE waste also was excluded from MIMS reporting implies that these waste types are somehow distinct from typical NRC classifications of low-level radioactive waste. Unfortunately, without at least a rough estimate of how these new waste types fit into the existing NRC classification scheme (as described above), no further analysis can be performed on the DOE (2010b) values.

Results and Discussion

Activity Level Comparisons

The majority of my radioactivity calculations are summarized in Table D5 of Appendix D, which compares the activity levels of Class A LLRW placed in commercial storage annually (subdivided by both national totals documented in the MIMS database, and by Class A LLRW produced by nuclear power plants only) to those of coal ash produced and placed into disposal annually (including both national totals and values for fly ash only). Along with results from Table D7 of Appendix D, this information is duplicated in Table 4.1 on the next page.

As I have documented in the notes to that table and in the “A Cautionary Note on DOE Dataset Reliability” section above the fact that the MIMS dataset has demonstrably underreported national NRC-regulated LLRW volumes placed into disposal facilities

Table 4.1: Summary comparison of activity levels (Ci/ft^3 and Ci/kWh) for Class A LLRW materials produced and placed into commercial storage by nuclear power plants, and for PCCs produced and placed into disposal by coal-fired power plants in 2008.

	Activity Level per Kilowatt-Hour of Electricity Produced (Ci/kWh)		**Activity Level per Cubic Foot of Waste Material (Ci/ft^3)**
All Coal Ash Types Captured and Placed into Disposal in 2008	*Low:*	4.17 x 10^{-11}	1.10 x 10^{-7}
	Medium:	1.16-1.21 x 10^{-10}	3.05-4.37 x 10^{-7}
	High:	2.02 x 10^{-10}	5.32 x 10^{-7}
All Fly Ash Captured and Placed into Disposal in 2008	*Low:*	3.47 x 10^{-11}	1.64 x 10^{-7}
	Medium:	1.01 x 10^{-10}	4.77-6.68 x 10^{-7}
	High:	1.68 x 10^{-10}	7.98 x 10^{-7}
All MIMS-Reported Class A LLRW Placed into Commercial Disposal in 2008 (All Sources)		8.66 x 10^{-9}	0.124
All MIMS-Reported Class A LLRW Produced by Nuclear Power Plants Only and Placed into Commercial Disposal in 2008		2.80 x 10^{-9}	3.64 x 10^{-3}

annually over the last several years (and thus the activity levels of this material), thus regretfully adding an unknown – but potentially large – amount of error to my calculations and comparison here. For the sake of debate here, however, let us for the moment assume that the low-level waste streams which have been excluded from the MIMS database (see Table D2 of Appendix D) do *not* originate from utility sources; *i.e.*, that all LLRW produced in the U.S. annually by electrical generation activities at nuclear power plants are reported fully in MIMS.

If this scenario is accurate, then the results shown in Table 4.1 above demonstrate that the case for reclassifying coal ash as LLRW is largely a matter of perspective. When comparing total average activity levels for both coal ash placed into disposal and nuclear utility-sourced Class A LLRW placed into commercial storage in 2008, coal ash has levels that are approximately one order of magnitude lower (230-240 Ci versus 2,255 Ci). Similarly, the activity levels per cubic foot of material placed into disposal annually are significantly higher for nuclear utility-sourced Class A LLRW in 2008 (0.00364 Ci/ft^3) than the average ranges for all coal ash types (3.05-4.37 x 10^{-7} Ci/ft^3). However, when comparing average activity levels per kilowatt-hour of electricity produced per power source in the same year, the two waste streams become somewhat more similar: compare 1.16-1.21 x 10^{-10} Ci/kWh for coal ash (1.01 x 10^{-10} Ci/kWh for fly ash only) to 2.80 x 10^{-9} Ci/kWh for nuclear utility-sourced Class A LLRW.

Cost Estimates, and a Dose of Realism

As shown in Table D8 of Appendix D and summarized in Table 4.2 on the next page, the potential costs for regulating coal ash disposal under the same NRC regulations used for low-level radioactive waste are simply untenable. At the \$400/ft^3 rate for commercial LLRW storage cited by GAO (2004), the storage costs for coal ash types

Table 4.2: Summary comparisons of estimated commercial storage costs for both fly ash and all coal ash types that were produced and placed into disposal from 2001-2008, if these materials instead had been reclassified as LLRW under NRC regulations and stored for $400/ft^3 (as per GAO, 2004).

	Estimated Commercial Storage Costs for Materials That Were Produced and Placed into Disposal in 2008		**Estimated Commercial Storage Costs for Materials That Were Produced and Placed into Disposal, 2001-2008**	
All Disposed Coal Ash Types				
All Disposal Formats	*Minimum:*	$219,558,481,546	*Minimum:*	$1,678,313,769,192
	Maximum:	$302,111,341,689	*Maximum:*	$2,309,035,364,940
Surface Impoundments Only	*Minimum:*	$67,636,596,165	*Minimum:*	$516,635,652,117
	Maximum:	$93,120,294,618	*Maximum:*	$711,233,766,862
All Fly Ash				
All Disposal Formats	*Minimum:*	$173,738,463,728	*Minimum:*	$1,325,139,781,751
	Maximum:	$243,233,849,219	*Maximum:*	$1,855,185,694,452
Surface Impoundments Only	*Minimum:*	$54,095,235,271	*Minimum:*	$412,595,729,943
	Maximum:	$75,733,329,380	*Maximum:*	$577,634,021,920

produced in 2008 alone would be between $219.6 and $302.1 billion for all storage formats (*i.e.*, ash which would otherwise have been placed into both surface impoundments and landfills). For coal ash destined for disposal in surface impoundments only, the 2008 disposal costs are only somewhat lower, totaling between $67.6 and $93.1 billion. Comparing these costs to EPA estimates for the various regulatory options under the June proposed rule (see Table A1 of Appendix A), we see then that even the most expensive Subtitle C approach is several times lower than these NRC estimates, costing about $1.4 billion in average annualized equivalent value per year; and the two Subtitle D options presenting still lower annual regulatory costs ($587 and $236 million, respectively).

As I have stated earlier in this paper, my intention in these calculations was not to suggest that the reclassification of PCCs as LLRW is a viable regulatory alternative to the proposed EPA disposal rule – indeed, between the cost estimates presented above and my observations in Chapter 4, Endnote #1 on p. 100, I doubt that such an approach would ever win legislative approval. Rather, as a continuation of the "worst-case" trace element concentration concepts presented in Chapter 3 and the discussion of their potential public and environmental effects from Chapter 2, the cost estimates presented here are important not necessarily for their real-world applicability, but instead for raising a very valid question about the cost-benefit tradeoffs inherent in all environmental regulation.

EPA historically is no stranger to public criticism regarding its maximum contaminant levels (MCLs), the upper allowable limit for elemental or chemical concentrations in air, water, food, etc.; above this limit, contaminants are considered to be at "unsafe" levels, with potential adverse health effects increasing in occurrence and severity with increased chemical levels and human exposure. However, some recent studies have suggested that these traditional approaches to toxicology are not necessarily applicable in all cases: rather than health effects always scaling linearly with dosage and exposure, the dose-response curves may rather assume non-linear shapes for some chemicals (*e.g.,* a U-shaped curve, in which adverse health effects are seen most acutely at both very high and very low concentration levels).[11]

Although this remains a relatively new and controversial area of research, the implications are potentially quite interesting. Namely, if acute health effects are indeed visible at very low concentrations for certain elements or chemicals, we must revise our understanding and acceptance of MCLs as "safe" doses. From this point of view, then, the seemingly-absurd scenario presented here to control leaching of trace elements may in fact gain some relevance – and therein lies the challenge to regulators.

Inarguably, environmental regulation plays a critical role in protecting human and environmental health; yet as research continues into hormetic and particularly epigenetic effects of chemical exposure among both humans and animals, it is conceivable that even naturally-occurring levels of trace elements might be found to have some adverse health effects. (See again the EPA (1989) statement that no dose of ionizing radiation may be a "safe" dose.) The challenge for policymakers will lie in determining where to set MCL (or similar "boundary" levels) so as to afford maximum health protection for both adults and children while preventing runaway costs of regulation. If the regulatory costs for the above "worst-case scenario" are any benchmark to go by, the current proposed EPA disposal rule for coal ash may in fact represent this "happy medium" quite well.

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[11] See Chapter 4, Endnote #2, p. 101.

Conclusions

SUMMARY

In this paper, I have attempted to address some of the numerical and analytic inconsistencies present in EPA's June 2010 proposed PCC disposal rule, and have presented some of my own calculations in hopes of reconciling at least some of the most pressing information "gaps" in our current understanding of PCC disposal rates and trace element content. My primary points may be summarized as follows:

1. *There is an unresolved 20% discrepancy between EPA estimates and industry statistics from EIA and ACAA regarding annual PCC disposal for 2008. My calculations tentatively indicate EPA's stated value may be an overestimate, but more research is needed. (Chapter 1)*

   My attempt to resolve this discrepancy by estimating total PCC production and disposal from annual U.S. whole coal production and consumption statistics resulted in a set of values more closely resembling ACAA/EIA data than EPA estimates. Specifically, I have found PCC production in 2008 to be between 122-130 million short tons, comparable to EIA and ACAA respective estimates of 131 and 136 million tons, but lower than the 141 million ton value provided by EPA. For the same year, I calculated national PCC disposal to be between 68-69 million short tons; again, this was comparable to the ACAA reported value of 75 million short tons, but significantly lower than the 94 million short ton estimate provided by EPA in the June proposed disposal rule.

   Although the above calculations are clearly an oversimplification of a very complex situation, and may have been affected by any of several potential sources of error, these results nevertheless provide a much-needed third point of view in what is otherwise a binary debate. It is my hope that they may serve as a helpful launching point for further investigation of this matter by EPA and other groups.

2. *EPA's estimates regarding the total amount of coal ash and other PCCs which have been placed into disposal since the 1920s appear to be alarmingly low. (Chapter 1)*

If the statistics regarding historical fly ash disposal presented in EPA (2010a, p. 3) have not been misstated but are in fact an accurate summary of the Agency's available data on this subject, something is severely wrong. EPA (2010a) estimates total landfill disposal of fly ash since the 1920s to be between 100 and 500 million tons; yet my calculations for the 2001-2008 period alone estimated this amount to be almost 200 million short tons (and as discussed in the point above, this too may be an underestimate).

Currently, the June proposed disposal rule would not impose regulations on closed or non-operational disposal sites; yet due to their age, it is likely that many of these sites possess only minimal clay liners or no liners at all, and thus could be key sources of leachate. By excluding this secondary wastestream from its calculations, EPA may therefore underestimate total public and environmental health impacts created by leaching activity on stored PCC material. Gathering more information about these inactive "legacy" disposal sites therefore is critical to determining the leaching hazard they pose – and, in the process, may also prompt a change in direction of disposal legislation itself.

3. *More research is needed to understand trace element leaching behavior of PCC under low-pH conditions, and particularly whether erosion of vitreous particles can result in release of encapsulated trace elements within the 10,000-year time scale of EPA leaching hazard analysis. (Chapter 2)*

Cherry *et al.* (1984) observed that disposal of PCCs in surface impoundments could result in extreme "spikes" of very low pH (*i.e.,* creation of very acidic aquatic conditions), often accompanied by increased tissue uptake and bioaccumulation of heavy metals in local organisms. The impact of pH on trace element leaching has been only recently "rediscovered" by EPA, and much research still remains to be done in this area (*cf. Federal Register,* 2010, p. 35141-35142; Chapter 3, Endnote #1, p. 97).

With this said, however, special care should be taken to assess not only the chemical reactions in low-pH conditions, but also the *physical* reactions of the PCC particles themselves to the acidic environment (*i.e.,* how quickly the vitreous surfaces of particles erode). Because radionuclides such as uranium and thorium are most frequently "trapped" within rather than bound to the surfaces of vitreous PCC particles, they have traditionally been regarded as inert, and of little threat to human health and the environment. However, very little is known of how this vitrified material reacts to erosion, and particularly to the high-pH "spikes" observed by Cherry *et al.* (1984) upon exposure of PCCs to water. As noted by USGS (1997), the solubility of radioactive elements can be "enhanced" by acidic conditions; thus, if released by erosion, it is possible that radionuclides could undergo the same "accelerated" tissue uptake as some heavy metals in Cherry *et al.* (1984). Because such findings likely would significantly impact EPA's long-term assessment of PCC leaching risks (particularly frequency of cancers) as presented in EPA (2010b), thorough assessment of this topic should be made a particular priority of future research in this area of study.

