

Proceedings of

STATE REGULATION OF COAL COMBUSTION BY-PRODUCT PLACEMENT AT MINE SITES: A TECHNICAL INTERACTIVE FORUM

Held at
Wyndham Hotel
Harrisburg, Pennsylvania
May 4-6, 2004



Edited by:
Kimery C. Vories
Anna Harrington



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FOREWORD

Beginning in May of 1994, the Office of Surface Mining (OSM) has taken an active role in encouraging and promoting technological advances, research, and technology transfer related to the use and disposal of coal combustion by-products (CCBs) at mine sites. The primary activities and accomplishments of OSM in this area have been the establishment of a multi-interest group steering committee that has:

- (1) conducted national technical interactive forums on:
 - “Coal Combustion By-Products Associated with Coal Mining” in October of 1996 at Southern Illinois University
 - “The Use and Disposal of Coal Combustion By-Products at Coal Mines” in April of 2000 at the National Energy Technology Laboratory (NETL) in Morgantown, West Virginia
 - “Coal Combustion By-Products and Western Coal Mines” in April of 2002 in Golden, Colorado
 - and now “State Regulation of CCB Placement at Mines Sites” in May of 2004 in Harrisburg, Pennsylvania;
- (2) edited, published, and distributed thousands of copies of the forum proceedings;
- (3) provided technical assistance to the American Society for Testing Methods (ASTM) on draft guidance for CCBs on mine sites; and
- (4) developed and managed an Internet Website dedicated to providing a user friendly guide to CCB literature, organizations, EPA rule-making, and educational events.

Concerning cooperation with other Federal agencies, OSM has signed a Memorandum of Understanding with the NETL to collaborate on CCB research and issues. OSM serves on the NETL national steering committee of the Combustion By-Products Recycling Consortium in order to assist in directing CCB research efforts by the U.S. Department of Energy. OSM is currently working with the U.S. Environmental Protection Agency to investigate whether or not additional Federal regulations are necessary to protect the public and environment when CCBs are placed at the mine site. OSM is contributing to the effort by the Interstate Mining Compact Commission and member States on developing a State consensus on how best to regulate CCB placement at both SMCRA and non-SMCRA mine sites.

OSM contributes to other professional organizations that focus on the development of useful products and applications of coal combustion by-products such as the:

- (1) technical program committee for the biennial International Ash Symposium conducted by the University of Kentucky Center for Applied Energy Research; and
- (2) national steering committee of the 2005 World of Coal Ash.

OSM staff have presented technical papers related to CCB placement on Surface Mining Control and Reclamation Act (SMCRA) mine sites at numerous technical conferences.

The purpose of the May, 2004 forum was to provide: an organized format for discussion of technical issues concerning the placement of CCBs at mines regulated primarily by State agencies; a field tour of Pennsylvania Anthracite Region where fluidized bed combustion ash is used to reclaim abandoned mine land sites; an easily understood, state-of-the-art summary talk by knowledgeable speakers; a published, post-forum proceedings that summarizes the presentations and participant discussions and recommendations; and provide access to the discussions for all interested participants.

Based on the results of the above efforts, OSM will assess the outcomes of the forum and CCB activities and make recommendations for potential revisions to OSM policy and plans for enhancement of additional technology transfer events. I would like to sincerely thank the steering committee, invited speakers, and participants for their time and efforts in making this program a success.

Kimery C. Vories
CCB Steering Committee Chairperson



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RETIREMENT

Stephen Smith was a long time employee of TXU in Dallas, Texas. He has served on the CCB Steering Committee since 1998. He recently retired to take up country living in Texas. When asked if he had some parting words of wisdom he offered the following:

“I enjoyed the time we spent together in planning the CCB forums and think that was some time well spent. I know you and the Steering Committee will do a great job in planning and carrying out the event next spring.

I have always been partial to a line from an old C&W song called, *Cherokee Fiddle*. I think it also applies to the work you and the Steering Committee are doing and will continue to do ...

“If you’re going to
make a living boy,
you got to put on
a good show!”

I know that the Steering Committee will put on a good show. Good Luck.”

WELCOME

Brent Wahlquist
Appalachian Regional Director
Office of Surface Mining

Good morning and welcome to the fourth in a series of OSM sponsored Technical Interactive Forums on the Placement of Coal Combustion By-Products at Mine Sites. It is indeed a pleasure to be here today at the beginning of two days of discussion and information-sharing on this important environmental topic. I am glad that so many people from so many parts of the country are participating, from all levels of government, from industry, universities, and the general public. Please give a special welcome to Dr. Richard Kruger who came all the way from South Africa to join in this discussion. This is an excellent opportunity for communicating problems, solutions, and concerns related to placement of Coal Combustion By-Products at mine sites.

The goal of the first two forums in 1996 and 2000 was to establish a national state-of-the-art on CCB placement at coal mine sites. The third forum in 2002 was designed to look at the issue from the perspective of the semi-arid Western U.S. in addition to regulatory changes at the national level. The goal of this forum is to focus on: (1) CCB placement at mine sites where State Mining Programs are the primary regulator; (2) the unique application of Fluidized Bed Combustion Ash, produced from recycling waste coal into electrical energy, and the beneficial application for remediation of acid mine drainage and the reclamation of abandoned mines; and (3) the timely issue of proven environmental damage cases, how they happened, and what we should do to avoid them in the future.

As you are aware, the National Academy of Science has been recently tasked with investigating this issue. The information provided by this forum and those that went before it should go a long way in aiding the National Academy in their efforts to inform Congress on the ability of current mining regulatory programs to protect the public and the environment when these materials are permitted for placement on mine sites.

We are already off to a very good start after that excellent field tour yesterday where you were able to visit several abandoned mines sites that are being reclaimed with Fluidized Bed Combustion Ash produced by the innovative Pennsylvania program for converting waste coal into electrical energy. Albert Dalberto and Michael Menghini and their associates at the Pennsylvania Department of Environmental Protection have been absolutely tireless in making sure that the field trip was very successful. Our hosts on the field tour included: Paul Freeman at the Mount Carmel Cogen Site; Ed Griegal at the Wheelabrator Frackville Energy Company Site; David Martin at the Gilberton Power Plant; Vince Devine and Rick Letarte at the Schuylkill Energy Resources Site; and Randy Oswald and John Olebrach at WPS Westwood Generating. They have shown us real world applications that most of us would not otherwise experience.

I would like to commend the support and commitment of our cosponsors Lynn Brickett at the U.S. DOE National Energy Technology Laboratory, Dr. Ron Carty at the Illinois Clean Coal Institute, and Ken Ladwig at the Electric Power Research Institute whose sponsorship support has been essential in being able to ensure that we can provide quality speakers for a quality experience.

