

CHAPTER 1: OVERVIEW: COAL COMBUSTION WASTE AND MINEFILLING IN PENNSYLVANIA

1.1 Introduction

To understand the behavior of CCW in mines, one must consider many factors, including the behavior of CCW in other disposal environments, its chemical composition, and CCW's propensity to change over time. To understand the motivation to place CCW in mines, one must examine the relationship between abandoned mine lands (AML), acid mine drainage (AMD), waste coal plants, and the economics of waste disposal in Pennsylvania. This chapter first reviews numerous studies describing how, why, and where CCW has caused adverse environmental impacts. Next the processes by which coal ash is produced and the chemical makeup of various types of ash are described. Lastly, this chapter briefly examines the complex relationship between coal ash placement in mines, AML, AMD and waste coal plants in Pennsylvania.

1.2 Adverse environmental impacts from coal combustion waste: review of the literature

The burning of coal produces large amounts of fly ash, bottom ash, boiler slag and flue gas desulfurization sludge that are collectively called coal combustion wastes or CCW. Today CCW is the second largest industrial waste stream in America, surpassed only by mining waste. As efforts to control emissions from coal combustion increase, so have the volumes of CCW. For example, the volume of CCW produced nationally increased by 30 - 40% to approximately 130 million tons annually in 2004 largely due to requirements in the Clean Air Act Amendments of 1990 to control acid rain.¹ These amendments resulted in the use of emission control devices known as scrubbers that are now generating approximately 26 million tons of flue gas desulfurization sludge (scrubber sludge) annually. The recently promulgated Clean Air Interstate Rule (CAIR) and the proposed rule to control mercury are likely to increase total CCW generation further with estimates of as much as 170 million tons being generated annually by 2015.² The disposal of CCW has caused a variety of environmental problems particularly to soils and waters, due to an extreme pH and high concentrations of soluble salts, trace metals and other pollutants that leach from different CCWs.

Coal and CCW have been analyzed and characterized by a number of researchers. The composition of coal and coal combustion wastes varies widely. According to Block and

¹ Degeare, Truett, U.S. Environmental Protection Agency, Office of Solid Waste. Overview of U.S. Environmental Protection Agency Coal Combustion Waste (CCW) Mine Fill Issues. Undated. <http://www.mcrcc.osmre.gov/PDF/Forums/CCB3/5-1.pdf>

² U.S. EPA, Clean Air Markets Division. Memorandum to the Docket entitled Economic and Energy Analysis for the Proposed Interstate Air Quality Rulemaking. January 28, 2004

Dams (1976), the composition of fly ash is “significantly different from the original coal composition.” For example, in comparison to coal, fly ash is relatively enriched in elements such as chlorine, copper, zinc, arsenic, selenium, and mercury (Block and Dams, 1976). According to Carlson and Adriano (1993), fly ash is also enriched in boron, strontium, molybdenum, sulfur, and calcium. Trace elements in the ash are concentrated in the smaller ash particle sizes.

In a pilot study by the Electric Power Research Institute (1983), chemical analysis of coal, bottom ash, and fly ash from a southwest U.S. power plant burning southwestern subbituminous coal revealed large quantitative differences in elemental concentrations of the three materials:

	Aluminum (ppm)	Arsenic (ppb)	Barium (ppm)	Chromium (ppm)	Iron (ppm)	Magnesium (ppm)	Lead (ppb)	Silicon (ppm)
Coal	29100	7100	71.8	6.1	5130	1130	11000	50200
Bottom Ash	142000	24100	2830	29	28100	4640	23000	260000
Fly Ash	144000	32750	3110	31	24800	5260	51500	258000

A quantitative evaluation of the mobilization of trace metals from coal-burning power stations in Europe, including stack emissions and the quantities retained by the electrostatic filters, and thus present in the ash residue, was completed by G. Bignoli (Bignoli, 1989). Bignoli notes that modern electrostatic filters can filter out 99.8% of particulate matter and thus the environmental impact of most of the metals from coal-burning power plants will be due mostly to the potential releases from solid wastes. (Sabbioni, Goetz and Bignoli, 1984) Analyses of trace metals present in the coal revealed that of the total mobilization of these metals in the combustion process the great majority of the metals were retained in the solid waste and only a very small percent were present in the atmospheric stack emissions. For example, for trace metals, arsenic, cadmium, chromium, lead, antimony and selenium, 97%, 97.2%, 99%, 97.5%, 97.7% and 91.5% of the total mobilization of each of these metals, respectively, was retained and concentrated in the coal ash.

Vorre (1986) has described the nature of mineral matter in coal. This matter includes the minerals present in the original plants that were altered over time to produce coal. Geologic events such as subsidence and volcanic eruptions provided additional inorganic material. These biological and geologic processes contributed to the formation of coals each with their own signature suite of minerals.

The mineral and trace element content of CCW can vary substantially depending on the locations of parent coals within different coal basins, different coal seams within the same basin, and different locations within single coal seams. Using the Lower Kittanning Coal seam in western Pennsylvania, Rimmer and Davis (1986) analyzed the physical, chemical and biological processes that affected the mineral composition of coals. The Lower Kittanning Coal seam demonstrates lateral variations in mineral compositions that

were related to the depositional environments.³ Research by Lindahl and Finkelman (1986) and Harvey and Ruch (1986) support the variability model of the mineral content within and between coal seams and between regional basins. For example, the mean concentrations of lead, chromium, nickel, and arsenic are three to five times higher in Appalachian and Illinois Basin coals than in coals of the Rocky Mountains and Northern Plains.