4. *EPA's demographic analysis of public health risks posed by leaching from PCC disposal sites may not necessarily be an accurate portrayal of actual exposures for communities living near these facilities. (Chapter 2)*

First, it appears that EPA's demographic analysis assumes that PCCs are disposed at the same site at which they were produced; yet this is not always the case in reality, as producers may choose to ship the materials to off-site facilities, and occasionally to other states entirely (as in the 2008 Kingston ash spill cleanup). In these situations, the modeled leaching activity would be attributed to the wrong ZIP code, and the wrong demographic groups incorporated into the analysis as a result.

Additionally, EPA's analysis appears to include several power plants which do not dispose PCCs at all, but which instead sell the entirety of annual production of these materials to industry for beneficial uses. This likely would result in an overestimate of health risks posed to communities situated near the power plants, as even if disposal sites were utilized to handle small quantities of PCCs on a temporary storage basis throughout the year, their collective leaching profile would almost certainly differ from those of permanent disposal sites.

Finally, I am concerned that EPA's demographic analysis is insufficiently granular to accurately capture the characteristics of those communities most likely to be affected by leaching activities. Because ZIP codes are not a standardized unit of area, the demographic analysis may include population statistics for communities located several miles from the power plant; particularly if located upgradient on the water table, these communities would not receive the same leaching exposures as individuals living in the more immediate vicinity of the power plants and associated disposal sites. Similarly, EPA's analysis focuses only on the ZIP code of the power plant's location, but does not account for cross-ZIP code contaminant plumes (*i.e.*, where a plant is situated near the boundary of two ZIP code areas, and the community likely to be most affected by the contamination is in the adjacent area). All of these factors could potentially distort the conclusions of EPA's risk analysis, and deserve closer consideration.

5. More research is needed to determine how trace element concentrations in PCCs are modified by the combustion and pollution-control technologies utilized by coal-fired power plants. (Chapter 3)

As summarized in Table C8 of Appendix C and discussed above in Chapter 3, EPA has found higher mean concentrations of certain trace elements in PCCs than Swanson *et al.* (1976) records in whole coal. In other words, there appears to be one or more variables in the combustion process which causes trace elements to be "transferred" from whole coal to PCCs at ratios higher than would otherwise be indicated by the simple loss of volume by whole coal combustion.

Several possible mechanisms may be responsible for this disproportional transfer of trace elements to PCCs during combustion, including pollution-control technologies and the combustion chamber design itself (see Chapter 3, Endnote #2, p. 98), but currently this research area is almost completely unexplored. If certain technologies were found to yield particularly low transfer rates of trace elements to PCCs, however, EPA may find top-down regulation encouraging installation of these "cleaner" technologies to be a more financially-effective method of reducing leaching dangers from PCCs than the various Subtitle alternatives presented in the June proposed rule. As such, I highly recommend EPA prioritize at least preliminary research on this topic during this current public feedback period for the June proposed rule, and use any information gained to revise future drafts of the legislation accordingly.

6. *Although coal ash might qualify for regulation by NRC as a low-level radioactive waste, the costs of implementation are infeasibly high. In light of this, the cost estimates presented for EPA's June proposed disposal rule may in fact be quite reasonable, and are certainly much closer to an ideal "happy medium" of cost/benefit tradeoff. (Chapter 4)*

I began my calculations in Chapter 4 with two simple questions: Did PCCs contain sufficiently high levels of radionuclides to qualify as low-level radioactive waste (LLRW) under NRC regulations? And if so, how would the costs of this alternative regulatory method compare to those presented by EPA in the June proposed disposal rule?

Table 4.1 on p. 74 answers the first question with a tentative "yes." Radioactivity levels per cubic foot are much lower for PCCs than for the comparable group of LLRW produced by nuclear power plants annually (3.05-4.37 x 10^{-7} Ci/ft^3 versus 0.00364 Ci/ft^3). However, the two groups of wastes differ only by an order of magnitude for both total radioactivity levels (230-240 Ci for PCCs versus 2,255 Ci for nuclear utility-sourced Class A LLRW) and activity levels per kilowatt-hour of electricity produced (1.16-1.21 x 10^{-10} Ci/kWh versus 2.80 x 10^{-9} Ci/kWh).

Nevertheless, Table 4.2 on p. 76 demonstrates that even if PCCs were reclassified as LLRW, the costs of regulating this under current NRC standards would be too high to be feasible. In 2008, if all coal ash that otherwise would have been disposed had instead been placed in LLRW commercial storage facilities, the cost would be \$220-302 billion (\$67.6-93.1 billion for coal ash in surface impoundments alone). In comparison, even the most expensive Subtitle C option in the EPA June proposed disposal rule is projected to cost \$1.47 billion per year in averaged annualized equivalent value; Subtitle D and D prime options are expected to cost \$587 and \$236 million per year in average annualized equivalent value, respectively. Thus, the costs of the June proposed rule, while still significant, may still be more reasonable than some of the potential regulatory alternatives.

WHAT IS NEXT FOR PCC DISPOSAL REGULATION?

Closing the Information Gaps

Due to the "lag time" inherent in scientific research, it is an unfortunate reality that regulatory decisions often must be made without possession of all the desired facts. In this respect the June proposed disposal rule is no exception. EPA has used the best information available in modeling leaching risks; yet there are nevertheless significant "information gaps" in several areas of knowledge regarding PCC production and disposal. Although these are all complex topics which likely will not be solved in time to impact the June proposed rule, each may have potential to spur further future regulatory reform for PCC disposal.

One such gap is the lack of research regarding how trace element leaching rates for PCCs are affected by both length of time the material has spent in disposal, and particularly in the case of surface impoundments, by the frequency with which new quantities of PCC waste were added to the disposal site. In particular, if these “legacy” disposal facilities (particularly younger units) were found to generate leachate at rates comparable to active facilities, EPA likely would need to consider extending disposal regulations to cover some proportion of these closed units as well, possibly under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA, the “Superfund Act”).

A second knowledge gap involves our relatively poor understanding of the mechanisms resulting in PCC formation at power plants – namely, how the use of certain combustion or pollution control technologies by coal-fired power plants might affect the concentrations of trace elements bound to particles surfaces and thus easily-available to leaching activities. If such technologies were found and determined to be cost-effective, this could conceivably lead to a regulatory scenario in which EPA would find it more reasonable to encourage installation of the desired control technologies at power plants, perhaps under jurisdiction of the Clean Air Act.

A third and final knowledge gap is the emerging research area of hormetic response to contaminants, in which environmental impacts and public health effects potentially may be observed at low-level exposures to leached trace elements, in addition to the higher dosage levels considered in EPA risk assessment models. Although it is somewhat doubtful whether research in this area would yield any definitive results within a reasonable timeframe, the fact that EPA thus far has not included health risks posed by radionuclide levels in leachate may be an area the Agency wishes to address in future drafts of the disposal regulation, if nothing else to provide clarity regarding the earlier statements of EPA (1989, p. 264).

Considering Stigma Effects and Regulatory Tradeoffs

Nevertheless, these remain but small aspects of the whole picture of PCC regulations, and I bow to the industry's years of expertise in the much larger central debate regarding the merits and drawbacks of each Subtitle proposal presented in the June proposed disposal rule. However, two points of discussion are worth mentioning briefly.

First, it has been noted that production of coal ash and other PCCs is in itself a tradeoff with existing federal air pollution regulations – that is, by improving emissions-filtration systems at power plants under the Clean Air Act and other similar laws at both state and federal levels, the fine particulates such as fly ash which might otherwise be emitted to the atmosphere in flue gases would be captured as solids instead, thereby increasing the total amount of PCC materials to be managed by the power facilities. To provide just one example of this relationship, the Maryland Healthy Air Act requires power plants to reduce sulfur dioxide (SO_2) emissions by 85% by 2013 with the use of flue gas desulfurization equipment. This regulatory measure is estimated to reduce the state's total SO_2 emissions by over 200,000 tons, but will also result in a significant increase in coal combustion wastes (CCWs) produced at the 9 coal-fired facilities currently in operation: annual capture of these materials would increase from the current 2 million tons of fly and bottom ash to 4.5 million tons by 2013 (U.S. House, 2008, p. 15). Although it is unclear to what extent – if any – EPA has accounted for future air quality regulations in its cost estimates for the three disposal options presented in the June proposed rule, the Maryland example demonstrates just how rapidly adoption of similar standards among other states could affect PCC production and disposal at the national scale.

As a second point of discussion, organizations such as the American Coal Ash Association have noted in the public comments for the June proposed rule, air pollution tradeoffs are not necessarily the only route by which annual PCC production might increase. Namely, ACAA is concerned that power plants producing and disposing these materials would attempt to compensate for at least some of the new regulatory expenses by increasing costs for PCCs currently in demand for various industry applications (*e.g.,* fly ash for use in manufacturing of Portland cement). This economic "stigma" would thus not only discourage repurposing of PCCs by industry and increase the wastestream volume placed into disposal facilities annually, but would also reduce many of the environmental benefits which reuse of PCC material allows over the mining and use of new raw materials in industrial applications:

> [B]eneficially using CCRs as a substitute for industrial raw materials contributes (a) $4.89 billion per year in energy savings, (b) $0.081 billion per year in water savings, (c) $0.239 billion per year in GHG (*i.e.*, carbon dioxide and methane) emissions reduction, and (d) $17.8 billion per year in other air pollution reduction. In addition, these applications also result in annual material and disposal cost savings of approximately $2.93 billion. All together, the beneficial use of CCRs provides $25.9 billion in annual economic and environmental benefits (relative to 2005 tonnage).
>
> (*Federal Register*, 2010, p. 35155)

Although EPA has attempted to estimate the potential economic impact of such a "stigma" on costs of the Subtitle C and D regulatory options and believes that even in such a scenario off-site commercial PCC disposal should not increase significantly (*Federal Register*, 2010, pp. 35159, 35219; see also Table A1 of Appendix A), the fact remains that these are estimates only. Similarly, despite ACAA's insistence that the June proposed rule would “severely cripple” the PCC beneficial use industry, it admits that it does not possess any data to support these claims, and would require external assistance by consulting or academic groups to compile this information (ACAA, 2009b). As such, while further research certainly should be performed in this area if at all possible, in the absence of any further, firm information on this topic, these concerns about stigmatic effects of regulation are unfounded, and likely will not prevent EPA from continuing with its plans for PCC disposal legislation.

CLOSING STATEMENTS

EPA’s proposed rule for coal ash disposal is a very timely one: not only does coal currently compose nearly 50% of the U.S. energy supply, but EIA (2009b, p. 7) projects that by 2035, consumption will have increased by approximately 15% over current values to an annual total of 1,319 million short tons. As an ever-larger wastestream of PCC materials is placed into disposal each year, the potential for leaching events to adversely impact public or environmental health must also be considered to scale proportionally.

Particularly in light of the growing number of proven leaching damage cases now recognized by EPA (*Federal Register*, 2010, p. 35234-35239), and the Agency’s own modeling studies (EPA, 2010b), it seems clear that the danger of leaching is in fact real, particularly among unlined disposal units, and that some form of external corrective action is needed. Industry measures alone are simply not sufficient to combat this problem: at the current replacement rate, it would take decades to install protective liners and groundwater monitoring systems on the more than 600 older landfills and surface impoundments currently in operation nationwide – not to mention the still-unknown number of closed “legacy” PCC disposal sites, many of which are likely to be similarly unlined (*Federal Register*, 2010, p. 35151).

As demonstrated by EPA’s numerous solicitations for public comment and information both within and after publication of the June proposed rule, the document in its current form is only an early step in what will undoubtedly be a much longer road to a final PCC disposal regulation. Thus, while I have been critical of some of the specifics of EPA’s proposal in this paper – namely, several information gaps and flaws in analyses used to support its conclusions regarding leaching rates and associated public and environmental health impacts – I am hopeful that these issues can be clarified with further research, and potentially could be completed in time to improve future drafts of the current regulation.

Endnotes

INTRODUCTION

1. Readers familiar with the coal-fired utility sector or the beneficial coal ash use industry may find this acronym unfamiliar. This is in fact quite deliberate.

There is currently no consensus on what to call these products of coal combustion, a matter which has led to a veritable proliferation of acronyms, often with little to no actual indication as to what definitions might be behind them, or how they might differ. In a single Congressional hearing, for example, we see the terms “CCRs,” “CCWs,” “CCPs,” and “CC byproduct” being used apparently interchangeably by various governmental agencies and interest groups (U.S. House, 2008, pp. 10, 15, 18, 38).