I would also like to thank the CCB Steering Committee who have been working hard to organize this event since January of 2003. They include:

- Dave Goss of the American Coal Ash Association
- Robert Bessette of the Council of Industrial Boiler Owners
- Dan Wheeler of the Illinois Department of Commerce & Economic Opportunity, Office of Coal Development
- John Mead of the Coal Research Center at Southern Illinois University
- Dr. Tarunjit Butalia of The Ohio State University
- Alfred Dalberto of the Pennsylvania Bureau of Mining and Reclamation
- Sid Stroud with TXU
- Debra Pflughoeft-Hassett with the University of North Dakota Energy & Environmental Research Center
- Lynn Brickett with the National Energy Technology Laboratory
- Hendric van Oss with the USGS Minerals Information Team
- and the staff of OSM-Joe Galetovic, Peter Michael, Randall Mills, and Kimery Vories

Please feel free to contact any of the steering committee with questions or concerns about this or future events.

It is always true that the more we know, the more options we have. I am optimistic that constructive dialogues, such as those held here, will lead to a better understanding of the benefits and risks involved with the beneficial placement of coal combustion by-products at mine sites.

I commend all the forum participants for being part of this valuable information exchange. The public and the coalfield residents can only benefit from the information that is shared and the knowledge that is gained at this event. I thank you for applying your minds to the task and I wish you success in your efforts on behalf of the coalfield environment.

WHAT IS A TECHNICAL INTERACTIVE FORUM?

Kimery C. Vories
USDI Office of Surface Mining
Alton, Illinois

I would like to set the stage for what our expectations should be for this event. This is the fourth in a series of technical interactive forums cosponsored by OSM on aspects of Coal Combustion By-Product (CCB) placement at mine sites. Copies of these earlier forums are available on OSM's technology transfer CD and at the CCB Information Network Website at www.mcrcc.osmre.gov/ccb.

The steering committee has worked hard to provide you with the opportunity for a free, frank, and open discussion on the state of the art in CCB placement at mine sites that is both professional and productive. Due to the sometimes controversial nature of this subject, I must stress that by using the term "professional" it is expected that all discussions should be respectful and not personal.

Our rationale for the format of the technical interactive forum is that, unlike other professional symposia, we measure the success of the event on the ability of the participants to question, comment, challenge, and provide information in addition to that provided by the speakers. We anticipate that, by the end of the event, a consensus will emerge concerning the topics presented and discussed and that the final proceedings will truly represent the state of the art in State Regulation of CCB placement at mine sites.

During the course of these discussions, we have the opportunity to talk about technical, regional, and local issues, while examining new and existing methods for finding solutions, identifying problems, and resolving controversies. The forum gives us the opportunity to:

- share our experiences and expertise concerning CCB placement at mines,
- outline our reasons for taking specific actions, and
- give a rationale for our actions concerning testing, permitting, water monitoring, material handling, reclamation, and protection of the environment concerning CCB placement at mines.

A basic assumption of the interactive forum is that no person present has all the answers or understands all of the issues. It is also assumed that some of these issues, solutions, and concerns may be very site, region, or CCB type specific.

The purpose of the forum is to:

- present you with the best possible ideas and knowledge, during each of the sessions, and
- promote the opportunity for questions and discussion by you, the participants.

The format of the forum strives to improve the efficiency of the discussion by:

- providing a copy of the abstract and biography for each speaker that you may want to read before hand in order to improve your familiarity with the subject matter and the background of the speaker;
- recording the talks and discussions for later inclusion in the post forum publication so that you do not have to worry about taking notes. For this reason, we will require that all participants speak into a microphone during the discussions;
- In order for us to make the most efficient use of time, and ensure that you, the participants, have the opportunity to provide questions and comments, we require our session chairpersons to strictly keep to the time schedule;
- A **green light** will be displayed at the beginning of the talk. A **yellow light** will be displayed for the last 5 minutes of the talk. A **dim red light** will be displayed for 30 seconds followed by a **blinking red light** that will signal that the talk is over and the speaker has 5 minutes for questions;
- In the post forum publication, issues raised during the discussions will be organized based on similar topic areas and will not identify individual names. All registrants will receive one electronic copy of this proceeding. This publication will be very similar to the proceedings of earlier forums conducted by OSM and are available for your viewing at the OSM exhibit.

It is important to remember that there are four separate opportunities for you, the participants, to be heard:

- 5 minutes will be provided for questions at the end of each speaker's talk;
- 20 plus minutes of participant discussion is provided at the end of each topic session. The chairperson will recognize each participant that wishes to speak and they will be requested to identify themselves and speak into one of the portable microphones so that everyone can hear the question;
- At the end of the forum, we will conduct an open discussion on where we should go from here;
- and finally, a yellow forum evaluation form has been provided in your folder. This will help us to evaluate how well we did our job and recommend improvements for future forums or workshops. Please take the time to fill out the yellow evaluation form as the forum progresses and provide any additional comments or ideas. These should be turned in at the registration desk at the end of the forum.

One of the reasons for providing refreshments during the breaks and lunch is to keep people from wandering off and missing the next session. In addition, the breaks and lunch provide a better atmosphere and opportunity for you to meet and discuss concerns with the speakers or other participants. Please take advantage of the opportunity at break time to visit the exhibits and posters in the break area. When the meeting adjourns today, all participants are invited to a social reception where refreshments will be provided.

Finally, the steering committee and I would like to thank all of the speakers who have been so gracious to help us with this effort and whose only reward has been the virtue of the effort. I would also like to thank each of you, the participants, for your willingness to participate and work with us on this important issue. Thank you.

COAL COMBUSTION BY-PRODUCTS STEERING COMMITTEE RECOMMENDATIONS

Future Forum Topics

1. Technical case studies of “damage case claims” about coal mine sites.
2. Case studies that analyze benefits and negative aspects of placement.
3. Leachate testing and CCB characterization of different materials and suitability for different mine sites.
4. Economics of placement options.
5. Impacts of evolving power plant technology on ash placement (i.e. complications with mercury removal).
6. Movement of Boron in leachate and spoil.
7. Additives to reduce leachate mobility.

OVERVIEW OF THE FLUIDIZED BED COMBUSTION PROCESS AND MATERIAL

Francois Botha
Illinois Clean Coal Institute
Carterville, Illinois

Abstract

The advent of fluidized bed combustion (FBC) technology has proven to be one of the most promising of the emerging Clean Coal Technologies. Low-grade fuels, that have a large ash and high sulfur content, can usually not be utilized in pulverized coal combustion (PCC) units. FBC units can not only efficiently burn low-grade fuels, but can also handle wide variations in fuel quality while still achieving strict air emission requirements.

With the commercial application of FBC units gaining widespread acceptance, there is a need to understand the similarities and differences in not only the FBC and PCC processes, but also in the characteristics of the solid byproducts that are generated. In the FBC process, limestone is added to the boiler to act as a sorbent for sulfur. The operating conditions for FBC units, in addition to the fuel and sorbent characteristics, directly contribute to the chemical characteristics of the residues. FBC bottom ash is a mixture of fuel ash, unburned carbon residues, and lime particles coated with sulfate layers. The ash properties are substantially different from typically PCC residues marketed as ASTM Class C or Class F fly ash. While PCC ash has found an application mainly in the cement industry, the utilization options for FBC ash is more diverse due to the influence of the sorbent on the ash chemistry.