Numerous researchers have documented adverse environmental impacts caused by CCW to groundwater⁴ and surface waters, plants, aquatic life, and other organisms. Carlson and Adriano (1980) maintain that the major environmental impacts of CCW include: leaching of potentially toxic substances into soils, groundwater and surface waters; hindering effects on plant communities; and the accumulation of toxic elements in the food chain. Adriano et al. (1980), Elseewi et al. (1980), Phung et al. (1979), and Menon et al. (1990) analyzed the chemical and physical composition of fly ash under various experimental conditions to determine the environmental impact of inorganic constituents at disposal sites, such as the release of trace elements in water and treated soils. Sandhu et al. (1993) specifically studied the leaching of nickel, cadmium, chromium, and arsenic from coal ash impoundments of different ages. The general conclusion indicated that leaching produces a measurable release of metals into the environment from both old and new ash deposits: “[A]sh deposits... weathered and leached for over 10 years, yet still may provide a source of metal contamination to infiltrating water. Thus, ash disposal basins may be potential sources of ground water contamination for many years after ash deposition has ceased” Sandhu et al. (1993).

Researchers such as Rowe et al. (2002) documented the negative effect of coal combustion waste on the physiology, morphology and behavior of aquatic organisms and the health of aquatic ecosystems. According to Rowe et al (2002), the “release of CCR [coal combustion residues such as fly ash] into aquatic systems has generally been associated with deleterious environmental effects. A large number of metals and trace elements are present in CCR, some of which are rapidly accumulated to high concentrations in aquatic organisms. Moreover, a variety of biological responses have been observed in organisms following exposure to and accumulation of CCR-related contaminants. In some vertebrates and invertebrates, CCR exposure has led to numerous histopathological, behavioral and physiological (reproductive, energetic and edocrinological) effects.”⁵

³ For example, high pyrite concentrations occurred in areas where the overlying shale indicated brackish, swamp-like, anoxic conditions. High quartz content in the northern part of the coal seam coincided with a source area where quartz was transported into the swamp by water and/or air from a topographic high area.

⁴ Groundwater is particularly important because half of the population of the United States relies on groundwater as its source of potable water either through public or domestic supplies (Solley et al., 1998).

⁵ Rowe et al. (2001) studied the adverse impact of trace metals from CCW on the standard metabolic rate of crayfish (*Procambarus acutus*). Other researchers such as Hopkins et al. (2000) studied the detrimental impact of trace elements on lake chubsuckers (*Erimyzon sucetta*). Fish exposed to lake sediments polluted with coal ash exhibited “substantial decreases in growth and severe fin erosion.”

Lemly (1999) found that selenium leaching from coal ash landfills posed a great danger to fish populations and documented the elimination of a diverse fish population at Belews Lake, North Carolina from such contamination. Rowe et al. (2000) studied the effects on southern toads living in an environment polluted by coal ash concluding that major reductions occurred in local populations because less food algae could survive in the polluted water and the toxicity of coal ash trace elements in the sediments and surviving food algae killed larval toads. The study suggests that the widespread practice of disposing of coal ash in open aquatic basins may result in sink habitats for some amphibian populations. Cherry (2000) and coworkers, after evaluating the level of toxicity at 32 CCW sites throughout the world, concluded that coal combustion wastes have adverse impacts on ecosystems. Namely, trace elements and other constituents such as sulfates, chlorides, sodium, boron, manganese, iron, selenium, arsenic, lead, chromium, nickel, copper and zinc leach from CCW ash particle surfaces at toxic levels into groundwater and surface water and threaten human and aquatic life.

Elseewi et al. (1980) maintain that solutions from fly ash are mostly alkaline, have a high salt content primarily due to the dissolution of Ca^{2+} and OH^- ions, and contain an elevated concentration of boron that may be toxic to plants. According to Adriano et al. (1980), "...coal ash usually is not suitable for agricultural uses due to the high cost of handling and transportation from the source, very low C [carbon] and N [nitrogen] contents, and usually high pH and toxic B [boron] contents. Thus, if lands are to be used for fly ash disposal purposes, application rates should be balanced between environmental impacts and economics of waste disposal. Massive applications are usually associated with adverse effects to soils and growing plants."

There has been very little research undertaken on the environmental impacts of organic constituents in CCW. Researchers have long known that coal fly ashes contain a number of polyaromatic hydrocarbons (PAHs), (Griest and Guerin, 1979) (Hanson et al, 1983, citing Sucre et al, 1979) (Bennett et al, 1979) (Hanson et al, 1979, 1980 & 1981). Examples of these PAHs include naphthalene, acenaphthylene, anthracene, dibenzofuran, fluorene, and fluoranthene. A number of the PAHs in CCW are toxic, mutagenic and/or carcinogenic in laboratory studies. Their bioaccumulation appears to be limited due to metabolism. The metabolism itself, however, may produce oxidation damage in tissues and breakdown products that are more mutagenic than their precursors. Researchers have documented that fly ashes from both pulverized coal combustion and fluidized bed combustion contain PAHs that readily cause bacteria to mutate, (Hanson et al., 1983, citing Chrisp et al, 1978, Fisher et al, 1979, Kubitschek and Haugen, 1980, Clark and Hobbs, 1980, Hill et al, 1981, and Wei et al, 1982). Hansen et al, 1983 documented that treatment of FBC fly ash with N_2O_4 increased its mutagenic potency by as much as 3200 times on a laboratory strain of salmonella bacteria. They concluded that potent, direct acting mutagens such as dinitropyrenes and dinitrofluoranthenes in fly ash from fluidized bed combustion power plants might be products of reactions between PAHs in the ash with nitrogen oxides in combustion gases. However, Harrison et al, 1986, concluded that the concentrations of PAHs detected in fly ash probably would not pose an environmental hazard, although they acknowledged difficulties in their ability to detect and measure

PAHs in the fly ash. Griest and Guerin, 1979, concluded similarly that PAHs adhere strongly to ash, making analysis of their quantities and types in CCW difficult.