Charles H. Norris, consultant at Geohydro, Inc., explains the driving factors behind this maze of terminology as follows:

> Much time and fury is devoted to the nomenclature associated with the materials that remain after the combustion of coal with or without other fuels – far more time than is necessary or constructive. It’s coal combustion (CC) waste. It’s CC product. It’s CC byproduct. It’s CC residual. [The beneficial-use sector believes that] it’s pejorative to use “waste” and that makes it harder to convince people to reuse it. Euphemistic phraseology lowers the perception of the need for protection, [and] if one defines the vocabulary, one controls the debate.
>
> (U.S. House, 2008, p. 38-39)

With this in mind, I have here created the novel term “PCC” to function as a catch-all term for all fly ash, bottom ash, boiler slag, flue gas residuals, cenospheres, and any other products of coal combustion. It is my hope that “starting over” with a neutral term will avoid any pre-existing assumptions associated with use of any of the above terms and in doing so, perhaps help to bridge some of these political and semantic divides.

2. To clarify: As used in this document, “coal ash” is a general term which collectively refers to the sum total of all fly ash, bottom ash, and boiler slag produced annually by whole coal combustion in the United States. When not referred to directly by name, these subcategories of coal ash may also be termed “coal ash types”; thus, “coal ash” as used here may be considered to be simply a briefer form of the lengthier term, “all coal ash types.”

CHAPTER 1

1. In its “2008 Coal Combustion Product Production and Use Survey,” for example, ACAA noted that the reported values were based on survey data from, respectively, “107 of a total industry-wide 266 coal-fired electric utilities (i.e., 40%) [and] 274 of a total industry-wide 509 coal-fired electric utility generating stations (i.e., 54%).”

2. As noted in Introduction, Endnote #1, p. 91 above, industry and federal agencies do not agree upon a single term that would collectively refer to the byproducts formed during the combustion of whole coal (*i.e.,* the group of materials I have here called PCCs, which includes but is not limited to coal ash). Although lack of definitions can sometimes make it difficult to compare values across datasets, for the reasons outlined below this does not appear to be a problem for the ACAA, EIA, and EPA values covered in this chapter.

As stated in the June proposed disposal rule, EPA defines both “coal combustion residuals” (CCRs) and “coal combustion products” (CCPs) as containing “fly ash, bottom ash, boiler slag, and flue gas desulfurization materials” (*Federal Register*, 2010, p. 35130). Similarly, ACAA defines “coal combustion products” (CCPs) as the combination of fly ash, bottom ash, boiler slag, materials produced by flue gas desulfurization (FGD) systems (specifically, FGD gypsum, and FGD material from wet scrubbers, dry scrubbers, and other sources, respectively), and ash produced by power facilities utilizing fluidized bed combustion (FBC) technology (ACAA, 2009a). Although EPA is unclear in *Federal Register* (2010, p. 35212) regarding the definition of “coal combustion residuals” utilized by EIA, the fact that ACAA relies heavily on EIA datasets for compilation of its own annual reports would imply that EIA definitions are likely very similar – if not identical – to the above ACAA definitions of CCPs.

In short, it appears that the “baseline” values for PCC production and disposal in 2008, as provided by ACAA (2010a) reports and EIA and EPA in *Federal Register* (2010) are in fact referring to the same set of end-materials. Thus, despite the differences in terminology (“CCR” and “CCP” vs. “PCC”), it appears that there is sufficient overlap between the stated definitions to allow comparison on equal footing.

3. Specifically, ACAA (2009a) states that approximately 136 million tons of “coal combustion products” were produced in 2008. According to EPA, the EIA data for the same period indicated the production of “coal combustion residues” for this same period of time was 131 million tons (*Federal Register*, 2010, p. 35212). EPA itself placed the amount at approximately 141 million tons (*Federal*

Register, 2010, p. 35212), with 94 million tons going for disposal versus ACAA (2009a)'s estimate of 75 million tons. In other words, while the PCC annual production value remained more or less unchanged, the amount of PCCs going to disposal increased by 20% in EPA estimates over ACAA reported values.

Because EPA PCC production estimates did not represent a significant change over ACAA values, it is unlikely that the unknown number of <100 MW coal-burning facilities excluded from EIA statistics is the primary cause of the observed 20% discrepancy (*Federal Register*, 2010, p. 35212). In other words, the 5-10 million ton increase in total national PCC production resulting from inclusion of these facilities in EPA's "revised" estimates by itself could not have accounted for the entire 19 million ton increase in annual PCC disposal rates provided in the EPA proposed rule.

EIA has noted that even values for the same variable can vary among the agency's own report series due to use of different "baseline" populations and sampling methodologies (*cf.* EIA, 2010d, p. ii; and my own observations in the "Key Assumptions" section of Chapter 1). Without a closer examination of the methods used by both EIA and EPA in this instance to derive their calculations, it is not currently possible for me to determine what other factors might be driving this 20% discrepancy. Regardless of the ultimate answer, however, I believe this example remains a compelling illustration of just how large – and basic – are some of the challenges facing regulators.

4. Although EIA (2010f) includes a comprehensive breakdown of coal distribution from all U.S. origin states to a variety of sectors, one drawback is that the reported distribution values in the 2006, 2007, and 2008 annual reports added delivery

statistics for synfuel to those of bituminous, subbituminous, and lignite coal, with no available indication as to what quantity of the total is composed by the synfuel versus the other, “traditional” coal types. While I suspect that synfuel remains only a small proportion at best of the total amount of fuel coal consumed annually for electrical generation purposes in the U.S., the very limited timeframe in which this paper was written precluded efforts to obtain more specific information on this matter from EIA databases.

5. Even a crude back-of-the-envelope calculation will demonstrate just how inaccurate is the historical landfilled fly ash value cited by EPA (2010a). If we total EIA (2010j)’s records for historical whole coal consumption in the U.S. over the 1970-2009 period and assume that even only 5% of this amount was transformed into coal ash – approximately half of the “standard” 10% estimate of ash by weight of whole coal – we obtain a result of 1.71 billion short tons of coal ash produced. Further assuming that historical ratios of the three coal ash types remain similar to modern values (*i.e.*, fly ash composing 70-75% of the total), as it does today, we may therefore estimate that about 1.20-1.28 billion short tons of this amount is fly ash. Finally, if we conservatively – and unrealistically – estimate that historical coal ash disposal techniques over this period were similar to modern ones (*i.e.*, that 30-40% of fly ash was utilized rather than disposed annually, and that approximately 30% of disposed fly ash was placed into surface impoundments rather than landfills), we obtain final results of between 503 and 629 million short tons of coal ash produced in the 1970-2009 period and disposed in landfills, thus easily exceeding the maximum 500 million short ton estimate cited by EPA (2010a).

Clearly this is only a rough estimate, and based on implausibly-conservative initial conditions; utilizing more-realistic values to represent historical disposal techniques and ash production would certainly increase these totals even further. It goes without saying, then, that the addition of fifty more years' annual totals for coal consumption in the U.S. to fill the gap between 1920 and 1970 would push these values far beyond the stated upper limits of the ACAA estimate.

CHAPTER 2

1. If the calculated "excess" risk level of exposure from a wastestream is calculated by EPA to be greater than or equal to 1 x 10^{-5} (one in one hundred thousand) above "baseline" risk levels for an individual's probability of developing cancer within his or her lifetime, the wastestream is considered to be at a "level of concern," and may be listed as a hazardous waste. At a risk level at or above 1 x 10^{-4} (one in ten thousand), the waste stream "generally will be considered to pose a substantial present or potential hazard to human health and the environment and generally will be classified as hazardous waste" (EPA, 2010c, p. 3).

   For non-cancer-causing contaminants present in wastestreams, EPA utilizes a hazard quotient (HQ), a simple ratio by which the estimated human exposure to the contaminant is divided by the exposure level at which no adverse health effects are expected to result. (In the case of human exposure from drinking water, this latter point is identical to MCL levels as established by the Safe Drinking Water Act.) Thus, a "risk of concern" in this pathway is an HQ > 1 (EPA, 2010c, p. 5).

2. EPA has identified 495 power plants nationally which utilize coal for electrical generation activities and would be affected by the June proposed rule; however, data from the 2000 U.S. Census was only available for 464 (94%) of these facilities. As such, EPA notes in *Federal Register* (2010, p. 35229) that it has extrapolated the available data to account for all 495 facilities in this demographic analysis, but neither the proposed rule nor the supplemental Appendices in EPA (2009a, p. 285-315) clarify what methods were used to perform these calculations.

CHAPTER 3

1. As reported by EPA (2009b) and duplicated in *Federal Register* (2010, p. 35141-35142), much knowledge regarding the variation of PCC leaching rates by pH has been only-recently derived, and even then only from a relatively small dataset. Specifically, the conclusions of EPA (2009b, p. x) are based on only 34 samples of fly ash, and nowhere does leaching data appear to have been uniquely obtained for either bottom ash or boiler slag (although it is possible these products might be included in the "blended CCRs" [coal combustion residues] category, the study is not clear on this matter). Yet these two coal ash types nevertheless compose about 20-25% of all annual coal ash production in the United States, and due to their vitreous rather than "fluffy" physical character, should exhibit very different profiles of surface-bound or encapsulated trace elements than fly ash particles (*cf.* Wang *et al.*, 1999).

   Furthermore, as per EPA (2009b, p. A1-A12), no geographic location (*i.e.*, city and/or state) is provided for 28 of the 31 facilities at which sampling took place, and information regarding the geographic origin of the fuel-coal utilized by these facilities is also sparse (with information either absent entirely or described only vaguely; “eastern bituminous coal,” for example, could potentially originate from any one of several states along the East Coast, but precise chemical composition would clearly differ by mining location). Finally, it remains unclear whether this variability in trace element content may be attributed only to the geographic origin of the fuel-coal, or whether other factors such as coal pre-treatment or post-combustion emissions-management technologies utilized by the power plants may play an additional role in the process.

The end result of these variations is perhaps best depicted by EPA (2010b, p. 4-47)’s graphical comparison of EPA (2009b) results to previous datasets, which notes that several of the facilities from the 2009 study are shown to fall at relatively-extreme percentile ranges of the elemental-concentration spectrum for PCCs. Thus, the EPA (2009b) leaching study may be best viewed not necessarily as a true depiction of “average” leaching levels on the greater national scale, but rather as an interesting case study of the challenges facing environmental regulators in attempting to determine what is an “average” versus “extraordinary” trace element content for PCCs in the first place.

2. My purpose in establishing this third set of user-adjustable variables was driven by recognition that, just as the age and pollution controls of coal-fired power plants would control their ultimate ash output, so too might the combustion and emissions-control processes utilized by these facilities affect the ultimate trace element composition of the coal ash. To my knowledge, only EPA (2006) has specifically compared variations in trace element levels in relation to emissions technologies among various facilities, but even this report has focused only on fly

ash samples, and then only from six facilities. Although EPA (2006, p. 15) notes that the "facility configuration is representative of 75% of the coal-fired utilities in the U.S.", the extremely-small nature of the sample size must necessarily lend itself to cautious review of the study results.

Although the EPA study is certainly a start, more leaching data is clearly necessary – both from a greater number of facilities, and considering a larger number of elements (trace radioisotopes such as U and Th, for example, were not included in the EPA (2006) analysis). As such, I do not feel comfortable speculating how concentrations might vary for other elements and other coal ash types at the current time; I have only built this capacity into my spreadsheets in anticipation of a future time at which such information might be available. If such values *did* exist, a (completely fictional) example "adjustment" in my calculations for trace element concentrations might appear as follows:

Knowing that mercury tends to be concentrated on the surfaces of vitrified fly ash particles, my "baseline" variables might show that 90% of the element levels present in the original whole coal remain in an average fly ash at a "Grandfathered"-category power plant after rudimentary emissions controls, as compared to a 70% respective presence in bottom ash and boiler slag outputs at the same facility, due to the different processes by which these particles are formed. However, if a "Modern"-category power plant utilized not only scrubbers but a sorbent injection to further treat the produced ash prior to its disposal, the percentage of mercury "transfer" from whole coal to the three coal ash types might be closer to 50% for fly ash, 20% for bottom ash, and 10% for bottom slag, this latter material for which the sorbent might show particular affinity. Thus, in my spreadsheets, the user would be able to input these element-transfer percentages for all coal ash types at both Grandfathered and Modern facilities, and thereby revise the ultimate trace element content of the coal ash output by both categories accordingly.

Alternately, of course, these same variables could be utilized to calculate *enrichment* of trace elements during combustion, by which some proportion of elements present in whole coal are not lost or otherwise consumed during combustion, but rather become highly-concentrated on surfaces of remaining "waste" particles (*i.e.,* coal ash and other PCCs). Thus, the same example could in fact reflect gains rather than losses of trace elements in the transformation from whole coal to coal ash: "Grandfathered" power plants might show a 200% enrichment of arsenic in fly ash in comparison to the baseline levels present in the original whole coal, whereas "Modern" power plants utilizing a different boiler technology might observe 250% enrichment rates for fly ash, where higher temperature combustion processes speed or otherwise encourage binding of elements to particle surfaces.

Once again, the above examples are highly-speculative and in no way based on actual field observations; they simply demonstrate how these adjustable variables might theoretically be employed to manipulate the final trace element concentrations in annual ash outputs by coal-burning power generators.