Combustion Process and Conditions

Pulverized coal combustion (PCC) is the most commonly used method in coal-fired power plants. The technology is well developed, accounting for well over 90% of coal-fired capacity. The feed coal for a PCC unit is ground (pulverized) to a fine powder with about 75% of the coal particles below 75 μm in size. The pulverized coal is blown with part of the combustion air into the boiler through a series of burner nozzles. Combustion takes place at temperatures from 1300-1700°C, and the particle residence time in the boiler is typically 2-5 seconds.

In PCC, flue gas cleaning consists of particulate emissions control technology to capture the fly ash, such as ESP's (Electrostatic precipitators) and fabric filters, and NO_x and SO_x control. Primary measures for NO_x control (burner optimization, low- NO_x burners and overfire air) are now considered integral parts of a newly built power plant and existing units retrofit them whenever they are required to reduce their NO_x emissions. SCR (Selective catalytic reduction) is accepted throughout the world as the proven commercial option to achieve high NO_x removal. Sulfur is typically removed via Flue Gas Desulfurization (FGD) technology, where calcium-, sodium- and/or ammonium-based sorbents react with the SO_2 in the flue gas. Wet FGD scrubbers are the most widely used for SO_2 control throughout the world. Emissions from new PCC units with appropriate flue gas cleaning units can meet all current requirements, although the capital cost of these measures can represent about one third of the cost of the unit when meeting the most stringent current standards.

The advent of fluidized bed combustion (FBC) technology has proven to be one of the most promising of the emerging Clean Coal Technologies. Lower grade fuels, that have a large ash and high sulfur content, can usually not be utilized in pulverized coal combustion (PCC) units. FBC units can not only efficiently burn low-grade fuels, but can also handle wide variations in fuel quality while still achieving strict air emission requirements.

With the commercial application of FBC units gaining widespread acceptance, there is a need to understand the similarities and differences in not only the FBC and PCC processes, but also in the characteristics of the solid byproducts that are generated. Relatively coarse particles at around 1 mm to 3 mm in size are fed into the combustion chamber. Combustion takes place in a FBC unit at temperatures from 800-900°C, resulting in reduced NO_x formation compared with PCC. N_2O formation is, however, increased. In the FBC process, limestone is added to the boiler to act as an in-situ sorbent for sulfur. The direct injection of limestone into the bed offers the possibility of economic SO_2 removal without the need for expensive FGD equipment. Thermal decomposition of

CaSO₄ in normal fluidized bed combustion conditions is not probable because of low temperatures in the reactor. Circulating beds use a higher fluidizing velocity, so the particles are constantly held in the flue gases, and pass through the main combustion chamber and into a cyclone, from which the larger particles are extracted and returned to the combustion chamber. Because of recirculation of the bed material, particle residence times are relatively long compared with the gas residence time, and can be measured in tens of seconds. Commonly only particulate removal (via ESP) is required in order to meet emissions limits, but in some locations additional NO_x reduction is also required.

Comparison Between Pulverized Coal Combustion (PCC) and Fluidized Bed Combustion (FBC)

Parameters	PCC	FBC
CaO in Boiler	No	yes
Coal Particle Size	< 75 μm	1 - 3 mm
Coal Residence Time	seconds	minutes
Combustion Temperature	high 1300-1700 °C	low 800-900 °C
Ash Porosity	low	high
Ash pH	neutral - acidic	alkaline
Net Thermal Efficiency, %	33-35	36-40
* Emissions: SO ₂	1.2	0.1
* Emissions: NO _x	0.6	0.15
NO _x origin	thermal, fuel	fuel
Cost of Electricity	100	90

* Emission values in lbs/mmBTU

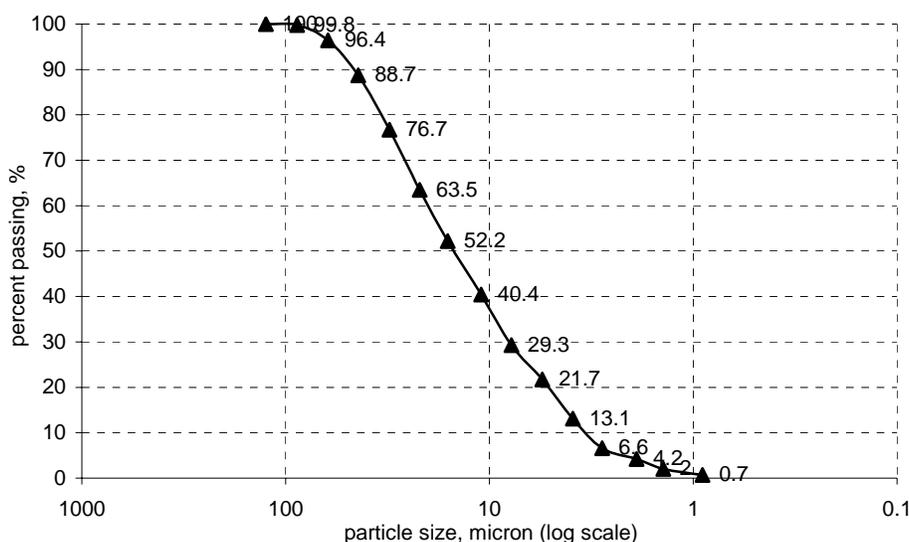
FBC By-Products

The operating conditions for FBC units, in addition to the fuel and sorbent characteristics, directly contribute to the chemical characteristics of the residues. FBC bottom ash is a mixture of fuel ash, unburned carbon residues, and lime particles coated with sulfate layers. The residues consist of the original mineral matter, most of which does not melt at the combustion temperatures used. Where sorbent is added for SO₂ removal, there will be additional CaO/MgO, CaSO₄ and CaCO₃ present. In general, the FBC fly ash composition distinctly differs from that of fly ash produced by conventional combustion, above all by the absence of glass, mullite and other high-temperature phases, showing on the other hand much higher calcium sulfate content and higher portion of unmetamorphosed or only slightly thermally influenced coal minerals (micas, feldspars, clay minerals). FBC ashes are composed principally of gypsum [CaSO₄], lime [CaO], quartz [SiO₂], and associated oxides of iron, magnesium, and dehydroxylated clays originating from the fuel ash components. Due to the high free lime content the leachates from FBC ash will be strongly alkaline. Carbon-in-ash levels are higher in FBC residues than in those from PCC.

Oxide Compositions (%) for Various Coal Combustion Ashes

Oxide Analysis	FBC bed material	FBC ash	C-ash	F-ash
Si	45	40	36	44
Al	6	14	15	17
Fe	3	8	7	18
Si+Al+Fe	54	62	57	79
Ca	26	15	22	5
SO ₃	16	8	12	1
LOI (950 °C)	2	11	1	8

Particle Size Distribution Curve for FBC Fly Ash



The particle size distribution curve shows that about 96% of the FBC fly ash particles are smaller than 63 μm (-250 mesh) in size. The average particle size (D_{50}) is about 15 μm based on the plot. A uniformity coefficient greater than 4 shows that most of the grains are of the same size. More than 80% of the particles are within the range of 44 microns to 5 microns in size. The average specific gravity of the FBC fly ash was 2.66.