In its Report to Congress on Wastes from the Combustion of Fossil Fuels, (March, 1999), the US EPA cited data collected by the Electric Power Research Institute in 1997 showing that coburning of petroleum coke wastes, coal gasification wastes, mixed plastics, tire-derived fuels and other organic wastes with coal generates CCW with detectable levels of benzene, chlorobenzene, cyanide, dioxins, furans, PCBs, chlorophenol, and polycyclic aromatic hydrocarbons. Despite documentation of the existence of harmful organics in CCW, monitoring for organics in groundwater and surface water surrounding CCW sites is extremely rare. Thus little is known about actual impacts to the environment from organic compounds in CCW.

Although the US EPA exempted CCW from hazardous waste regulatory classification in its May 2000 Regulatory Determination on Waste from the Combustion of Fossil Fuels, it documented concerns in the Determination about the potential adverse impacts from placement of CCW in coal mines.⁶ US EPA concluded that safeguards were likely needed to prevent adverse impacts on water quality. US EPA also found that more research and information addressing the impacts to water quality and the environment from minefilling is needed to determine the nature of these safeguards.

1.3 Pennsylvania coalfield geology

Coals are technically not classified as rocks but as *fossils* composed of compacted plant remains. The sedimentary rock types found with coals in the bituminous coalfields include *conglomerate*, *sandstone*, *siltstone*, *shale*, *claystone*, *limestone* and *coal*. Most of these rocks are composed of mineral grains locked together by some cement, usually calcite, quartz, or clay.

The bituminous coalfield in Pennsylvania covers most of the western half of the state, eastward to the Allegheny Front. This area is known as the Allegheny Plateau physiographic province and is subdivided into the Pittsburgh Low Plateau, Pittsburgh High Plateau, Allegheny Mountain, and Mountainous High Plateau provinces. Its bituminous coalfield extends into Ohio and West Virginia and represents the largest bituminous coal reserve in the United States. The coal-bearing sedimentary rocks in this coalfield are of Pennsylvanian and Permian ages, 330 to 290 million years ago (mya) and 290 to 250 mya, respectively.

In western Pennsylvania, almost all of these coals are included in the lower Allegheny Group of the Pennsylvanian time period. These include three major coal beds, the deepest being the Clarion, followed by the Kittanning, with the Freeport bed closest to the surface. Only one of the permits in western Pennsylvania analyzed in this report, the Hartley Strip, involved the mining of younger coal beds (from the Permian time period) not in this group. The coals in the lower Allegheny Group were formed in near-shore

⁶ US EPA, Regulatory Determination on Waste from the Combustion of Fossil Fuels. Final Rule. Federal Register, Volume 65, number 99, page 32213. May 22, 2000.

marine and/or brackish conditions and contain more sulfur than younger coals. Higher sulfur coals contribute more to the AMD problem. Younger coals including those higher in the Allegheny Group were generally deposited in more fresh water, terrestrial environments, resulting in lower sulfur contents.

The anthracite coalfield is found within the Ridge and Valley physiographic province of eastern Pennsylvania. This province contains most of the anthracite found in the United States. All of the numerous coal beds in the eastern Pennsylvania anthracite region have low sulfur contents due to post-depositional metamorphism (changes to rocks from heat and pressure) and generally produce lower levels of acidity than coal beds in western Pennsylvania's bituminous region.

The coal and other sedimentary layers from the Pennsylvanian Period in western Pennsylvania are generally flat-lying, except for some folds or “humps” (*anticlines*) towards the eastern margin of the bituminous coalfields. Further east, the coals and rock layers in the Anthracite field show steeply pitching geometries due to a more intense folding activity closer to the collision margin between the continental plates of North Africa and North America occurring about 250 mya.

Water movement through coal-bearing sedimentary rocks is affected by four major variables: (1) local rock structure; (2) ability of water to flow through different rock types; (3) topography; and (4) man-made activities. Due to higher permeabilities, certain rocks such as limestones and sandstones convey enough water to be considered aquifers. Due to high degrees of fracturing, coal veins may also be aquifers. Natural contacts between rock layers, joints, and faults are also pathways of enhanced groundwater flow. The structural tilt (a.k.a. “dip”) of coals and surrounding rock layers also strongly influences the rate of groundwater flow. Groundwater flow through shallow, unconfined earth above the sedimentary layers generally follows the topography.

Human activities such as coal mining can greatly affect the direction and rate of groundwater flow. Coal seams mined out from deep mining can become major man-made conduits for underground water movement, known as “mine pools,” that can provide pathways for groundwater exit from a site at different elevations and rates than occurred at original points of discharge. Other fractures from blasting and overburden removal operations can also become man-made conduits. Furthermore when the broken up overburden or spoil from strip mining is placed back in the working pit as mining progresses, it will transmit groundwater at a much greater rate than undisturbed rock due to the greater amount of void space in this broken material. Since many surface mines in Pennsylvania have been previously mined and mining activity can change the direction and flow of groundwater, a thorough study of water movement is needed at proposed coal mine ash placement sites to develop effective monitoring systems, as groundwater behavior and the often substantial changes in that behavior from previous and proposed mining will be site specific.