CHAPTER 4

1. For the sake of argument, my calculations in this chapter have focused on the possibility of reclassifying coal ash as Class A low-level radioactive waste. In reality, however, it is extremely unlikely that coal ash would ever earn this

classification – if it is to be regulated, it will be out of concern for the potential leaching of trace elements such as heavy metals, not radionuclides.

As shown in Table D1 of Appendix D, NRC does not provide any specific lower boundary for activity levels to distinguish Class A LLRW from naturally-occurring radioactive material (NORM), the category to which trace element-containing whole coal belongs. This is because, as per *Federal Register* (1995, p. 37560-37561),

> Naturally-occurring radioactive material and accelerator-produced radioactive material lie outside NRC's regulatory authority and are subject to health and safety regulations by States and other Federal agencies.

The upper defined limits of NORM therefore vary on the national scale according to individual state environmental policies, but Lowenthal (1997, p. 9) notes that two general subclassifications appear to be present: "diffuse" waste, with activity levels of <2 nCi/g ^{226}Ra or equivalent, and "discrete" waste, with activity levels >2 nCi/g ^{226}Ra or equivalent. For comparison, then, the estimates provided by EPA (2009c) for low, average, and high activity levels in fly ash are 0.0020, 0.0058, and 0.0097 nCi/g, respectively, which would place them firmly in the NORM ranges of least concern – indeed, with radioactivity levels lower than some naturally-occurring geologic deposits (*cf.* USGS, 1997, p. 2).

2. This concept of nonlinear response to chemical concentrations is formally known as "hormesis," and – as with many emerging areas of scientific thought – remains controversial. Coinciding with the equally-recent trend of research interest into epigenetics, research interest into hormesis has only truly gained traction within the last decade, but it may have particularly relevant applications for regulations

regarding “safe” levels of endocrine-disrupting or -mimicking chemicals. Furthermore, if future research confirms the conclusions of researchers such as Rothenberg and Rothenberg (2005) and Sharma and Dubé (2004) regarding nonlinear dose-response relationships for elements such as lead and arsenic, this may require EPA to rethink current MCL classifications in order to better protect children’s health, as this segment of the population is generally most vulnerable to contaminants, particularly heavy metals.

Although a more complete discussion of the research and controversy within this field is beyond the scope of this particular paper, interested parties may wish to begin with a recent overview by leading researchers Davis and Svensgaard (2010). For a more critical counterpoint, see also Mushak (2007)’s historical overview of hormesis research between 1990 and 2005.

Appendix A: Background Material

Table A1: Key differences between RCRA Subtitle C and Subtitle D approaches in EPA’s 2010 proposed rule for PCC disposal regulation.

	Subtitle C approach	Subtitle D approach	Subtitle “D prime” approach
Permit Issuance	States must issue permits for PCC transport and disposal activities.	No permits are required.	
Requirements for PCC Storage	Yes; PCC disposers must meet federal guidelines.	No storage requirements.	
Surface Impoundments Constructed Prior to Rule Finalization	Would create strong incentives to phase-out use of existing surface impoundments in favor of a transition to landfills for PCC disposal and storage. All solids must be removed from the impoundment within five years of rule finalization. The site can then either close, or be retrofitted with a liner and otherwise modified to meet land disposal criteria so that it can continue receiving (dry) PCC wastestreams.		Existing surface impoundments can continue to operate for their useful lifespans. Regular monitoring will be required for sites identified as having “High” or “Significant” hazard potential ratings via NID criteria.
Surface Impoundments Constructed After Rule Finalization	New surface impoundments must meet both land disposal restrictions and liner requirements.	No land disposal restrictions imposed on new surface impoundments, but composite liners must be installed.	

Landfills Constructed Prior to Rule Finalization	No liner requirements, but groundwater monitoring is required.		
Landfills Constructed After Rule Finalization	Liner requirements must be met, and groundwater monitoring is required.		
Requirements for Closure and Post-Closure Care	Yes. States and EPA monitor and take corrective action when necessary.	Yes. Monitoring and corrective action are self-implementing.	
Enforcement	State and federal enforcement.	Enforcement through citizen suits; states can act as citizens.	
Annual Regulatory Costs, Present Values (in millions, $USD, 2009) *	$20,349	$8,095	$3,259
Annual Net Benefits, Present Values (in millions, $USD, 2009) **	**A:** $66,872 to $81,842 **B:** ($251,166) to ($236,196) **C:** ($17,617) to ($2,647)	**A:** $26,869 to $33,666 **B:** ($6,297) to ($130) **C:** ($6,297) to ($130)	**A:** $10,852 to $14,242 **B:** ($2,666) to $724 **C:** ($2,666) to $724
Annual Regulatory Costs, AAEV (in millions, $USD, 2009) *	$1,474	$587	$236
Annual Net Benefits, AAEV (in millions, $USD, 2009) **	**A:** $4,845 to $5,930 **B:** ($18,199) to ($17,115) **C:** ($1,277) to ($192)	**A:** $1,947 to $2,439 **B:** ($502) to ($9) **C:** ($502) to ($9)	**A:** $786 to $1,032 **B:** ($193) to $52 **C:** ($193) to $52

Notes:

* As per *Federal Register* (2010, p. 35215-35217), costs and benefits for both "present value" and "average annualized equivalent value" categories (this latter for reasons of space abbreviated to "AAEV") are presented in 2009 U.S. dollars. Values enclosed in parentheses are negative (*i.e.*, representing a net loss).

AAEV values have been calculated by multiplying 50-year present values by a 7% discount rate (*i.e.,* the regulatory costs and benefits listed here are intended to reflect average annual values over the time period between 2012 and 2061).

** As listed here, the categories of "A," "B," and "C" represent the following:

A: The cost/benefit calculation EPA deems "most likely" to occur as a result of the proposed rule; *i.e.,* higher disposal costs under subtitle C regulation will induce industry instead to seek out beneficial-use markets which can utilize PCC material.

B: "Negative" scenario for beneficial use, in which the EPA proposed rule creates a negative stigma discouraging PCC reuse by industry.

C: "Neutral" scenario for beneficial use, in which the EPA proposed rule does not impact industry opinion of PCCs, and beneficial use neither increases nor decreases over current reuse rates.

Source:

All information from *Federal Register* (2010) and EPA (2010f).

Table A2: Summarized results of EPA risk modeling of human and environmental health risks associated with constituent leaching and migration from PCC landfills and surface impoundments over a 10,000-year period of analysis.

Exposure Pathway	**Risk Percentile**	**Landfills Managing CCRs Only**		**Surface Impoundments Managing CCRs Only**		
		Unlined	**Clay-Lined**	**Unlined**	**Clay-Lined**	**Synthetic Liner**
A	50th	6×10^{-6} (Cancer risk, As)	4×10^{-6} (Cancer risk, As)	1×10^{-4} (Cancer risk, As)	6×10^{-5} (Cancer risk, As)	n/a
	90th	4×10^{-4} (Cancer risk, As) HQ = 3 for Tl HQ = 2 for Sb HQ = 2 for Mo	2×10^{-4} (Cancer risk, As) HQ = 2 for Tl	2×10^{-3} (Cancer risk, As) Nitrate/nitrite exceeded MCL by factor of 20; Pb by a factor of 3 HQ = 8 for Mo HQ = 7 for B HQ = 2 for Se	9×10^{-4} (Cancer risk, As) Nitrate/nitrite exceeded MCL by factor of 10 HQ = 5 for Mo HQ = 4 for B	n/a
B	50th	1×10^{-6} (Cancer risk, As)	n/a	n/a	n/a	n/a

	90th	n/a	n/a	8 x 10^{-6} (Cancer risk, As) HQ = 3 for Se	4 x 10^{-6} (Cancer risk, As) HQ = 2 for Se	n/a
C	50th	n/a	n/a	HQ = 7 for B	n/a	n/a
	90th	HQ = 281 for B HQ = 8 for Pb HQ = 2 for arsenic V HQ = 2 for selenium VI HQ = 2 for Ba	HQ = 78 for B	HQ = 2,375 for B HQ = 22 for Pb HQ = 13 for arsenic V HQ = 12 for selenium VI HQ = 6 for Co HQ = 3 for Ba	HQ = 854 for B HQ = 7 for Pb HQ = 4 for arsenic V HQ = 4 for selenium VI HQ = 3 for Co	HQ = 257 for B HQ = 2 for Pb HQ = 5 for arsenic V HQ = 5 for Co
D	50th	n/a	n/a	n/a	n/a	n/a
	90th	HQ = 58 for Pb HQ = 11 for arsenic III HQ = 5 for Cd HQ = 2 for Sb	HQ = 3 for arsenic III	HQ = 311 for Pb HQ = 127 for arsenic III HQ = 30 for Cd	HQ = 58 for Pb HQ = 55 for arsenic III HQ = 9 for Cd	HQ = 4 for Pb HQ = 31 for arsenic III HQ = 2 for Cd

Notes:

The four exposure pathways in EPA analysis represented by the letters above may be defined as follows:

- **A:** Human exposure by ingestion of groundwater contaminated by leachate migration plumes
- **B:** Human exposure from consumption of fish affected by leachate migration to nearby surface water bodies
- **C:** Exposure for aquatic organisms living in nearby water bodies contaminated by leachate migration plumes
- **D:** Exposure for organisms which come into direct contact with or ingest sediment in nearby water bodies
  contaminated by leachate migration plumes

"Percentile risk" here refers to modeled nationwide exposure risks to leached contaminants that are either in the 50th percentile (also called "central tendency estimates") or the 90th percentile ("high-end risk estimates"), respectively.

The disposal units referred to in this table receive coal combustion residues (CCRs) only; that is, they do not co-manage wastestreams of either FBC waste or those of "coal refuse metal" (which I presume is used in reference to coal mine tailings).

Cells marked as "n/a" indicate that either the modeled rate of leaching from the disposal unit was too low for the contaminant plume to reach the water body or drinking water source in the 10,000-year timeframe of analysis, or that the contamination which occurred fell below the threshold for significant risk.

Note that while a subcategory exists in the above table for surface impoundments with synthetic liners, no similar subcategory exists for PCC landfills; this is because the EPA report found no instances where leaching posed a leaching hazard significantly above the risk criteria (*i.e.*, all results would have been redundantly recorded here as "n/a").

Source:

EPA, 2010b.

Appendix B: Coal Ash Tonnage Calculations

Table B1: “Unadjusted” quantity of coal ash theoretically able to be produced from combustion of coal by electrical generation activities in the United States, 2008.

State of Origin of Coal Delivery	**Total Amount of Whole Coal Exported per State for Electrical Generation Use in Other States, 2008 (short tons) [1]**	**Percentage of Ash by Weight Present in Annual Coal Shipments from State of Origin [2]**	**Total Amount of Coal Ash Theoretically Able to be Produced by Combustion for Electrical Generation Activities, 2008 (short tons)**
Alabama	8,977,000	13.02	1,168,805
Arizona	7,958,000	10.05	799,779
Colorado	29,932,000	9.70	2,903,404
Illinois	23,181,000	8.89	2,060,791
Indiana	32,821,000	9.21	3,022,814
Kansas	180,000	12.96	23,328
Kentucky	107,639,000	10.53	11,334,387
Louisiana	3,855,000	11.56	445,638
Maryland	1,842,000	17.43	321,061
Mississippi	3,018,000	15.92	480,466
Missouri	188,000	15.86	29,817
Montana	38,914,000	7.68	2,988,595
New Mexico	23,046,000	19.70	4,540,062
North Dakota	23,827,000	9.88	2,354,108
Ohio	23,204,000	10.24	2,376,090
Oklahoma	559,000	26.28	146,905
Pennsylvania (Anthracite) *	0	16.29	0

Pennsylvania (Bituminous) *	49,884,000	16.29	8,126,104
Tennessee	1,755,000	9.44	165,672
Texas	37,925,000	16.37	6,208,323
Utah	21,879,000	11.83	2,588,286
Virginia	10,632,000	11.17	1,187,594
Washington	-	-	-
West Virginia	101,161,000	11.77	11,906,650
Wyoming	450,968,000	5.18	23,360,142
State Totals	**1,003,345,000**		**88,538,819**

Notes:

* Because EIA data is provided only by state, and does not distinguish between different varieties of coal, I have chosen to utilize the same values for percentage ash composition of both anthracite and bituminous coal from Pennsylvania.

Note also that this table and Table B2 below both distinguish between zero (“0”) and dashed (“-”) values. Specifically, a zero value means that while the state in question did produce coal for use by other sectors in a given year (*e.g.,* coke plants, or other industrial and commercial uses), none of this coal was utilized by the electrical generation sector itself. A dashed value, on the other hand, indicates that values for the state’s coal production are missing from EIA reports for that year, and thus may or may not necessarily equal zero.