FBC Fly Ash Utilization

Diverse utilization options have been studied for FBC coal ashes. The potential applications include:

- Construction applications: cement substitute, concrete block production, brick production, soil stabilizer, roadbase/subbase materials, structural fill materials, and
- synthetic aggregates;
- agricultural applications: liming and soil amendment;
- waste stabilization: acidic waste stabilizer and sludge stabilizer.

Ash Use in Construction Applications: FBC fly ashes cannot be classified as Class F or C, because of low FAS (ferric oxide, alumina, and silica) and high SO_3 content. Even though the fly ash may not qualify as a Portland cement admixture, it may have the potential for use in concrete blocks. Bottom and fly ash can be used as an aggregate and pozzolan in concrete blocks. The bottom ash used as an aggregate has a lower unit weight than many naturally occurring aggregates, thus reducing the weight of the block. FBC fly ash with properties of Class C and F can be used as a partial replacement for Portland cement in some block plants. The free lime, carbon content and sulfate contents of FBC ashes can limit their utilization in concrete block production. Free lime in ash will form water-soluble calcium hydroxide, resulting in weakening of the block from contact with moisture. As with Portland cement concrete, ettringite can form in concrete blocks due to high sulfate levels resulting in mechanical weakening of the block. It is possible that FBC ash may replace some Class C or F fly ash or Portland cement in more moderate strength blocks. Such materials may not be preferred for heavy construction applications.

Soil/Mine Spoil Amendment: Ash streams from CFB boilers firing bituminous coals may be suitable for liming, depending upon how calcium is partitioned between the fly ash and bottom ash. FBC ash streams can also be used to stabilize waste streams from a variety of processing operations. This stabilization includes solidification and fixation of sludge materials for landfilling, neutralization of acidic wastes, and municipal sludge waste sludge. For each of these applications, the suitability of CFB ash is enhanced by its free lime content. Greenhouse studies demonstrated that FBC fly ash amended soils resulted in higher plant productivity than typical ag-lime-amended spoils. These results possibly are due to pH and nutritional issues, but root penetration was undoubtedly a factor.

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RE-MINING AND RECLAMATION OF LARGE ABANDONED BITUMINOUS COAL REFUSE PILES USING FBC ASH: A WATERSHED RESTORATION PERSPECTIVE

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Abstract

Large abandoned bituminous refuse piles present one of the more difficult abandoned mine land challenges. Using typical methods (re-grading and re-vegetating the piles in place), reclamation cost is large, and water quality problems usually remain. Burning the piles as fuel in waste-coal plants and using the alkaline ash in site reclamation presents an alternative reclamation method that consumes no Abandoned Mine Land (AML) monies. Abandoned refuse piles of various sizes dot the Blacklick Creek watershed in Cambria and Indiana Counties, Pennsylvania. Large piles at the towns of Revloc and Colver are significant sources of AMD pollution to the Blacklick Creek. These piles must be abated to restore the watershed. For the past thirteen years, Ebensburg Power Company has been re-mining and reclaiming the Revloc refuse piles using FBC ash. Downgradient groundwater is improving at Revloc, pollution loading from seeps has declined markedly, and improvement in stream quality has been substantial. On the North Branch, Maple Coal Company is re-mining and reclaiming the Colver refuse pile using FBC ash. The site is less than one-half completed. Downgradient groundwater quality has improved noticeably, as have seeps from the pile; stream quality has also improved. If current trends continue, completion of both these re-mining projects will be significant steps toward the restoration of Blacklick Creek.

Introduction

Abandoned bituminous coal refuse piles present significant challenges to abandoned mine land (AML) reclamation efforts, especially from a water quality aspect. Standard AML reclamation typically includes grading the piles, covering them with soil, and establishing vegetation. This approach is successful at eliminating erosion and sedimentation, and certainly improves aesthetics. It may reduce, but cannot eliminate, infiltration. After traditional reclamation, abandoned piles are likely to continue to contaminate groundwater and surface water with worst-case acid mine drainage (AMD). The average AML contract cost for abandoned refuse piles in PA of \$11,600 is about twice that for reclaiming abandoned surface mines (Lighty, 2004). While seepages from the piles usually are not high volume, the extremely poor quality is often beyond the capability of current passive treatment technology. Abandoned refuse piles were often placed in lowland areas or draped over hillsides so that any discharge frequently occurs at or near stream banks, leaving little room for treatment. And, treating the discharges does not address polluted groundwater.

Burning coal refuse in fluidized bed combustion (FBC) power plants to generate electricity, and in some cases heat, has provided another avenue to address the significant AML challenge presented by these piles. The typical refuse reprocessing operation screens the refuse, sending the fine material to the power plant for fuel. The coarse material, such as harder rock and red dog clumps, is set aside. Alkaline-rich ash from the FBC process, in which limestone is injected into the boiler to capture air pollutants, such as sulfur dioxide, is returned to the site. The coarse reject material is layered and compacted with the ash. The combustion process oxidizes sulfide minerals in the refuse, eliminating its acid-producing potential. The ash includes excess lime, giving it both physical and chemical properties that make it a beneficial material for encapsulating the reject material, and limiting the potential for further AMD production.

Blacklick Creek is a major tributary of the Conemaugh River in southwestern PA, within the Ohio River Basin. The Blacklick Creek watershed encompasses approximately 192 mi² in Cambria and Indiana Counties. While some tributaries of Blacklick Creek are of good chemical quality, much of the watershed is AMD-impacted, due to a long history of both surface and deep mining. Blacklick Creek begins at the town of Vintondale on the border of Cambria and Indiana Counties; it is formed by two tributaries, the South Branch Blacklick Creek (South Branch), and the North Branch Blacklick Creek (North Branch). The South Branch supports native brook trout down to the village of Revloc, where it flows through a large coal refuse pile; immediately downstream of the pile it is devoid of

aquatic life. While it suffers from flow loss to underlying deep mines, the North Branch is stocked with trout, and is of decent chemical quality until it encounters Elk Creek at SR 422. Elk Creek is badly degraded by mine drainage, with a large coal refuse pile near the town of Colver being a significant source of the mine drainage. Both the North and South Branch have other significant mine drainage problems on them, however, neither stream can be effectively restored without eliminating or greatly reducing the pollution load from the abandoned refuse piles that first degrade them. Refuse reprocessing operations have been re-mining the Revloc and Colver refuse piles to recover fuel for two waste-coal power plants, and have been reclaiming the piles with FBC ash.

The Blacklick Creek Watershed Association (BCWA) recently celebrated its tenth year of existence. BCWA is a community-based volunteer organization dedicated to restoring stream quality. In recent years, BCWA and its partners, such as the Cambria County Conservation and Recreation Authority, AMD & Art, the Aultman's Run Watershed Association, the US Army Corp of Engineers, and PA DEP have completed over \$5 million worth of stream assessments and treatment system construction projects. The dedication and success of BCWA gives reason to hope that the quality of Blacklick Creek may one day be restored, but several large abandoned refuse piles must be remediated for that to occur.