Understanding chemistry of the specific ash and the geochemistry of the coal mine sites in which the ash is to be placed is also an important step to predicting impacts to water

quality that will result from minefilling. The depositional environment in which the coal or coals were formed at these sites must be understood. In the case of previously mined sites, chemical analysis not only of overburden but also of coal mine spoils and coal refuse should be undertaken when assessing the potential for acidity and other water quality impacts. Care must also be taken to calculate the net alkalinity available in coal ash that will be exposed indefinitely to this acidity and to design placement of ash that will maximize the buffering benefits of that alkalinity. Additional knowledge of the different leaching potentials for major constituents, minor metals and trace elements in the ash and surrounding mine materials when major factors such as pH and the redox environment are changing is also important to predicting results of importing large volumes of ash into coal mines. All of these geochemical variables will be very site specific and ash specific and should be addressed on a site-to-site basis.

1.4 CCW used in Pennsylvania coal mines

1.4.1. Types of CCW: Conventional Pulverized Coal Ash, Fluidized Bed Combustion Ash and Waste Coal CCW

CCW is a waste product resulting from the combustion of coal to generate electricity. Coal is composed mostly of carbon, volatiles (oxygen and hydrogen) and non-combustible materials including clay (aluminum silicates), silica (SiO₂), pyrite and marcasite (FeS₂), and other metallic oxides. After the coal is burned, these non-combustible components are discarded as ash or CCW.

There are two types of coal ash that are the focus of this report: 1) Conventional power plant ash from a pulverized coal (PC) plant and 2) Fluidized Bed Combustion (FBC) power plant ash. Both of these coal ashes are further divided into bottom ash and fly ash. PC bottom ash and fly ash are generated from burning primarily mine run coal, and FBC bottom and fly ashes are primarily generated by burning mixtures of coal and waste coal. There are two different types of waste coal. “Coal refuse” or “gob” is waste coal originating from bituminous coal. Waste coal from anthracite coal is called “culm.” Waste coal consists of the impurities cleaned from mined coal to prepare it for burning.

Fly ash and bottom ash are easily distinguished. Bottom ash particles are larger and heavier than fly ash particles and do not become airborne as a result of the combustion process. Thus bottom ash is generated on the grate of the boiler. Fly ash is very fine grained and becomes airborne during combustion. In PC plants, fly ash is captured by pollution control devices such as electrostatic precipitators and bag houses installed inside and adjacent to the stacks, and bottom ash is taken out of the furnace using a conveyor type of grate. Another major type of CCW, flue gas desulfurization sludge, also known as “scrubber sludge,” is generated by spraying lime or other highly alkaline material in a liquid or powdered state across flue gases to remove sulfur dioxide and other pollutants. Pennsylvania’s beneficial use regulations exclude scrubber sludge from the definition of “coal ash,” (See 25 PA Code Section 287.1) and therefore do not permit its placement in coal mines under the state’s beneficial use program.

The FBC ash studied in this report typically results from burning coal piles, which are comprised of waste coal and broken overburden rock (also called spoils) left in abandoned mine areas. These piles contain less than 50% coal, the remainder is composed of waste coal and spoils. In the bituminous coal field of western Pennsylvania, some previously unexcavated coal is often also mined along with the removal of the waste coal piles. The mined mixture is crushed and then injected into the furnace combustion chamber and kept airborne by heated air insuring complete burning of most of the coal. The waste material mixed with the coal is also burned if there is any “coaly” material in this waste rock. During the combustion process, limestone (CaCO_3) is introduced into the combustion chamber and burns to form CaO (lime). The addition of lime to the FBC ash increases its alkalinity, thus improving its potential for use as AMD remediation material. Larger incombustible particles are eventually removed from the furnace as FBC bottom ash, while lighter fly ash is captured in bag filters. Because the average ash content of waste coals is two to three times higher than parent coals and because limestone is injected into the combustion process, FBC power plants produce several times more CCW per megawatt of power than PC power plants. While FBC plants produce only 8% of the total megawatts generated by all Pennsylvania coal-fired plants, FBC ash constitutes over 60% of the CCW produced by all plants.⁷

1.4.2. Chemical Composition of CCW

An accounting of all the chemical phases of coal before and after the coal is burned can provide a better understanding of the final chemical makeup of the CCW that is being placed in Pennsylvania coal mines. The major elements in coal are carbon, hydrogen and oxygen. The hydrogen and oxygen are volatilized and escape the boiler furnace as gasses when the coal is burned. The carbon is also burned or oxidized, with the resulting heat of combustion that boils water into steam and drives electrical generators. During combustion, the carbon reacts with the oxygen in the coal and oxygen from the atmosphere to produce carbon monoxide (CO) and carbon dioxide (CO_2), both of which escape as gasses. Trace amounts of carbon are left behind in fly ashes as polyaromatic hydrocarbons and other organic compounds. These compounds usually form as products of incomplete combustion when emissions cool in the flue gasses.