Sources:

[1] EIA, 2010e
[2] EIA, 2010l

Table B2: “Unadjusted” quantity of all coal ash theoretically able to be produced by combustion of coal by electrical generation activities in the United States annually, 2001-2008.

State of Origin of Coal Delivery	**Total Theoretical Coal Ash Production by Electrical Generation Activities (short tons)**							
	2001 [a, i]	**2002** [b, j]	**2003** [c, k]	**2004** [d, l]	**2005** [e, m]	**2006** [f, n]	**2007** [g, o]	**2008** [h, p]
Alabama	925,688	164,383	186,112	193,088	202,288	1,336,801	1,198,298	1,168,805
Arizona	1,261,728	1,243,919	1,187,533	1,257,152	1,246,024	764,170	773,064	799,779
Colorado	2,497,900	2,684,999	2,895,624	2,882,855	2,848,745	2,804,089	3,115,253	2,903,404
Illinois	2,430,775	2,050,763	1,832,729	1,703,124	1,747,824	2,105,855	2,248,384	2,060,791
Indiana	2,940,710	2,353,835	2,040,151	1,715,385	1,802,543	2,981,540	2,884,483	3,022,814
Kansas	35,376	40,426	24,902	9,094	21,539	64,090	64,596	23,328
Kentucky	10,801,143	8,737,663	6,881,004	7,355,910	9,035,992	11,757,852	10,447,219	11,334,387
Louisiana *	510,611	505,702	541,591	-	518,879	488,704	408,437	445,638
Maryland	753,944	610,426	580,490	461,237	612,042	927,784	327,184	321,061
Mississippi *	-	474,840	582,162	549,374	561,347	567,694	545,129	480,466
Missouri	60,683	37,575	60,548	56,277	90,711	60,801	33,205	29,817
Montana	2,321,029	2,399,597	2,437,467	2,612,487	2,624,256	2,704,143	2,708,357	2,988,595
New Mexico	5,192,250	5,137,039	5,164,622	5,019,451	4,879,158	4,990,107	4,836,446	4,540,062
North Dakota	2,334,312	2,321,654	2,373,385	2,331,377	2,299,097	2,419,430	2,387,814	2,354,108
Ohio	2,551,701	1,823,854	1,583,674	1,805,790	1,780,211	1,961,298	1,989,967	2,376,090
Oklahoma	106,582	149,883	164,692	150,560	213,443	211,515	181,100	146,905

Pennsylvania (Anthracite)	124,603	153,284	185,871	283,258	257,730	218,089	241,033	0
Pennsylvania (Bituminous)	4,326,840	4,632,125	4,975,895	6,747,077	6,020,090	8,046,327	8,605,603	8,126,104
Tennessee	166,323	172,917	136,040	110,945	143,986	192,592	157,165	165,672
Texas	7,024,443	6,839,117	7,652,552	7,252,544	7,088,514	7,063,728	6,767,099	6,208,323
Utah	1,856,907	1,985,445	2,129,995	2,445,750	2,312,741	2,544,747	2,590,409	2,588,286
Virginia	1,771,621	1,444,488	1,523,405	1,475,184	1,527,944	1,936,121	1,441,760	1,187,594
Washington **	-	1,164,235	974,062	853,038	807,804	420,540	-	-
West Virginia	13,925,934	11,202,656	9,708,822	9,128,707	8,969,110	12,826,294	13,679,761	11,906,650
Wyoming	18,572,978	18,671,155	18,693,906	19,627,285	20,192,986	22,742,848	22,951,708	23,360,142
State Totals	**82,494,080**	**77,001,980**	**74,517,234**	**76,026,945**	**77,805,004**	**92,137,159**	**90,583,474**	**88,538,819**

Notes:

As described in the notes to Table B1 above, cells containing dashed values indicate that EIA data is missing for these state and year combinations. Specifically, states listed here with single- or double-star designations indicate that they belong to the following respective categories:

* No EIA annual data is available for the average percentage ash content of coal deliveries originating from Louisiana in 2004, or from Mississippi in 2001. This resulted, respectively, in 3,889,000 million short tons of coal being excluded from the total 2004 analysis, and in 475,000 million short tons of coal being excluded from the 2001 analysis.

** No EIA annual data is available for the average percentage ash content of coal deliveries originating from the state of Washington in 2001, resulting in exclusion of 4,623,000 million short tons of coal from the total 2001 analysis.

Additionally, in 2007 and 2008, information was missing both for average annual percentage ash content of coal deliveries from Washington state, as well as for overall annual coal production by the state. From the pattern established by previous years, it is likely that production values were non-zero during this timeframe, but without further information it is not possible to determine what final quantity of coal would be involved in exclusion of these two null datapoints.

In both cases, then, exclusion of data results in overall underreporting of actual annual coal ash production values for the year in question.

Sources:

[a] EIA, 2007a
[b] EIA, 2007b
[c] EIA, 2007c
[d] EIA, 2007d
[e] EIA, 2007e
[f] EIA, 2008a
[g] EIA, 2009a
[h] EIA, 2010e
[i] EIA, 2004, p. 2
[j] EIA, 2006a, p. 5
[k] EIA, 2006b, p. 5
[l] EIA, 2006c, p. 5
[m] EIA, 2007f, p. 5
[n] EIA, 2008b, p. 5
[o] EIA, 2010d, p. 5
[p] EIA, 2010d, p. 5

Table B3: User-adjustable variables utilized to refine the coal ash production and disposal calculations by adjusting for power plant technology level.

Facility Type	*Grandfathered*	*Modern*
Percentage of Industry Composed by This Facility Type	20%	80%
Average Percentage of Each Coal Ash Type Captured by the Facility Type and Prevented from Loss to Atmosphere		
Fly Ash	95%	99%
Bottom Ash	99%	99%
Boiler Slag	100%	100%

Notes:

As described in Chapter 1, the values depicted above are only arbitrary “placeholders,” and are entirely adjustable by the user within the spreadsheet itself. I have simply begun with these values to provide a rough demonstration of how variations in coal ash treatment and capture technologies may affect overall ash production on a national scale. Thus, all values derived prior to application of these additional variables are known as “unadjusted,” and those which have been modified by these multipliers, “adjusted,” respectively.

Note also that, as described in Chapter 3 and associated footnotes, the spreadsheet has been constructed to allow individual manipulation of the “carryover” trace element percentages (*i.e.*, the percentage of trace element originally present in the whole coal which is retained in the coal ash) for each coal ash type produced by both Grandfathered and Modern facility types. For the moment these variables have all been set at defaults of 100% (indicating full carryover, and thus not currently modifying any output values), but these, too, are user-adjustable, and may be modified at will.

Table B4: “Adjusted” quantity of all coal ash produced by combustion of coal by electrical generation activities in the United States and captured by the generator prior to use or disposal, 2001-2008.

State of Origin of Coal Delivery	Total Adjusted Coal Ash Production by Electrical Generation Activities (short tons)							
	2001 [a, i, q]	**2002** [b, j, r]	**2003** [c, k, s]	**2004** [d, l, t]	**2005** [e, m, u]	**2006** [f, n, v]	**2007** [g, o, w]	**2008** [h, p, x]
Alabama	840,469	167,888	172,484	174,758	203,513	1,200,871	1,095,276	1,102,490
Arizona	1,145,572	1,270,437	1,100,577	1,137,813	1,253,570	686,467	706,601	754,401
Colorado	2,267,941	2,742,238	2,683,596	2,609,189	2,865,999	2,518,959	2,847,425	2,738,671
Illinois	2,206,996	2,094,481	1,698,530	1,541,449	1,758,409	1,891,724	2,055,083	1,943,866
Indiana	2,669,985	2,404,015	1,890,764	1,552,546	1,813,461	2,678,366	2,636,494	2,851,306
Kansas	32,119	41,288	23,078	8,230	21,669	57,573	59,042	22,004
Kentucky	9,806,779	8,923,934	6,377,151	6,657,623	9,090,718	10,562,272	9,549,037	10,691,299
Louisiana *	463,603	516,483	501,934	-	522,022	439,010	373,323	420,354
Maryland	684,536	623,439	537,985	417,452	615,749	833,444	299,055	302,844
Mississippi *	-	484,963	539,534	497,222	564,747	509,969	498,263	453,205
Missouri	55,096	38,376	56,114	50,935	91,261	54,618	30,350	28,125
Montana	2,107,352	2,450,752	2,258,987	2,364,487	2,640,150	2,429,177	2,475,511	2,819,029
New Mexico	4,714,246	5,246,552	4,786,449	4,542,961	4,908,709	4,482,695	4,420,640	4,282,469
North Dakota	2,119,413	2,371,148	2,199,597	2,110,062	2,313,021	2,173,414	2,182,526	2,220,541
Ohio	2,316,788	1,862,735	1,467,712	1,634,369	1,790,993	1,761,867	1,818,883	2,241,276
Oklahoma	96,770	153,078	152,632	136,267	214,736	190,007	165,531	138,570

Pennsylvania (Anthracite)	113,132	156,552	172,261	256,369	259,291	195,913	220,311	0
Pennsylvania (Bituminous)	3,928,507	4,730,873	4,611,542	6,106,585	6,056,551	7,228,148	7,865,751	7,665,047
Tennessee	151,011	176,603	126,079	100,413	144,858	173,009	143,653	156,272
Texas	6,377,766	6,984,914	7,092,203	6,564,070	7,131,445	6,345,463	6,185,309	5,856,076
Utah	1,685,958	2,027,771	1,974,028	2,213,578	2,326,748	2,285,988	2,367,703	2,441,432
Virginia	1,608,524	1,475,282	1,411,855	1,335,147	1,537,198	1,739,249	1,317,807	1,120,213
Washington **	-	1,189,054	902,737	772,060	812,697	377,778	-	-
West Virginia	12,643,899	11,441,476	8,997,906	8,262,131	9,023,431	11,522,070	12,503,666	11,231,093
Wyoming	16,863,132	19,069,189	17,325,068	17,764,093	20,315,285	20,430,275	20,978,474	22,034,741
State Totals	**74,899,595**	**78,643,521**	**69,060,804**	**68,809,808**	**78,276,230**	**82,768,326**	**82,795,714**	**83,515,326**

Notes:

Just as for Tables B1 and B2 above, cells containing dashed values indicate that EIA data is missing for these state and year combinations. Specifically, states listed here with single- or double-star designations indicate that they belong to the following respective categories:

* No EIA annual data is available for the average percentage ash content of coal deliveries originating from Louisiana in 2004, or from Mississippi in 2001. This resulted, respectively, in 3,889,000 million short tons of coal being excluded from the total 2004 analysis, and in 475,000 million short tons of coal being excluded from the 2001 analysis.

** No EIA annual data is available for the average percentage ash content of coal deliveries originating from the state of Washington in 2001, resulting in exclusion of 4,623,000 million short tons of coal from the total 2001 analysis.

Additionally, in 2007 and 2008, information was missing both for average annual percentage ash content of coal deliveries from Washington state, as well as for overall annual coal production by the state. From the pattern established by previous years, it is likely that production values were non-zero during this timeframe, but without further information it is not possible to determine what final quantity of coal would be involved in exclusion of these two null datapoints.

In both cases, then, exclusion of data results in overall underreporting of actual annual coal ash production values for the year in question.

Sources:

[a] EIA, 2007a
[b] EIA, 2007b
[c] EIA, 2007c
[d] EIA, 2007d
[e] EIA, 2007e
[f] EIA, 2008a
[g] EIA, 2009a
[h] EIA, 2010e
[i] EIA, 2004, p. 2
[j] EIA, 2006a, p. 5
[k] EIA, 2006b, p. 5
[l] EIA, 2006c, p. 5
[m] EIA, 2007f, p. 5
[n] EIA, 2008b, p. 5
[o] EIA, 2010l, p. 5
[p] EIA, 2010l, p. 5
[q] ACAA, 2003a
[r] ACAA, 2003b
[s] ACAA, 2004
[t] ACAA, 2005
[u] ACAA, 2008a
[v] ACAA, 2007, p. 2
[w] ACAA, 2008b
[x] ACAA, 2009a

Table B5: “Adjusted” quantity of all coal ash types produced by combustion of coal by electrical generation activities in the United States, captured by the generator, and placed into either landfills or surface impoundments for disposal annually, 2001-2008.