The Revloc Sites

Site Characterization/Setting

The Ebensburg Power Company Revloc sites are located directly east of the village of Revloc and south of highway US 422 in Cambria County. Refuse is from the Bethlehem Mines Corporation Mine 32 Lower Kittanning deep mine that operated during the middle decades of the twentieth century. The refuse was placed in a lowland area where an unnamed tributary entered the South Branch. The refuse actually dammed the South Branch, producing a pond on the upstream side of the pile.

Ebensburg Power Company obtained separate mining permits on the northern and southern sections of the Revloc pile, which are separated by the South Branch. The company permitted the larger northern part of the pile under surface mining permit # 11880201 (Revloc 1), which DEP issued in 1989. Revloc 1 contained approximately 3.8 million tons of coal refuse spread over approximately 56 acres (Ebensburg Power Company, 1988). In 1997, the company obtained surface mining permit #11960202 (Revloc 2), which included 0.7 million tons of coal refuse (Ebensburg Power Company, 1996). The Revloc 2 site contained primarily reject material from an earlier, unsuccessful reprocessing attempt by another company on the Revloc 1 area.

Ebensburg Power Company began removing refuse from the Revloc 1 site at the end of 1990 and began bringing ash back to the site in very early 1991, when its 50-megawatt FBC cogeneration facility, located in Ebensburg, PA, went online. Mining began on the northern end of the site adjacent to Route 422 and has advanced toward the southwest. The company activated the Revloc 2 site in the fall of 1997. Excavation on Revloc 2 began on the eastern side of the pile, in an area known to be burning. Thus, the fire was extinguished early in the operations to end that source of air pollution and to preserve the useable fuel in the pile.

The mining plan for both Revloc sites is similar. The refuse is screened to remove oversized material. The oversize and already burned material (red dog) is set aside. The fine refuse is sent to the power plant for fuel and ash is trucked back to the site. The ash is mixed, layered and compacted with the oversized reject and red dog in an area behind the working face. On the Revloc sites, the operator has been removing the refuse material down to original ground, even in areas where the pile has been extensively burned. This practice, which results in virtually all potentially acid-forming material being encapsulated in alkaline ash, insures that there are no coarse-grained conduits for water or air within the reclaimed site. In some areas, soil buried beneath the pile is recovered for use in final site reclamation. A layer of soil typically one-foot thick is spread and then seeded with a grass/legume mixture. A portion of the Revloc 2 site has also been planted in black locust.

Mining operations on the Revloc sites are nearing completion, with about one to two years of mining and reclamation to be completed. The Revloc 2 site is about 50% completed as of this writing, but because of its smaller size, and because it is being mined concurrently with Revloc 1, it too will likely be completed in the next 1 to 2 years.

Monitoring Results to Date at the Revloc 1 Site

Water quality results presented in this chapter will sometimes be compared to the maximum contaminant levels (MCL's) for drinking water supplies. The MCL's are a common benchmark with which many people are familiar. MCL's tend to be conservative to protect human health. Results that exceed an MCL must be considered in terms of cause/effect relationships, the use(s) or lack thereof of the water being tested, the volume of the water if it is a discharge, background conditions that existed prior to the condition being studied, and the overall setting of the project and sample point(s). Figure 1 shows the approximate locations of monitoring points on the Revloc sites.



Figure 1. Aerial photo circa 1988 showing the Revloc sites and key associated monitoring reports. The photo was obtained from the permit application for Revloc 1.

Figure 2 displays the available results at downgradient monitoring well MW-1 for acidity, iron, and sulfate, three of the parameters most commonly elevated in mine drainage. Groundwater downgradient of the pile was, not surprisingly, severely degraded by acid mine drainage prior to the Ebensburg Power Company operation. The data show a steady trend of declining concentrations for acidity, iron, and sulfate throughout the monitoring period. The decline appears to have begun prior to initiation of Ebensburg Power Company's operations. The site had been disturbed by another operation approximately 10 years before Ebensburg Power Company permitted the site; it is possible that some of the earlier declines in concentrations are due to the natural attenuation of the results of that earlier disturbance of the pile.

Among trace metals at MW-1, chromium, copper, and barium, with an occasional exception, have remained below detection limits, and below the maximum contaminant level (MCL) for drinking water supplies. Cadmium has consistently been below the detection limit, but the limit reported by the laboratory in this case (0.05mg/L) is in excess of the MCL. The available data show no trends in lead concentrations; the lab reporting the data has used a relatively high detection limit of 0.1 mg/L. Other than for a spike in arsenic and selenium concentrations during the 1992-1994 time period, which may have been due to initial disturbance of the coal refuse, trace metal concentrations at MW-1 have mostly been unremarkable. The median concentrations of the data collected after coal ash began to be added to the site are either below detection limits or, where detectable, below the MCL's. Table 1 summarizes the trace metal concentrations reported from MW-1.

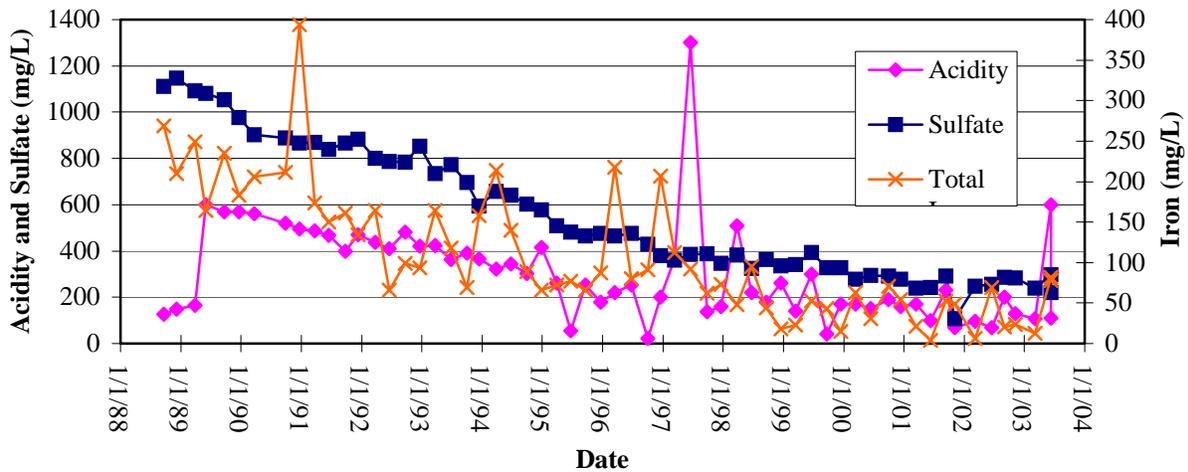


Figure 2. Acidity, sulfate and iron concentrations at MW-1.

Table 1. Comparison of median trace metal concentrations (mg/L) at Revloc 1 down gradient well MW-1.