The minor components in coal, sulfur, iron, silica, and clay are also heated, but all is not burned or oxidized. The sulfur is burned to form sulfur dioxide (SO_2), which, depending on the type of combustion, can escape as gas, be collected to produce gypsum, or be left in various oxidation states in the ash. The trace metals (many of which occur in pyrite) are separated from the sulfur and occur in the resulting ash as elemental metals and oxides. The iron is usually oxidized to form iron oxide, FeO . The shaley, or boney material in coal, particularly in FBC processes, contains silica (quartz or SiO_2) and clays, which are complex hydrated aluminum silicates. The water in the clays is driven off by

⁷ According to DOE EIA data for 2002, waste coal burning plants in Pennsylvania total 1,559.5 MW in nameplate capacity while all other coal-fired plants in Pennsylvania total 18,920.2 MW in nameplate capacity. EIA 2002 data was not available for Pennsylvania’s newest waste coal facility in Seward, PA. 521 MW was used as the nameplate capacity for that plant.

the heat of combustion, leaving aluminum oxides, and the silica does not react, initially. The small quartz grains fuse together in a process somewhat similar to the glassmaking process.

During combustion in a FBC plant, limestone is added in the furnace and is burned to lime, as outlined above. This lime reacts to form gypsum, thus removing most of the sulfur dioxide from the system. Some of the lime reacts with the silica to form calcium silicates. The excess lime is incorporated into the ash, increasing alkalinity. This lime, after the ash has been placed in a mine, *can* react with groundwater to form a cemented ash bound together by portlandite, $\text{Ca}(\text{OH})_2$. Portlandite forms by hydrating lime: $\text{CaO} + \text{H}_2\text{O} = \text{Ca}(\text{OH})_2$.

The greater the amount of limestone added to the furnace in a FBC plant, the higher the alkalinity of its ash, with most FBC ashes having significantly higher alkalinity than the alkalinity in conventional PC ashes. FBC ash also contains aluminum silicates, along with fused silica. Some sulfur is incorporated into the ash as anhydrite, CaSO_4 . Its metals can be incorporated, or “locked up” in some of the silicate minerals, or they can exist as oxides and sometimes as elemental metals.

Whether burned in PC or FBC plants, most metals in coal, including trace metals, are not volatilized during the burning process and remain in the ash. Their concentrations therefore increase as the coal volume reduces to ash volume, with a resulting higher metal component in coal ash than in the original coal. For example, data from the Electric Power Research Institute (1983) shows that aluminum, arsenic, chromium, magnesium, lead, and silicon have concentrations approximately five times greater in coal ash versus coal. Barium concentrations are 42 times higher in the ash than in the coal. Only detailed mineralogical analysis can determine which mineral phases of these constituents are soluble and mobile in groundwater, or fixed and relatively insoluble.

1.4.3. Analysis of coal ash chemistry in PADEP minefill permits

There are two methods used to analyze coal ash being placed in Pennsylvania coal mines: a bulk chemical analysis and a leaching test. Bulk chemical analysis reports most of the constituents that make up the ash as elemental concentrations. Sulfur is reported as sulfate. The analysis makes no attempt to determine the mineral phases that contain the constituents or the stabilities of those phases in the environment. Mineral phase identification (using X-ray diffraction and X-ray fluorescence techniques) has been used for CCW characterization to a small extent and would be a good focus for future studies.

A leaching test is used to determine the mobility of trace heavy metals and other metal oxides, as well as more prevalent inorganic constituents in the ash such as sulfate, chloride, and sodium. Pennsylvania requires the use of the Synthetic Precipitation Leaching Procedure (SPLP), described as Method 1312 in US EPA’s Test Methods for Evaluating Solid Waste, SW-846. In the test, a set amount of the CCW, usually 100 grams, is ground to a specified particle size and placed in a container in an extraction fluid that is 20 times the amount of the ash by weight. The extraction fluid has a pH of

4.2 Standard Units. The container is rotated end over end at 30 rotations per minute for 18 hours. The resulting fluid in the mixture, called leachate, is filtered and analyzed for dissolved concentrations of 15 metals and four other inorganic constituents. For a coal ash to be certified for placement in a Pennsylvania coal mine, the concentrations of these constituents cannot exceed maximum acceptable leachate concentrations set by the PADEP in guidelines for the beneficial use of coal ash in mines.⁸ Those concentrations are normally 25 times the “groundwater parameter,” equivalent to Pennsylvania’s drinking water standard, for metals and 10 times the groundwater parameter for nonmetals. Organic constituents are not analyzed in the test. Results of these tests are regularly reported pursuant to the section of PA mining permit permits called Module 25.

While this test determines the quantity of inorganic constituents that leach out of a coal ash sample under controlled laboratory conditions for short periods, it is not designed to simulate actual conditions in the coal mines where CCW is placed. The actual conditions in mines are far more geochemically complex. University researches and several federal agencies, including US EPA, Department of Energy, and the Office of Surface Mining, as well as state regulators, admit that standard leaching tests, like the SPLP, cannot adequately predict how CCW will behave in a mine or any other real-life disposal environment. Concentrations of metals and other constituents in groundwater from CCW in disposal environments are often markedly different from concentrations generated in leachate from CCW in tests such as the SPLP. These deficiencies in the bulk analysis and leaching test used by PADEP cause environmental protection advocates to fear that CCW minefilling permits in Pennsylvania are not sufficiently characterizing the potential of CCW to contaminate groundwater or pollute surface waters. (See detailed discussion of the adequacy of CCW characterization in Chapter 6, *infra*.)

1.4.4. Use of CCW as Alkaline Addition to treat AMD

Current regulatory programs in many eastern states allow for the treatment of AMD with alkaline materials (often lime and sometimes CCW) to effect neutralization and metal precipitation prior to discharge. Alkaline CCW is placed directly against or blended with AMD-forming rock, spoils and coal refuse to permanently treat AMD in situ. It is considered by officials in the coal and electric utility industries and some federal and state government agencies such as the PADEP to be an appropriate and economically sound alternative to conventional AMD treatment.