	Amount of Coal Ash Type Produced and Placed in Landfills Annually (in short tons)			**Amount of Coal Ash Type Produced and Placed in Surface Impoundments Annually (in short tons)**		
Year	**Fly Ash**	**Bottom Ash**	**Boiler Slag**	**Fly Ash**	**Bottom Ash**	**Boiler Slag**
2001	26,534,474	7,891,511	190,662	11,997,241	3,122,051	419,980
2002	27,443,418	6,990,559	93,908	12,408,209	2,765,615	206,856
2003	22,661,969	5,445,196	19,517	10,246,334	2,154,236	42,991
2004	22,402,688	4,975,661	55,411	10,129,104	1,968,478	122,057
2005	24,912,617	6,260,914	18,243	11,263,938	2,476,951	40,185
2006	24,444,259	6,556,176	94,597	11,052,175	2,593,763	208,373
2007	24,819,904	7,014,282	116,878	11,222,019	2,775,000	257,452
2008	26,137,594	6,709,935	96,564	11,817,796	2,654,594	212,706
Total	**199,356,923**	**51,844,234**	**685,781**	**90,136,816**	**20,510,687**	**1,510,599**

Notes:

The ratios of coal ash types placed in landfills versus surface impoundments for disposal annually is derived from ACAA (2003) data for 2001 ash disposal statistics. Because subsequent annual ACAA reports lacked these disposal type breakdowns, these ratios were assumed to remain constant for 2002-2008 ash disposal calculations.

Table B6: Comparison of “unadjusted” quantities of all coal ash types versus all products of coal combustion (PCCs) produced by electrical generation activities in the United States, captured by the generator, and placed into disposal annually, 2001-2008.

	“Adjusted” Amount of Coal Ash Produced and Placed into Disposal Annually (short tons)		**“Adjusted” Amount of PCCs Produced and Placed into Disposal Annually (short tons)**	
Year	**Produced**	**Disposed**	**Produced**	**Disposed**
2001	82,494,080	50,973,452	108,762,619	68,754,709
2002	77,001,980	50,737,589	100,900,757	67,687,117
2003	74,517,234	41,250,211	100,704,272	58,712,599
2004	76,026,945	40,319,847	103,219,644	56,885,690
2005	77,805,004	45,724,224	105,670,656	64,509,653
2006	92,137,159	45,692,414	123,602,648	63,829,778
2007	90,583,474	46,965,062	129,287,619	68,771,529
2008	88,538,819	48,419,500	129,665,653	68,948,938
Total	**659,104,695**	**370,082,300**	**901,813,869**	**518,100,012**

Table B7: Comparison of “adjusted” quantities of all coal ash types versus all products of coal combustion (PCCs) produced by electrical generation activities in the United States, captured by the generator, and placed into disposal annually, 2001-2008.

	“Adjusted” Amount of Coal Ash Produced and Placed into Disposal Annually (short tons)		**“Adjusted” Amount of PCCs Produced and Placed into Disposal Annually (short tons)**	
Year	**Produced**	**Disposed**	**Produced**	**Disposed**
2001	74,899,595	50,155,920	98,749,826	67,651,994
2002	78,643,521	49,908,564	103,051,776	66,581,146
2003	69,060,804	40,570,242	93,330,328	57,744,780
2004	68,809,808	39,653,399	93,421,141	55,945,424
2005	78,276,230	44,972,848	106,310,652	63,449,581
2006	82,768,326	44,949,343	111,034,293	62,791,749
2007	82,795,714	46,205,534	118,172,336	67,659,343
2008	83,515,326	47,629,189	122,308,716	67,823,541
Total	**618,769,324**	**364,045,039**	**846,379,068**	**509,647,557**

Appendix C: Trace Element Tonnage Calculations

Table C1: Relationship between individual states and the general "geographic subdivisions" of the United States, as adapted from Swanson *et al.* (1976).

Geographic Subdivision *	**State Assignments ****
Appalachia	Alabama [a] Kentucky Maryland Ohio Pennsylvania (Bituminous) Tennessee Virginia West Virginia
Pennsylvania (Anthracite)	Pennsylvania (Anthracite)
Interior	Illinois [b] Indiana Kansas Missouri Oklahoma
Northern Great Plains	Montana North Dakota Wyoming [a]
Gulf	Alabama [a] Louisiana [b] Mississippi Texas [b]
Rocky Mountains	Arizona Colorado New Mexico Utah Wyoming [a]

Notes:

* Swanson *et al.* (1976) classifies one or more states into geographic "regions," which are themselves part of larger "province" categories. For the purposes of this report, note that what I here term "geographic subdivision" is generally equivalent to the "province" designation of Swanson *et al.*, with the exception of the "Pennsylvania anthracite" and "Appalachian" subdivisions. In the original text, these are technically regions composing the larger "Eastern" province; I have here chosen to deal with them individually because the trace element profile of Pennsylvania anthracite is quite unique among other states of the province, and annual production of this coal type large enough to merit special consideration.

** As noted in the table above, states marked with a superscript "a" (*i.e.,* Alabama and Wyoming) are actually listed twice among the regional designations of Swanson *et al.*: Alabama belonging to both the Appalachia and Gulf provinces, and Wyoming to both the Rocky Mountain and Northern Great Plains provinces. For simplicity of calculations, I assumed that trace element concentrations of whole coal produced from these states could be represented by the average of those from both of the Swanson-assigned regions. Thus, statewide trace element concentrations in whole coal produced in Wyoming were calculated by averaging geometric means for trace element concentrations for the Northern Great Plains and Rocky Mountain regions; and trace element values for whole coal produced in Alabama were assumed to be an average of the geometric means for Appalachia and Gulf regions.

States marked with a superscript "b" in the table (*i.e.,* Louisiana, Illinois, and Texas) were present in the EIA coal production data I utilized in Chapter 1 for PCC production and disposal calculations, but were not present in the Swanson *et al.* datasets. I here assigned the "missing" states to the Swanson *et al.* province of closest geographical proximity, hoping that coal formations in the missing states might plausibly share the same geologic "history," and thus the same flat trace element concentrations in produced coal. Thus, Illinois was assigned the same trace element averages as the "Interior" province, whereas Louisiana and Texas were placed with other "Gulf" states.

Finally, although Swanson *et al.* provided trace element data for coal samples from Washington state, the number of these samples was so few (n=3, versus the n=34-331 range for other U.S. provinces) that I chose to err on the site of caution and exclude this data rather than introduce another potential source of error into the calculations. Thus, although this results in an underestimate of total trace element concentrations present in PCCs produced within the 2001-2008 timeframe, it appears unlikely that this will significantly impact final results, as the percentage of whole coal produced by Washington state annually is relatively small in comparison to national totals (*cf.* EIA, 2010e; EIA, 2010f).

Sample Calculation:

Demonstration of the region-to-state breakdown when calculating levels of selenium present in annual whole coal production by three different states:

All coal produced in Texas would share the same geometric mean concentration of selenium as the “Gulf” region (5.8 ppm), and all coal produced in Ohio would share the same geometric mean concentration as the “Appalachia” region (3.5 ppm). Because I have qualified Alabama as a “border state” sharing the characteristics of both regions, however, selenium levels in this state’s annual coal production are a simple average of the above two values (*i.e.*, (5.8+3.5 ppm) / 2 = 4.65 ppm).

Source:

Swanson *et al.*, 1976

Table C2: Geometric mean quantity of arsenic theoretically present in all whole coal produced and utilized for electrical generation activities in the United States annually, 2001-2008.

State of Origin of Coal Delivery	**Geometric Mean Arsenic Content (in short tons) of Domestically-Produced Whole Coal Utilized for Electrical Generation Activities**							
	2001	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**
Alabama	58	11	12	13	12	80	76	72
Arizona	13	13	12	13	13	8	8	8
Colorado	30	31	31	31	31	29	32	30
Illinois	337	295	269	247	249	291	314	278
Indiana	391	312	282	239	245	398	386	394
Kansas	2	2	2	1	2	5	5	2
Kentucky	1,078	920	725	751	925	1,198	1,088	1,184
Louisiana	19	19	20	19	20	21	16	19
Maryland	48	46	48	41	49	71	23	20
Mississippi	2	15	19	18	18	18	17	15
Missouri	4	2	4	5	7	5	3	2
Montana	74	71	69	75	75	79	81	78
New Mexico	28	27	27	27	26	26	24	23
North Dakota	48	50	50	49	48	48	47	48
Ohio	260	194	185	214	205	227	233	255

Oklahoma	12	11	13	11	11	11	9	7
Pennsylvania (Anthracite)	6	5	5	8	7	7	7	0
Pennsylvania (Bituminous)	583	435	395	504	466	674	696	549
Tennessee	23	22	16	13	18	26	20	19
Texas	211	216	224	218	218	210	208	190
Utah	21	19	19	19	18	21	21	22
Virginia	184	164	163	148	150	196	144	117
Washington	-	-	-	-	-	-	-	-
West Virginia	1,218	1,112	948	862	841	1,181	1,283	1,113
Wyoming	527	541	543	569	586	655	662	676
State Totals	**5,177**	**4,533**	**4,082**	**4,094**	**4,242**	**5,485**	**5,403**	**5,121**

Table C3: Geometric mean quantity of 28 trace elements theoretically present in all whole coal produced and utilized for electrical generation activities in the United States annually, 2001-2008.

Trace Element	**Geometric Mean Trace Element Content (in short tons) of Domestically-Produced Whole Coal Utilized for Electrical Generation Activities, 2001-2008**								
	2001	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**Total**
As	5,177	4,533	4,082	4,094	4,242	5,485	5,403	5,121	**38,136**
B	44,294	43,249	42,596	43,498	44,333	49,876	49,956	49,378	**367,181**
Ba	142,398	140,268	138,081	142,732	145,679	163,014	169,663	162,856	**1,198,690**
Be	1,024	915	839	846	873	1,095	1,079	1,026	**7,696**
Cd	239	225	214	219	225	264	263	257	**1,906**
Co	3,010	2,707	2,503	2,514	2,589	3,222	3,186	3,043	**22,773**
Cr	8,130	7,329	6,768	6,853	7,062	8,751	8,637	8,224	**61,753**
Cu	10,609	9,718	9,107	9,219	9,476	11,513	11,406	10,953	**82,001**
F	49,325	46,131	43,916	44,647	45,758	54,105	53,796	52,211	**389,888**
Ga	3,737	3,382	3,128	3,178	3,276	4,046	3,992	3,813	**28,552**
Hg	81	74	69	70	72	88	87	84	**626**
Li	9,554	8,604	7,912	8,056	8,313	10,326	10,189	9,706	**72,660**
Mn	83.116	72,903	65,249	66,306	68,847	88,952	87,465	83,008	**615,846**
Mo	1,328	1,218	1,137	1,158	1,193	1,453	1,444	1,398	**10,328**

Nb	2,595	2,462	2,355	2,425	2,486	2,896	2,890	2,830	**20,940**
Ni	7,850	6,958	6,359	6,367	6,573	8,330	8,210	7,774	**58,422**
Pb	6,874	6,179	5,696	5,729	5,907	7,365	7,312	7,030	**52,093**
Sb	510	466	434	440	453	554	549	528	**3,935**
Sc	2,133	1,969	1,858	1,880	1,930	2,322	2,300	2,214	**16,607**
Se	1,947	1,755	1,628	1,637	1,685	2,086	2,054	1,950	**14,741**
Sr	73,663	70,580	68,304	70,021	71,618	82,486	82,228	80,547	**599,447**
Th	2,233	2,095	1,994	2,041	2,094	2,471	2,459	2,389	**17,775**
U	900	843	807	816	835	985	979	947	**7,113**
V	13,346	12,234	11,486	11,607	11,923	14,451	14,306	13,730	**103,084**
Y	5,427	5,046	4,798	4,853	4,973	5,913	5,861	5,648	**42,518**
Yb	489	451	426	429	441	531	525	503	**3,795**
Zn	13,813	12,675	12,017	11,919	12,215	14,806	14,745	14,254	**106,443**
Zr	20,076	18,634	17,617	17,947	18,424	22,019	21,793	21,000	**157,510**

Table C4: Unadjusted ("ZCF") quantity of arsenic theoretically present in all coal ash ("adjusted" volumes) produced by combustion of coal by electrical generation activities in the United States, captured, and placed into disposal annually, 2001-2008.