MW-1	Al	As	Cd	Cr	Cu	Pb	Hg	Se	Zn
Pre-Ash Use	19.3	0.0035	*	*	*	*	*	0.0020	0.690
Post-Ash Use	6.5	0.0067	*	*	*	*	*	0.0213	0.390

*Indicates that data are dominated by values below the laboratory's reported detection limits.

Monitoring well MW-2 is an upgradient well for the Revloc 1 site. The data from it show low TDS water with no mine drainage influence, confirming its upgradient position. The data are generally unchanged with time and will not be further discussed herein.

Monitoring well MW-3 is located immediately adjacent to the north-central part of the Revloc 1 site. Water quality in MW-3 has been variable throughout the monitoring period, showing AMD contamination initially, clearing up for a while, and then showing AMD contamination once again. The trace metal concentrations in MW-3 do not show any noticeable trends, with the possible exception of selenium, which may have increased in the 1999 through 2002-time period. Selenium concentrations, however, have remained below the MCL of 0.050 mg/L, and the latest available analysis from a sample collected in June 2003 was less than 0.007 mg/L.

The Revloc 1 abandoned refuse site was producing six discharges of very poor acid mine drainage leachate, designated 4SP, 4SPA, 4SPB, 6SP, 22SP, and 23SP, when the site was permitted by Ebensburg Power Company. The discharges all emanate from the pile in close proximity to the South Branch. The most significant of these discharges in terms of both flow volume and pollution load is 4SP, which flows directly into the South Branch from an abandoned railroad grade that forms the southern boundary of the pile.

Figure 3 shows the change in flow and pollution load from 4SP with time. Acid and aluminum loads both been reduced substantially during the monitoring period. The decline in pollution load is the result of both improved water quality and a decrease in the discharge flow.

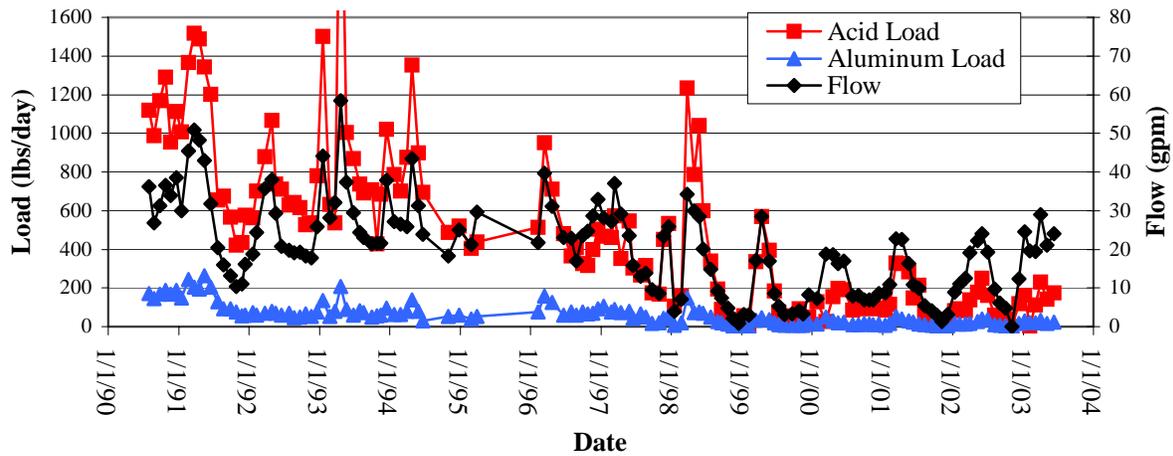


Figure 3. Flow, acid load and aluminum load at discharge 4SP.

Table 2 compares baseline and recent data for all the point-source polluting discharges from the Revloc 1 site. When the median values of the first two years of monitoring data are compared to the median values of the last two years of data, the pollutant load reductions to the South Branch from the discharges are: 1935 lbs/day of acidity; 283 lbs/day of aluminum; 3675 lbs/day sulfate. Additional reductions in pollutant loads in terms of direct groundwater baseflow to the stream are also likely occurring.

Table 2. Summary of flow and AMD pollutant load data for the discharges from the Revloc 1 site. Background data were collected in 1990-1991 and recent data were collected in 2002 to mid 2003. N=18 for most of the background and recent data sets. Flows are in gpm and loads are in lbs/day.

	Totals	4SP	4SPA	4SPB	6SP	22SP	23SP
Background Flow	67.8	31.6	13	17	2.7	2.9	0.6
Recent Flow	38.4	19.2	2.6	15.3	0	0.4	0.9
Background Acid Load	2137.7	1060	578	430.9	62.2	6.4	0.2
Recent Acid Load	203.4	116	28	58.3	0	0.7	0.4
Background AL Load	312.5	168.8	83.9	51.9	7.2	0.7	0.01
Recent AL Load	29.9	17.1	5	7.7	0	0.1	0.06
Background Fe Load	3.4	0.89	0.4	2	0.1	0	0.01
Recent Fe Load	0.45	0.15	0	0.3	0	0	0
Background Mn Load	8.61	4.1	2	2	0.3	0.2	0.01
Recent Mn Load	2.24	0.99	0.2	0.9	0	0.1	0.05
Background SO4 Load	5084	1406	724	2854	87	12.4	0.4
Recent SO4 Load	1409	260	51	1091	0	4.1	3.1

Monitoring Results to Date at the Revloc 2 Site

The Revloc 2 site, located directly across the South Branch from Revloc, has four ash monitoring points. They are: MW-4, an upgradient well; MW-5 a downgradient well; R2A, a downgradient discharge; R2B, a downgradient discharge.

MW-4, located on the hilltop to the south of the site, is unaffected by mine drainage and by the Ebensburg Power operation. It does show chloride concentrations higher than what may be expected, as high as 175 mg/L. W-4 is

located within 500 feet of an interchange for US Route 219 and road salt may explain the relatively high chloride. It will not be discussed further.

MW-5 is located between the Revloc 2 site and the South Branch along the north-central edge of the pile. Background data show that it was affected by mine drainage from the pile prior to Ebensburg Power Company's operation. The data also show that water quality in MW-5 has improved to the point where it is now uncontaminated. The improvement at MW-5 began before Ebensburg Power Company had substantially affected the Revloc 2 site, so factors other than the re-mining and reclamation, possibly site preparation work and well construction and purging techniques, apparently have contributed to the improvement in the water quality at MW-5. The data for MW-5 show no trends in terms of any trace metals and Ca and Mg have decreased.

Discharge R2B is located directly downgradient of an area that has been re-mined and partially reclaimed. R2B was a low-flow, extremely poor acid mine drainage quality prior to Ebensburg Power Company's operations. As re-mining and reclamation of the area upgradient of R2B has progressed, the water quality at the point has improved. Acidity values have fallen from several hundred mg/L, and higher, to 0 in recent samples, while concentrations of metals typically found in mine drainage have also declined. The pH has risen, while sulfate values have declined. Calcium, potassium, sodium, and chloride levels have increased. Among the toxic metals, copper, lead, and zinc concentrations appear to have declined, while selenium concentrations appear to have increased.