When ash is used for alkaline addition in a Pennsylvania coal mine, the total amount of calcium in the ash (equated to calcium carbonate found in overburden rock that can neutralize acidity) is ascertained through bulk analysis to derive the net neutralization potential of the ash. This neutralization potential is then measured against the deficiency in alkalinity in overburden rock in the mine to determine the amount of ash needed to neutralize acidity in the mine, usually with an additional amount of ash to assure that enough alkalinity is being applied. However, there is no attempt to quantify the physical availability of the alkalinity in the ash to actually buffer the acidity that will be generated

⁸ PADEP, *Certification Guidelines for Beneficial Uses of Coal Ash*, 563-2112-224, BMR PGM Section II, Part 2, Subpart 24.xxxxxx

from the spoils or coal refuse in the mine. When more pozzelanic or cementaceous CCW such as FBC ash is placed in solid configurations and not blended with the acidic materials in the mine, environmental protection advocates fear that alkalinity bound up in hardened deposits will not be available in amounts sufficient to neutralize high amounts of acidity in mines. Even worse, the CCW placement could result in the mobilization of additional metals from the ash into mine waters if the alkalinity of the ash is eventually exhausted by the AMD.

An extensive four-year column leaching study by Stewart, Daniels and Zelazny (1996) found that bulk-blending alkaline eastern bituminous coal fly ash with acid-forming coal refuse might present a disposal option that could control AMD. However, they concluded, “Our data clearly indicated that ash alkalinity and refuse potential acidity must be balanced to insure long term water quality protection from ash/refuse co-disposal practices and that the breakthrough of acidic leachates may take greater than five years under certain co-disposal scenarios as modeled in our study.” Stewart (1996) stated that “it is evident that most eastern fly ash does not contain sufficient alkalinity to be safely co-disposed with acid-forming coal refuse without addition of supplemental alkalinity.” In addition, exposing ash to strong acidic leaching environments increased the leaching of potentially hazardous trace metals. The leached amounts of metals such as manganese, iron, and copper increased proportionately with the total amount of ash applied (Stewart, 1996).

1.5 CCW, AML, AMD and Waste Coal Burning in Pennsylvania

1.5.1. Historical Perspective

It has been stated “Pennsylvania carries the heaviest burden of abandoned coal mines in the country.”⁹ Serious environmental and safety problems associated with AML include water-filled pits, dangerous vertical highwalls, subsidence and drainage problems, open shafts, abandoned gob and culm piles and, of course, AMD.

These problems represent the legacy of historic mining practices. Pennsylvania has been very successful over the last three decades, through SMCRA and extensive state-sponsored research, in preventing AMD from coal mining. A measure of this success is the relative rarity of AMD generation at new SMCRA-permitted mines. According to PADEP, 17% of the mining permits issued between the years 1977 and 1983 produced post-mining discharges related to AMD. That rate dropped to 2.2% for the period between 1987 and 1996.¹⁰ A large part of the success in preventing AMD in new mines can be attributed to the rigorous state program requiring pre-mining prediction of post-mining water quality. Based on that prediction, permits are customized to prevent AMD formation, or in some cases, permits are denied when prevention appears unlikely.

⁹ Pennsylvania General Assembly, Joint Legislative Air And Water Pollution Control and Conservation Committee, Report on a Proposed Moratorium on the use of Fly Ash in Reclamation Projects, February, 2004.

¹⁰ PADEP, Evaluation of Mining Permits Resulting in Acid Mine Drainage, 1987-1996: A Post Mortem Study, March, 1999

Although modern surface mining permitting decisions and operational practices have largely eliminated AMD as a problem in new mines, the legacy of past practices remains. PADEP estimates that AML constitutes about 250,000 acres in Pennsylvania.¹¹ AMD in Pennsylvania is a problem almost entirely associated with AML.¹²

It should be noted, however, that even in mine settings that generate large amounts of AMD, acidity, and the high levels of metals and sulfates associated with it, usually decline significantly with time. In surface mines, as mining breaks apart pyrite-bearing rocks and coals, increasing their surface area and exposure to oxygen, the acid products are leached from rocks exposed to weathering usually within 10 to 20 years (Meek, F.A. 1996). Leaching of most acidity-producing pyrites and alkalinity-producing carbonates occurs within 20 meters of the surface due to weathering (ATRI, Prediction of Water Quality at Surface Coal Mines, ATRI).

Thus the degree of exposure to air is a key, if not primary, factor involved in the creation of AMD in coal mines. A study by Ziemkiewicz and Meek of eleven, 400-ton piles of acid-producing sandstone and shale in an Upshur County, WV coal mine treated with various amendments of alkaline materials found that concentrations of sulfates exiting all of the piles from AMD declined dramatically (between ~ 85-97%) over an 11-year period and that the rates of sulfate exit from the original pyrite mass in the piles were fairly constant. The study's authors concluded, "It appears that gross physical phenomena, independent of pyrite forms, surface area, amendment, pH or micro properties of the rock control the rate of pyrite oxidation within relatively narrow limits. Since the rate increased with the proportion of sandstone, oxygen diffusion is the likely candidate."¹³ At the same time, whether in active or abandoned surface mines, pyrite oxidation near the surface of large piles of spoils or refuse produces a rind of precipitates from chemical reactions that along with the settling of the piles tends to limit oxygen exposure and therefore retard pyrite oxidation farther beneath the surface of those piles.