State of Origin of Coal Delivery	**"ZCF" Arsenic Content (in short tons) of Coal Ash Produced by Electrical Generation Activities and Placed into Disposal, 2001-2008**							
	2001	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**
Alabama	4	1	1	1	1	5	5	5
Arizona	1	1	1	1	1	0	0	0
Colorado	1	2	2	1	2	1	2	2
Illinois	17	16	12	10	12	12	14	13
Indiana	21	18	13	11	12	17	17	19
Kansas	0	0	0	0	0	0	0	0
Kentucky	71	61	41	42	57	62	58	66
Louisiana	2	2	1	-	1	1	1	1
Maryland	5	4	3	3	4	5	2	2
Mississippi	-	2	2	1	2	1	1	1
Missouri	0	0	0	0	1	0	0	0
Montana	3	3	3	3	3	3	3	3
New Mexico	3	3	3	3	3	2	2	2
North Dakota	3	3	3	2	3	2	2	2
Ohio	17	13	9	10	11	10	11	14

Oklahoma	1	1	1	1	1	1	1	1
Pennsylvania (Anthracite)	0	0	0	1	1	0	0	-
Pennsylvania (Bituminous)	28	32	29	38	38	42	48	47
Tennessee	1	1	1	1	1	1	1	1
Texas	21	22	20	19	20	17	17	16
Utah	1	1	1	1	1	1	1	1
Virginia	12	10	9	8	10	10	8	7
Washington	-	-	-	-	-	-	-	-
West Virginia	92	79	57	52	56	68	76	69
Wyoming	17	18	15	15	17	16	17	19
State Totals	**322**	**293**	**227**	**223**	**256**	**282**	**288**	**293**

Table C5: Unadjusted ("ZCF") quantity of 28 trace elements theoretically present in all coal ash ("adjusted" volumes) produced by combustion of coal by electrical generation activities in the United States, captured, and placed into disposal annually, 2001-2008.

Trace Element	**"ZCF" Trace Element Content for All Coal Ash Produced by Electrical Generation Activities and Placed into Disposal, 2001-2008**								
	2001	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**Total**
As	322	293	227	223	256	282	288	293	**2,183**
B	2,247	2,296	1,945	1,858	2,108	2,014	2,082	2,155	**16,705**
Ba	6,584	6,771	5,680	5,569	6,289	5,998	6,235	6,519	**49,645**
Be	65	60	48	47	54	57	58	59	**448**
Cd	13	13	10	10	11	12	12	12	**93**
Co	182	171	136	132	151	160	164	167	**1,261**
Cr	504	473	378	369	422	445	455	462	**3,508**
Cu	628	599	483	468	535	555	569	580	**4,417**
F	2,801	2,726	2,229	2,160	2,459	2,485	2,555	2,618	**20,034**
Ga	230	217	173	169	194	204	208	212	**1,608**
Hg	5	5	4	4	4	4	4	4	**34**
Li	586	549	436	429	491	520	532	540	**4,083**
Mn	5,179	4,708	3,621	3,618	4,150	4,576	4,670	4,776	**35,300**
Mo	73	69	54	54	61	65	66	68	**510**

Nb	136	133	108	107	122	122	126	130	**984**
Ni	498	462	366	354	406	436	445	451	**3,416**
Pb	390	364	284	279	319	342	352	361	**2,690**
Sb	30	28	22	22	25	26	27	28	**208**
Sc	127	122	100	96	110	112	115	117	**899**
Se	126	119	97	93	106	111	113	115	**880**
Sr	3,953	3,936	3,274	3,173	3,605	3,550	3,655	3,765	**28,911**
Th	122	119	97	95	108	109	113	116	**879**
U	52	51	42	41	46	46	48	49	**376**
V	806	771	626	603	690	712	728	742	**5,678**
Y	326	317	261	250	286	289	295	301	**2,325**
Yb	30	29	24	23	26	27	27	28	**214**
Zn	780	748	607	571	656	678	697	714	**5,451**
Zr	1,203	1,162	951	921	1,051	1,071	1,095	1,118	**8,573**

Table C6: Unadjusted (“ZCF”) quantity of 28 trace elements theoretically present per coal ash type (“adjusted” volumes) produced by combustion of coal by electrical generation activities in the United States, captured, and placed into either landfills or surface impoundments for disposal in 2008.

Trace Element	**“ZCF” Trace Element Content per Coal Ash Type Produced and Placed in Landfills in 2008 (in short tons)**			**“ZCF” Trace Element Content per Coal Ash Type Produced and Placed in Surface Impoundments in 2008 (in short tons)**		
	Fly Ash	**Bottom Ash**	**Boiler Slag**	**Fly Ash**	**Bottom Ash**	**Boiler Slag**
As	161	42	1	73	16	1
B	1,181	306	4	534	121	10
Ba	3,572	924	13	1,615	366	30
Be	32	8	<0.5	15	3	<0.5
Cd	7	2	<0.5	3	1	<0.5
Co	91	24	<0.5	41	9	1
Cr	253	65	1	114	26	2
Cu	318	82	1	144	33	3
F	1,434	371	5	648	147	12
Ga	116	30	<0.5	53	12	1
Hg	2	1	<0.5	1	0	<0.5
Li	296	77	1	134	30	2
Mn	2,617	677	10	1,183	268	22

Mo	37	10	<0.5	17	4	<0.5
Nb	71	18	<0.5	32	7	1
Ni	247	64	1	112	25	2
Pb	198	51	1	89	20	
Sb	15	4	<0.5	7	2	<0.5
Sc	64	17	<0.5	29	7	1
Se	63	16	<0.5	28	6	1
Sr	2,062	534	8	933	211	17
Th	63	16	<0.5	29	6	1
U	27	7	<0.5	12	3	<0.5
V	406	105	2	184	42	3
Y	165	43	1	75	17	1
Yb	15	4	<0.5	7	2	<0.5
Zn	391	101	1	177	40	3
Zr	612	158	2	277	63	5

Table C7: Unadjusted (“ZCF”) quantity of 28 trace elements theoretically present in all coal ash types (“adjusted” volumes) produced by combustion of coal by electrical generation activities in the United States, captured, and placed into either landfills or surface impoundments for disposal annually, 2001-2008.

Trace Element	**“ZCF” Trace Element Content per Coal Ash Type Produced and Placed in Landfills, 2001-2008 (in short tons)**			**“ZCF” Trace Element Content per Coal Ash Type Produced and Placed in Surface Impoundments, 2001-2008 (in short tons)**		
	Fly Ash	**Bottom Ash**	**Boiler Slag**	**Fly Ash**	**Bottom Ash**	**Boiler Slag**
As	1,193	314	4	539	124	9
B	9,136	2,391	32	4,131	946	70
Ba	27,151	7,106	94	12,276	2,811	207
Be	245	64	1	111	25	2
Cd	51	13	<0.5	23	5	<0.5
Co	689	181	2	312	72	5
Cr	1,917	504	7	867	199	15
Cu	2,414	634	9	1,091	251	19
F	10,951	2,873	38	4,951	1,136	85
Ga	879	231	3	397	91	7
Hg	19	5	<0.5	8	2	<0.5
Li	2,231	586	8	1,009	232	17
Mn	19,280	5,074	69	8,717	2,007	153

Mo	279	73	1	126	29	2
Nb	538	141	2	243	56	4
Ni	1,866	491	7	844	194	15
Pb	1,469	386	5	664	153	12
Sb	113	30	<0.5	51	12	1
Sc	492	129	2	222	51	4
Se	481	126	2	217	50	4
Sr	15,807	4,142	55	7,147	1,639	121
Th	480	126	2	217	50	4
U	205	54	1	93	21	2
V	3,103	815	11	1,403	322	24
Y	1,271	333	4	575	132	10
Yb	117	31	<0.5	53	12	1
Zn	2,979	782	11	1,347	309	23
Zr	4,685	1,229	16	2,118	486	36

Table C8: Comparison of ppm values for ten trace elements present in whole coal from Swanson *et al.* (1976) to those measured in coal combustion residues, as cited in EPA's June proposed disposal rule.

Trace Element	**(Geometric) Mean ***		**Minimum**		**Maximum**	
	EPA [1]	**Swanson [2]**	**EPA [1]**	**Swanson [2]**	**EPA [1]**	**Swanson [2]**
As	24.7	1-12	0.00394	0.5- <1	773	16-357
Ba	246.75	30-300	0.002	3-15	7,230	300-3,000
Be	2.8	0.3-2	0.025	0.07-0.3	31	1.5-7
Cd	1.05	0.12-0.3	0.000115	0.02- <0.11	760.25	0.5-100
Cr	27.8	3-20	0.005	<0.5 -5	5,970	30-70
Pb	25	2.8-10.9	0.0074	<0.7- <2.8	1,453	19.4-283
Hg	0.18	0.04-0.14	0.000035	<0.01- 0.03	384.2	0.49-3.3
Ni	32	2-18	0.0025	0.7-3	54,055	20- ≥200
Sb	6.32	0.3-0.8	0.00125	<0.1-0.2	3,100	2.5-34.6
Se	2.4075	0.5-5.8	0.0002	<0.1- 1.8	673	5.7-150

Notes:

* Swanson *et al.* (1976) specifies values for trace element levels in whole coal samples as being "geometric means," whereas EPA data in *Federal Register* (2010) specifies trace element levels in coal combustion residues as "mean" values only.

Sources:

[1] Information cited in *Federal Register* (2010), p. 35169, summarizing from the CCR database; this in turn may be found at Document ID: EPA-HQ-RCRA-2009-0640-0028 in the public-comment docket for EPA's June 2010 proposed rule, which can be searched at <http://www.regulations.gov/>.

[2] Swanson *et al.* (1976)

Appendix D: Radioactivity Level and Disposal Cost Comparisons

Table D1: Summary of the four major subclasses of low-level radioactive waste (LLRW) and their characteristics, as recognized by NRC.

LLW Waste Class	**Definition**
Class A	Low levels of radiation and heat. No shielding required to protect workers or public. Rule of thumb states that it should decay to acceptable levels within 100 years.
Class B	Has higher concentrations of radioactivity than Class A and requires greater isolation and packaging (and shielding for operations) than Class A waste. Nominally safe after 300 years.
Class C	Requires isolation from the biosphere for 500 years. Must be buried at least 5 meters below the surface and must have an engineered barrier (container and grouting).
Greater than Class C	Does not qualify for near-surface burial. Contains alpha-emitting transuranic nuclides (TRUs) that have half-lives greater than 5 years and activity greater than 100 nCi/g.

Source:

10 CFR 61.7, as summarized by Lowenthal (1997, p. 9).

Table D2: Total volume and activity levels of low-level radioactive waste (LLRW) and associated radioactive materials received for storage at Barnwell, Clive, and Richland disposal facilities, 2008.

Data Reported in MIMS Database [1]			
	Total activity of all low-level radioactive wastes received for storage at Barnwell, Clive, and Richland disposal facilities in 2008 (Ci):	782,881	
	Total volume of all low-level radioactive wastes received for storage at Barnwell, Clive, and Richland disposal facilities in 2008 (cu. ft.):	2,085,305	
	Volume of Class A waste (cu. ft.):	2,069,776	(99.3%)
	Volume of Class B waste (cu. ft.):	5,963	(0.3%)
	Volume of Class C waste (cu. ft.):	9,566	(0.5%)
Data Not Reported in MIMS Database [2]			
Total volume of DOE waste received for storage at Clive disposal facility in 2008 (cu. ft.):		1,571,983	
	Low-level waste [“LLW”] (cu. ft.):	1,488,193	(94.7%)
	Mixed low-level waste [“MLLW”] (cu. ft.):	83,790	(5.3%)
	Byproduct material [“11e.(2)”] (cu. ft.):	-	-
	Naturally-occurring radioactive material [“NORM”] (cu. ft.):	-	-
Total volume of non-DOE waste received for storage at Clive disposal facility in 2008 (cu. ft.):		658,410	
	Low-level waste [“LLW”] (cu. ft.):	-	-
	Mixed low-level waste [“MLLW”] (cu. ft.):	124,659	(18.9%)
	Byproduct material [“11e.(2)”] (cu. ft.):	505,697	(76.8%)
	Naturally-occurring radioactive material [“NORM”] (cu. ft.):	28,054	(4.3%)

Sources:

[1] DOE (2010a).

[2] DOE (2010b).

Table D3: Total volume and activity levels of MIMS-reported Class A LLRW produced in the United States and placed into commercial storage annually, for both all waste generator categories and nuclear utilities only, 2001-2008.

	All Class A LLRW Generator Categories		**Class A LLRW Produced by Nuclear Utilities Only**	
Year	**Activity Levels (Ci)**	**Volume (cu. ft.)**	**Activity Levels (Ci)**	**Volume (cu. ft.)**
2001	6,981	3,383,683	304,245	6,200
2002	6,728	2,618,198	457,606	5,847
2003	4,847	2,794,296	609,506	4,010
2004	7,391	3,832,474	1,956,123	4,125
2005	7,546	3,995,598	2,399,901	4,622
2006	8,901	4,026,355	1,837,427	4,287
2007	7,366	2,600,760	1,076,998	3,409
2008	6,979	2,069,776	619,635	2,255
Totals	**56,739**	**25,321,140**	**9,261,442**	**34,755**

Source:

DOE (2010a)

Table D4: Total radioactivity of all coal ash (“adjusted” volumes) produced by combustion of whole coal for electrical generation activities in the United States, captured, and placed into either landfills or surface impoundments for disposal annually, 2001-2008.