Discharge R2A is located along the northern edge of the Revloc 2 site, downgradient of an area where re-mining and reclamation are in progress. Background data show R2A to be of very poor AMD quality. While the discharge remains contaminated, the quality has improved in terms of AMD contamination over the past two years and flows have decreased. The quality at R2A is representative of very concentrated AMD leachate from the Revloc 2 pile. The concentrations of some trace metals in R2A are relatively high. For example, arsenic concentrations exceeded the MCL during every sampling event, except for June 10, 2003, and have been as high as 0.67 mg/L. Lead concentrations have typically exceeded the MCL, and chromium, copper, and cadmium have also done so on occasion; the results for each of these heavy metals are variable without any clear trends. The elevated toxic metal concentration in R2A existed prior to Ebensburg Power Company's operation, and illustrates that severe bituminous coal AMD, especially from coal refuse piles, can include significant toxic metal concentrations. Zinc concentrations at R2A have increased. Selenium concentrations at R2A have been highly variable, were less than detection limits during the background-sampling period, exceeded the MCL of 0.050 mg/L during much of the monitoring period, but returned to less than the MCL during 2002 and 2003. Table 3 presents a summary of the monitoring data on R2A and R2B for AMD parameters.

Table 3. Summary of flow and AMD pollutant load data from the Revloc 2 site discharges. Background data were collected from April 1996 through March 1997 (N = 12), and recent data were collected from June 2002 through June 2003 (N = 12). Flows are in gpm and loads are in lbs/day.

Parameter	Total	R2A	R2B
Background Flow	14.6	12.4	2.2
Recent Flow	6.14	6.1	0.04
Background Acid Load	298.2	281	17.2
Recent Acid Load	106	106	0
Background Iron Load	5.3	5	0.3
Recent Iron Load	0.05	0.05	0
Background Mn Load	7.2	6.5	0.7
Recent Mn Load	2.4	2.4	0
Background Al Load	47.7	45.9	1.8
Recent Al Load	16.6	16.6	0
Background SO4 Load	839	777	62
Recent SO4 Load	206	204	2.4

Sampling Point SP-1 is a stream monitoring point located on the South Branch below the Revloc 1 and Revloc 2 sites. This point is influenced by the direct discharges and groundwater baseflow from the piles into the stream. Table 4 compares the data in terms of median values collected from SP-1 prior to 1992 (N=14) to the 14 most recent samples at the time of this writing.

Table 4. Comparison of background median flow and mine drainage pollutant concentrations at SP-1, the monitoring point on the South Branch directly down stream of the Revloc 1 and 2 sites.

	Flow (gpm)	pH (su)	Acidity (mg/L)	Iron (mg/L)	Mn (mg/L)	Al (mg/L)	Sulfate (mg/L)
1988-91 Data	3261	4.30	134	1.61	1.03	21.0	191
2000-03 Data	2427	5.55	16	0.50	0.55	1.50	73

Figure 4 shows graphically the reductions in aluminum, acidity and sulfate at SP-1, when the background data medians are compared to the most recent data medians.

The data from SP-1 show the improvement to date in terms of mine drainage pollution in the South Branch that has resulted directly from the Ebsburg Power Company operations at the two Revloc sites. Note especially the reduction in aluminum and acidity concentrations along with the increase in pH. During times of low flow, the stream still experiences spikes in pollutant concentrations. However, if past trends continue, that condition should improve as re-mining and reclamation is completed.

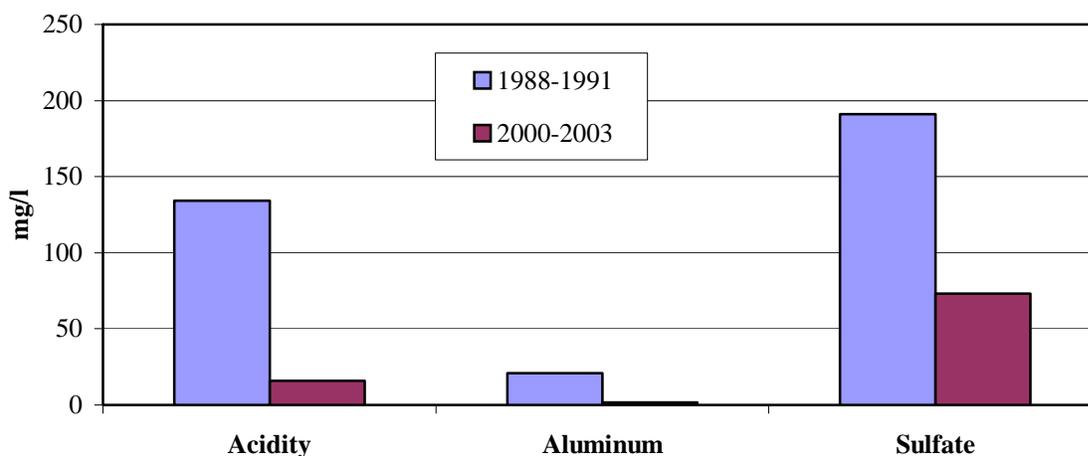


Figure 4. Comparison of background and recent median acidity, aluminum and sulfate concentrations at monitoring point SP-1, downstream of the Revloc sites.

While most toxic metal concentrations at the Revloc sites show no discernible change or some decrease, selenium concentrations have increased at some points. Selenium concentrations at well MW-1 have generally been higher than background, but typically remain below the MCL of 0.050 mg/L. The highest selenium values are at R2B and, especially R2A, seeps downgradient of the Revloc 2 site. Both of these points are low volume, and at times intermittent, with the median flow of the last 12 samples from R2B being 0.04 gpm and the median flow of the last 12 samples from R2A being 6.1 gpm. Areas upgradient of R2A and R2B were disturbed early in the mining on the Revloc 2 site, in part to accommodate extinguishing the on-site fire. Some areas where ash and refuse reject have been placed above those points remain exposed and unvegetated. It seems likely the elevated selenium in these points is related to the exposed ash/refuse mixture on the surface. Once reclamation of this area is completed, it is reasonable to expect that water quality at the two points will further improve, including a reduction in selenium concentrations. While selenium concentrations are clearly elevated at R2B and R2A, it is important to view these data in light of the low flows from these points; while concentrations are increased, the amounts of selenium discharging at these points are low, and the flows are decreasing as reclamation occurs in upgradient areas. Figures 5 and 6 show trace metal loading rates from seeps R2A and R2B respectively. The parameters displayed in these two

figures are the ones with the highest concentrations. The figures show that in terms of loading rates, most values have remained the same or decreased, with selenium being the lone exception. A sample collected by DEP on December 1, 2003 at SP-1, directly down stream from the Revloc sites, showed a selenium concentration below the detection limit of 0.007 mg/L; copper, chromium, arsenic, cadmium, mercury, lead, and nickel also were all below detection limits at SP-1 on that date. Another sample collected at SP-1 on 2/18/04 and analyzed for trace metals showed similar results.