Acid discharges from underground mines usually last longer than AMD from surface mines. However, underground mines that are below regional water tables set by creeks and rivers ("below drainage" mines), usually lose their acidity significantly faster than underground mines above the water table. This is because the pyrite reacts at much slower rates and produces only small amounts of acid when left in more anaerobic conditions underwater in flooded underground mines (Evangelou 1995, Fennemore et al. 1998, as cited by Demchak, Skousen and McDonald, 2003). In fact, researchers have found that within 30 years after closure, water in flooded underground mines in the UK, rose from an acidic to a neutral pH and iron concentrations decreased by over 80% (Wood et al. 1999). Other researchers found that iron and sulfate concentrations declined

¹¹ PADEP, "Healing the Land and Water, Pennsylvania's Abandoned Mine Reclamation Program," <http://www.dep.state.pa.us/dep/deputate/minres/reclaimpa/healinglandwater.html>.

¹² But it is important to recognize that not all AML has AMD. AMD is only a subset of the environmental problems associated with AML

¹³ Ziemkiewicz, Paul F. and F. Allen Meek, Jr. Long Term Behavior of Acid Forming Rock: Results of 11-Year Field Studies. Presented at the International Land Reclamation and Mine Drainage Conference and the Third International Conference on the Abatement of Acidic Drainage, Pittsburgh, PA, April 24-29, 1994.

by approximately 50 % over 25 years after closure and flooding in two below drainage mines in the Uniontown Syncline of Pennsylvania and concluded that water changed from acidic to alkaline within 30 years after closure and flooding of these mines (Lambert and Dzombak, 2000). A study of the Montour mine, a Pittsburgh coal seam underground mine in Pennsylvania found that a section that was flooded in 1982 changed from being strongly acidic (pH of ~3.0, acidity of 2,200 mg/L) to net alkaline (pH of 6.4, net alkalinity of 200 mg/L) only seven years after flooding, an improvement likely hastened by good quality water from a separate part of the mine that flooded in 1970, (Donovan et al. 2000).

Even most underground mines that are “above drainage” appear to undergo major improvements in water quality with time. These mines experience significant longer term oxygen exposure and more tenacious acidity than surface mines or below drainage underground mines. Yet a 2003 study of 44 such mines in northern West Virginia by Demchak et al. found significant improvements in water quality in 34 of the 44 mines between 1968 and 2000. Reductions in acidity, iron, and sulfate ranged from 50 to 80%. Thus while the AMD problem is very real, it is abating with time. Treatment programs should be designed with this eventuality in mind. Time and reduction in oxygen exposure are two critically important factors for addressing AMD.

1.5.2. Approaches to Remediation

The remediation of AMD has been a major concern in Pennsylvania for decades. Hundreds of miles of Pennsylvania streams have been, and are being, improved and even cleaned to near pristine conditions by these efforts. Two traditional approaches have proven extremely effective. The first is surface reclamation, which involves regrading and revegetating a site. The second is passively treating AMD using one of a number of constructed features such as wetlands, open limestone channels and anoxic limestone drains. When designed site specifically and adjusted to improve actual performance, these techniques are virtually 100% successful.¹⁴ Both of these approaches, although highly effective, require the expenditure of commonwealth funds.

It is estimated that the price of reclamation and remediation of AML and AMD in Pennsylvania approaches several billion dollars. Federal funding from Title IV of SMCRA, the mainstay of reclamation efforts, amounts to approximately \$25 million annually. To supplement inadequate federal funding, PADEP has sought ways to encourage private industry to reclaim AML. The lynchpins of these efforts have been the promotion of remining at AML sites where refuse piles can be remined (and the accompanying use of coal ash as an alkaline addition and/or fill) and the permitting of waste coal-burning power plants. Both of these actions have led to today’s burgeoning practice of coal ash placement in mines.

¹⁴ Milavec, Pamela J., undated, “Abandoned Mine Drainage Abatement Projects: Successes, Problems and Lessons Learned,” Bureau of Abandoned Mine Reclamation, on PADEP website at <http://www.dep.state.pa.us/dep/deputate/minres/bamr/bamr.htm>, last revised 1/17/03.

In the 1990s, Pennsylvania instituted mining regulations that specifically addressed remining areas impacted by AMD. These regulations provide regulatory relief and incentive for mining companies that reclaim problematic AML as a part of remining activities. A large percentage of the coal ash placement in mines occurs at remined sites, because CCW is often used as alkaline addition at these sites to treat AMD or as fill to achieve approximate original contour at the sites.

However, the success of remining as a means of remediating AMD is significantly less than the virtual 100 % success rate from passive treatment and the near 98 % success rate from preventing AMD at the permitting stage. An in-depth study of PA remining projects showed a success rate of 87%, or 21 of 24 sites evaluated.¹⁵ A broader PADEP study of 110 completed remining operations found net acidity loads were improved or eliminated in 47% of discharges, unchanged in 52 % of discharges and became worse in 1% of discharges. Although the number of discharges with reduced loads of iron, aluminum, manganese and sulfate was always substantially greater than those showing increased loads, 10% of discharges had higher loads of sulfates and 9% of discharges had higher loads of manganese. Furthermore, postmining manganese concentrations in aggregate at these sites actually increased, indicating that virtually all Mn load reductions and most of the reductions in Fe and SO₄ were due to flow reductions.¹⁶

The degree to which coal ash was used in the remining sites in these studies is not clear. However, the standards for defining success in these cases do not include measurement of groundwater quality nor concentrations of trace elements or other constituents that might migrate from coal ash to surface waters regardless of improvements in loadings for acidity, iron, aluminum, manganese and sulfates.