	Activity Levels of All Coal Ash Produced by Electrical Generation Activities and Placed into Landfills Annually (Ci)			**Activity Levels of All Coal Ash Produced by Electrical Generation Activities and Placed into Surface Impoundments Annually (Ci)**		
Year	**Low**	**Average Range**	**High**	**Low**	**Average Range**	**High**
2001	60	165-173	290	27	74-78	130
2002	60	167-174	291	27	75-78	130
2003	49	137-142	238	22	61-63	106
2004	48	134-139	232	21	60-62	104
2005	54	151-157	263	24	67-70	117
2006	54	150-156	262	24	67-70	117
2007	55	153-160	268	25	69-72	120
2008	57	159-166	278	26	71-74	124
Totals	**438**	**1,216-1,268**	**2,121**	**196**	**544-566**	**947**

Notes:

All activity levels listed here were calculated using low, average, and high activity level values from EPA (2009c). As described earlier, these values were held constant for all years of coal ash production, and boiler slag values were assumed to be identical to activity levels of bottom ash. The initial activity level values from EPA (in pCi/g) are as follows:

Fly ash: 2.00 x 10^{-12} (low), 5.80 x 10^{-12} (average), 9.70 x 10^{-12} (high)
Bottom ash: 1.60 x 10^{-12} (low), 3.50-4.60 x 10^{-12} (average), 7.70 x 10^{-12} (high)
Boiler slag: 1.60 x 10^{-12} (low), 3.50-4.60 x 10^{-12} (average), 7.70 x 10^{-12} (high)

Table D5: Comparison of radioactivity per kilowatt of electricity produced by nuclear and coal sources, and per cubic foot of waste produced and placed into disposal in 2008, as calculated from activity levels of nuclear LLRW placed into commercial storage and coal ash (“adjusted” disposal volumes).

	Nuclear [1, 2]		**Coal** [1, 3]		
Electricity generated in 2008 (kWh)	806,208,000,000		1,985,801,000,000		
Activity levels of low-level radioactive wastes produced and placed into disposal in 2008 (Ci)	**Class A LLRW, All Sources** *	**Class A LLRW, Utility-Sourced Only**		**Fly Ash Only**	**All Coal Ash**
	6,979	2,255	*Low estimate:* *Average estimate:* *High estimate:*	69 200 334	83 230-240 402
Activity levels of LLRW produced per kWh of electricity generated in 2008 (Ci/kWh)	8.66×10^{-9}	2.80×10^{-9}	*Low estimate:* *Average estimate:* *High estimate:*	3.47×10^{-11} 1.01×10^{-10} 1.68×10^{-10}	4.17×10^{-11} $1.16\text{-}1.21 \times 10^{-10}$ 2.02×10^{-10}
Activity levels of LLRW produced per ft^3 of waste disposed in 2008 (Ci/ft^3) **	0.124	3.64×10^{-3}	*Low estimate:* *Average estimate:* *High estimate:*	1.64×10^{-7} $4.77\text{-}6.68 \times 10^{-7}$ 7.98×10^{-7}	1.10×10^{-7} $3.05\text{-}4.37 \times 10^{-7}$ 5.32×10^{-7}

Notes:

* The MIMS database records various subcategories of Class A waste placed into commercial (*i.e.,* off-site) storage facilities in the U.S. annually; these classifications are, respectively, for recording waste produced by academic, government, industry, medical, utility, and "undefined" sources. While the "Utility" category for LLRW produced at nuclear power plants is clearly of greatest interest here for purposes of direct comparison to coal ash produced by electrical generation activities, I have here also included the Class A totals across all subcategories for reference purposes.

Please note, however, that as documented in Table D2 above, the MIMS information presented here may not necessarily be representative of the total amount of Class A LLRW produced and placed into commercial storage annually – and may, in fact, be quite significant underestimates. Without more detailed information regarding this excluded waste stream, however, I do not know what proportion of the non-reported "LLW" waste is indeed Class A, nor what amount of Class A waste was produced by nuclear power plants alone within the given timeframe. Equally, activity levels are unknown for all portions of these excluded waste streams.

As such, I would recommend that while the calculations presented both here and in subsequent tables are useful for very rough initial comparisons, they should be treated cautiously, as "placeholder" values only until more reliable information becomes available on this subject.

** For further discussion on how cubic-foot volumes of coal ash were calculated, see the notes to Table D6 below.

Sources:

[1] EIA (2010k)
[2] DOE (2010a)
[3] EPA (2009c)

Table D6: Total volume (in cubic feet) of all coal ash types ("adjusted" volumes) produced by combustion of coal by electrical generation activities in the United States, captured, and placed into either landfills or surface impoundments for disposal in 2008.

	Coal Ash Type Produced and Disposed in Landfills in 2008			**Coal Ash Type Produced and Disposed in Surface Impoundments in 2008**		
	Fly Ash	**Bottom Ash**	**Boiler Slag**	**Fly Ash**	**Bottom Ash**	**Boiler Slag**
Ash produced and placed into disposal (short tons)	26,137,594	6,709,935	96,564	11,817,796	2,654,594	212,706
Range for specific gravity of ash type [1,2]	2-2.8	2.1-2.7	2.3-2.9	2-2.8	2.1-2.7	2.3-2.9
Cubic feet of ash type placed into disposal (minimum estimate) *	299,108,071	79,629,705	1,066,937	135,238,088	31,503,214	2,350,188
Cubic feet of ash type placed into disposal (maximum estimate) *	418,751,300	102,381,050	1,345,268	189,333,323	40,504,132	2,963,281

Notes:

* As noted earlier, due to lack of information for specific gravity of boiler slag, I have for now performed calculations assuming that boiler slag possesses identical specific gravity values as bottom ash.

I have here determined the volume of coal ash types placed into disposal by dividing short tons by density of the ash type, with this latter variable calculated by multiplying the specific gravity of the ash type by the density of water (999.8395 kg/m^3 at STP).

Sources:

[1] DOT (2002)
[2] Majizadeh *et al.* (1979)

Table D7: Total radioactivity per cubic foot of all coal ash types (“adjusted” volumes) produced by combustion of coal by electrical generation activities in the United States, captured, and placed into either landfills or surface impoundments for disposal in 2008.

	Activity Levels per Cubic Foot for All Coal Ash Produced and Disposed, 2001-2008			**Activity Levels per Cubic Foot for All Fly Ash Produced and Disposed, 2001-2008**		
Year	**Low**	**Average**	**High**	**Low**	**Average**	**High**
2001	1.50 x 10^{-7}	3.02-4.34 x 10^{-7}	5.29 x 10^{-7}	2.30 x 10^{-7}	4.77-6.68 x 10^{-7}	1.12 x 10^{-6}
2002	1.51 x 10^{-7}	3.05-4.38 x 10^{-7}	5.32 x 10^{-7}	2.30 x 10^{-7}	4.77-6.68 x 10^{-7}	1.12 x 10^{-6}
2003	1.52 x 10^{-7}	3.07-4.39 x 10^{-7}	5.33 x 10^{-7}	2.30 x 10^{-7}	4.77-6.68 x 10^{-7}	1.12 x 10^{-6}
2004	1.52 x 10^{-7}	3.08-4.40 x 10^{-7}	5.34 x 10^{-7}	2.30 x 10^{-7}	4.77-6.68 x 10^{-7}	1.12 x 10^{-6}
2005	1.51 x 10^{-7}	3.06-4.38 x 10^{-7}	5.32 x 10^{-7}	2.30 x 10^{-7}	4.77-6.68 x 10^{-7}	1.12 x 10^{-6}
2006	1.51 x 10^{-7}	3.04-4.37 x 10^{-7}	5.31 x 10^{-7}	2.30 x 10^{-7}	4.77-6.68 x 10^{-7}	1.12 x 10^{-6}
2007	1.50 x 10^{-7}	3.03-4.36 x 10^{-7}	5.30 x 10^{-7}	2.30 x 10^{-7}	4.77-6.68 x 10^{-7}	1.12 x 10^{-6}
2008	1.51 x 10^{-7}	3.05-4.37 x 10^{-7}	5.32 x 10^{-7}	2.30 x 10^{-7}	4.77-6.68 x 10^{-7}	1.12 x 10^{-6}
Totals	**1.21 x 10^{-6}**	**2.44-3.50 x 10^{-6}**	**4.25 x 10^{-6}**	**1.84 x 10^{-6}**	**3.82-5.34 x 10^{-6}**	**8.93 x 10^{-6}**

Table D8: Estimated costs (in $USD, 2010) of commercial LLRW storage for all coal ash (“adjusted” volumes) produced by combustion of coal by electrical generation activities in the United States, captured, and placed into either landfills or surface impoundments for disposal, 2001-2008.

	Storage Cost of Disposed Coal Ash (Minimum Estimate)		**Storage Cost of Disposed Coal Ash (Maximum Estimate)**	
Year	**All Disposal Formats**	**Surface Impoundment Only**	**All Disposal Formats**	**Surface Impoundment Only**
2001	$231,356,434,116	$71,593,065,773	$317,548,542,487	$98,278,298,641
2002	$230,059,919,448	$70,840,329,004	$316,606,113,631	$97,548,888,007
2003	$186,986,288,686	$57,318,066,283	$257,619,591,406	$79,050,160,748
2004	$182,660,255,251	$56,249,102,960	$251,847,958,866	$77,605,770,077
2005	$207,332,597,121	$63,495,605,746	$285,489,318,703	$87,525,364,461
2006	$207,256,265,202	$63,824,102,159	$285,008,530,058	$87,818,468,624
2007	$213,103,527,822	$65,678,784,029	$292,803,968,099	$90,286,521,686
2008	$219,558,481,546	$67,636,596,165	$302,111,341,689	$93,120,294,618
Total	**$1,678,313,769,192**	**$516,635,652,117**	**$2,309,035,364,940**	**$711,233,766,862**

Notes:

* Commercial LLRW storage costs have been calculated utilizing the $400/ft^3 estimate provided by GAO (2004), here for simplicity assumed to function as a flat rate, applicable to all ash types and storage formats – *i.e.*, no ash recovery, handling, or transportation costs are included here.

Glossary

ACAA	American Coal Ash Association
ASTSWMO	Association of State and Territorial Solid Waste Management Officials
Al	Aluminum
As	Arsenic
B	Boron
Ba	Barium
Be	Beryllium
Br	Bromine
Ca	Calcium
CCP	Coal combustion products *(Along with "CCR," an equivalent term for "PCC." See Chapter 1, Endnote #2, p. 92 for further explanation.)*
CCR	Coal combustion residue *(See above glossary entry for "CCP.")*
Cd	Cadmium
Ce	Cerium
CERCLA	1980 Comprehensive Environmental Response, Compensation, and Liability Act
CFR	U.S. Code of Federal Regulations
Ci	Curies
Cl	Chlorine
Co	Cobalt
Cr	Chromium
Cu	Copper

DOE	U.S. Department of Energy
DOT	U.S. Department of Transportation
EIA	U.S. Energy Information Administration
EPA	U.S. Environmental Protection Agency
Eu	Europium
F	Fluoride
FBC	Fluidized bed combustion
FGD	Flue gas desulfurization
Ga	Gallium
GAO	U.S. Government Accountability Office
Hf	Hafnium
Hg	Mercury
HLRW	High-level radioactive waste
HQ	Hazard quotient
IEA	International Energy Agency
K	Potassium
kW(h)	Kilowatt(-hours)
La	Lanthanum
Li	Lithium
LL(R)W	Low-level (radioactive) waste
MCL	Maximum contaminant level
Mg	Magnesium
MIMS	Manifest Information Management System
MLLW	Mixed low-level waste
Mn	Manganese

Mo	Molybdenum
MW(h)	Megawatt(-hours)
Na	Sodium
Nb	Niobium
nCi	Nanocuries
Ni	Nickel
NID	National Inventory of Dams
NORM	Naturally-occurring radioactive material
NRC	U.S. Nuclear Regulatory Commission
Pb	Lead
PCC	Products of coal combustion *(A personal term that is equivalent to "CCP" and "CCR" as used by ACAA/EIA and EPA. See Introduction, Endnote #1, p. 91 and Chapter 1, Endnote #2, p. 92 for further explanation.)*
ppm	Parts per million
Ra	Radium
Rb	Rubidium
RCRA	1976 Resource Conservation and Recovery Act
Sb	Antimony
Sc	Scandium
Se	Selenium
Si	Silicon
Sm	Samarium
Sr	Strontium
Ta	Tantalum

TDEC	Tennessee Department of Environment and Conservation
Th	Thorium
Ti	Titanium
Tl	Thallium
TRU	Transuranic waste
TVA	Tennessee Valley Authority
U	Uranium
U.S.	United States of America
USGS	U.S. Geological Survey
V	Vanadium
Y	Ytterbium
ZCF	Zero Concentration Factor (*See Chapter 3, p. 55 for definition.*)
Zn	Zinc
Zr	Zirconium

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Vita

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