Second, several factors, including the drive to hasten reclamation, have encouraged the proliferation of FBC waste coal-burning plants. The initial driving force for the development of this industry was the passage of the Public Utilities Regulatory Policies Act (PURPA) in 1978. Created in response to the energy crisis in the 1970s, PURPA required electric utility companies to buy the power generated by facilities using non-traditional fuel, such as waste coal. PURPA required electric utilities to purchase this electricity at a rate that matched the traditional power plant cost to produce the electricity. PURPA spawned 16 FBC waste coal plants in Pennsylvania, most built between 1987 and 1995. Today this industry is experiencing a second boom. Reliant Energy's Seward Plant, the largest FBC waste coal plant in Pennsylvania, came on line in 2004. At least 2 more large plants, including the Nemaquin plant in Greene County (with slightly larger capacity than the Seward plant), plan to commence operations over the next few years. The Seward plant alone increases the amount of waste coal burned in Pennsylvania and the amount generated from this combustion by approximately 50%.

¹⁵ Hawkins, Jay. W. "Characterization and Effectiveness of Remining Abandoned Coal Mines in Pennsylvania," 1995. Report of Investigations 9562, U.S. Department of Interior.

¹⁶ Smith, Brady and Hawkins. "Effectiveness of Pennsylvania's remining program in abating abandoned mine drainage: water quality impacts," 2002, Society for Mining, Metallurgy, and Exploration, Inc

Pennsylvania’s “Alternative Energy Portfolio Standard” legislation, enacted in 2004, also encourages the burning of waste coal. Act 213 requires a certain percentage of the energy sold in Pennsylvania be derived from “alternative energy sources,” which included by definition waste coal. PADEP clearly sees waste coal plants/minefilling as a “win/win” proposition. According to PADEP, “Not only are abandoned mine lands reclaimed on the back end of the process through utilization of FBC ash, the reclamation realized at the front end of the process, converting polluting waste coal into an energy resource, could not economically occur if the FBC ash was landfilled.”¹⁷ FBC plants are the touchstone of the minefilling program; approximately 79% of the coal ash placed in mines is generated by FBC plants.¹⁸

The economics of waste disposal also increasingly motivate conventional coal-burning power plants to dispose of their CCW in mines. Seven of the permits examined in this report involved conventional CCW. In Pennsylvania and other coal mining states, coal operators offer attractive haulback provisions that reduce disposal costs for electric utilities. The 21 conventional coal-fired power plants in Pennsylvania produce about 5 million tons of coal ash each year. Over 1.3 million tons of this ash is placed annually in Pennsylvania mines.¹⁹

1.5.3. Driving Economic Forces vs. Environmental Concerns

The environmental community has raised concerns about the lack of safeguards for mine placement and disposal. The waste coal industry has responded with stiff resistance to any proposed change in their waste handling practices. Waste coal operators claim that any imposition of safeguards for the placement of coal ash in mines would result in closure of their plants.²⁰ This assertion is based on the increased disposal costs that would be incurred if the plants were to dispose of CCW in a residual waste landfill. Proponents of this argument assert that cost of landfilling the 5 million tons of CCW produced each year by the waste coal plants would be in excess of 300 million dollars. By their own estimation, the industry admits that minefilling CCW reduces their disposal costs by 89 to 95% over landfilling.²¹ Because waste coal plants must burn a substantially higher volume of waste coal, and produce a significantly higher volume of ash than PC plants, the plants are more susceptible to disposal cost increases.²² Conventional coal plants, because they produce a much smaller volume of ash relative to the power they produce, do not face the same landfilling costs as waste coal plants do.

¹⁷ PADEP Coal Ash Report, 2004.

¹⁸ Ibid.

¹⁹ Ibid. Chapter 1.

²⁰ Joint Legislative Report on Coal Ash.

²¹ Joint Legislative Committee Report states that “This conclusion is drawn with the understanding that the disposal cost per ton of material at a commercial residual waste facility (landfill) is between \$45 and \$90, including transportation....Taking the midrange of cost to be \$67.50 per ton, the cost of landfilling 5 million tons of ash produced each year by the Commonwealth’s waste coal facilities would be approximately \$337.5 million per year.”

²² Cogeneration plants are not permitted to increase prices to reflect increased operation costs. Under PURPA, cogeneration and other small power production facilities are entitled to sell electricity to utilities at a negotiated price. Utilities purchase this electricity through long-term (typically 20 years) contracts at a fixed price per kilowatt hour.

While no one questions the objective of reducing AMD through proper reclamation of active mines, regrading and revegetating abandoned mines and using passive treatment of acid discharges, the dispute centers on whether use of CCW with unstable pH, elevated levels of metals and soluble salts is a prudent long term approach to solving the AMD problem. In particular, PADEP and other proponents of the waste coal industry point out that the electricity produced and the reclamation of AML achieved by burning waste coal would not be economically possible if they have to meet the same disposal standards that generators of CCW must meet when disposing CCW outside of coal mines. However, environmental protection advocates question whether the fundamental objective of protecting and restoring the hydrologic balance in coal mines under SMCRA should be compromised by this economic objective and ask why public funds for reclamation of abandoned mines should not be significantly increased to address the objective of abating AMD on these lands.



CAPTION - The reddish, tan, brown FBC waste coal ash placed in many Pennsylvania Mines stands out against darker gray and black coal refuse. Coal refuse and waste coal are one and the same and called “culm” if from anthracite coal or “gob” if from bituminous coal. Here FBC culm ash is being piled on top of culm and mine rock at the AC Fuels Co Mine in northern Schuylkill County, a site not examined in this report. Photo by Steven Dreyer, McAdoo, PA.