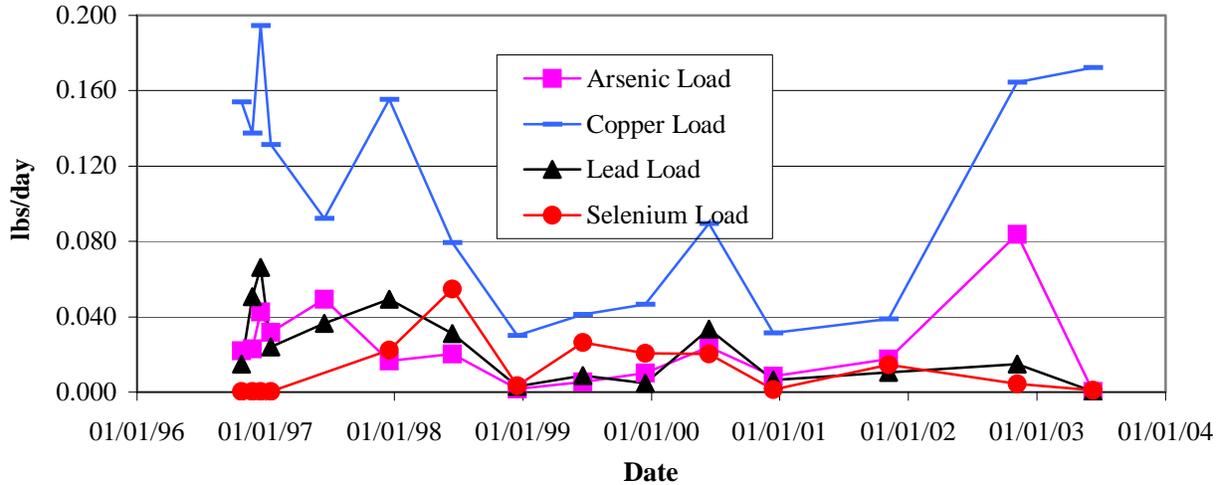


Figure 5. Trace metal loading rates in lbs/day at seep R2A.

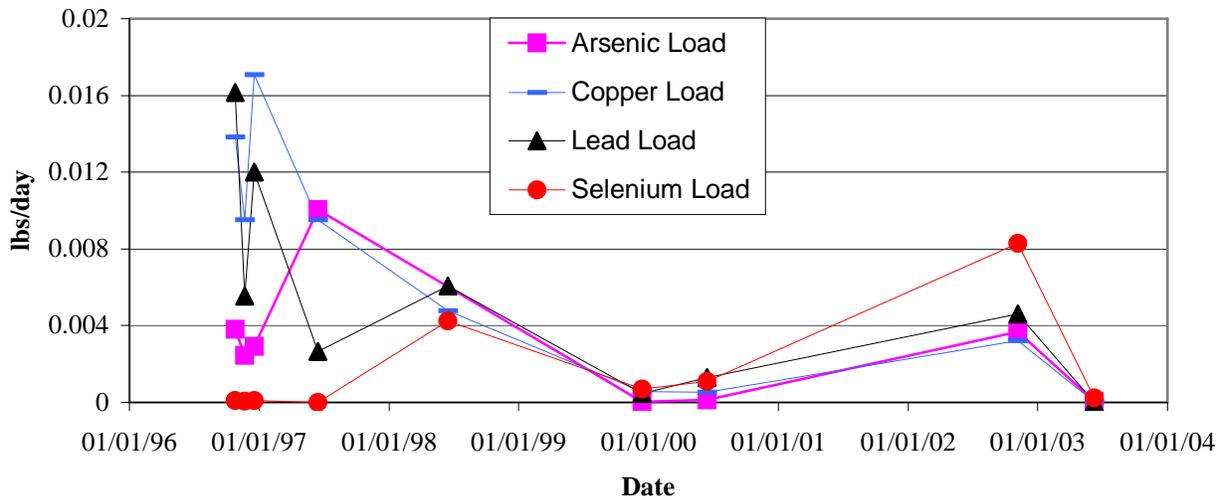


Figure 6. Trace metal loading rates in lbs/day at seep R2B.

Conclusions Regarding the Revloc Site

- Pollution loads have decreased at the 4 largest seeps from the Revloc 1 site (4SP, 4SPA, 4SPB, and 6SP); loading has increased somewhat at the smallest seep (23), but the net change has been a clear decrease in pollution loading from the site discharges.
- Groundwater quality at the downgradient well, MW-1, has improved; quality at the upgradient well MW-2 remains unchanged; quality at MW-3 also upgradient, but located at the base of the pile has been variable, and presently shows an influence from AMD.

- Groundwater at the downgradient monitoring well, W-5, at the Revloc 2 site has improved, but the timing of the changes suggest factors other than re-mining and reclamation are responsible. Quality at the upgradient well, W-4, remains unchanged.
- Based on data from seeps R2A and R2B, the abandoned coal refuse at the Revloc site leaches worst-case acid mine drainage that also includes metals such as arsenic, lead, and sometimes, chromium, cadmium, and copper in excess of the respective MCL's. This condition existed prior to the Ebensburg Power Company operation, and the abandoned coal refuse is the source for the elevated trace metals and severe AMD.
- Seep R2B has improved substantially in quality and declined in flow as a result of re-mining and reclamation on the Revloc 2 pile. Seep R2A has improved to a lesser degree, but has also decreased in flow as a result of re-mining and reclamation conducted upgradient of it. Reclamation is not completed up gradient of either of these two points.
- Taken collectively, the Revloc data indicate that selenium is one parameter for which concentrations may have increased as a result of the re-mining and reclamation with ash. Selenium at downgradient wells MW-1 and MW-5 is slightly elevated relative to background concentrations, although selenium has been below the MCL at MW-1 for the past three years and at MW-5 for the past two years. Selenium concentrations at R2B have been elevated relative to background and have consistently exceeded the MCL. Selenium concentrations at R2A have been highly variable, and for the past two years have been less than the MCL. R2A and R2B are both low volume seeps. A recent downstream sample at SP1 shows a selenium concentration of <0.007 mg/L. Selenium concentrations at R2A and R2B will likely decline with final reclamation of the up-gradient areas, and monitoring of these points will continue, and the seeps may eventually be entirely eliminated. Selenium concentrations appear to have peaked during the 1999-2000 time period, and have since declined.
- The major mechanism for decrease of mine drainage at Revloc 1 appears to be the removal of the marketable coal refuse and encapsulation of the remaining reject in the compacted FBC ash rather than the neutralization of mine drainage by alkaline leachate from the ash. This conclusion is supported by: 1) pH has remained low, while acidity and sulfate, as well as other mine drainage parameters, have declined at MW-1 and the larger of the pile discharges; 2) the flows of the primary discharge points at both the Revloc 1 and 2 sites have decreased, indicating that, as expected, the permeability of the areas reclaimed is less than that of the abandoned coal refuse.
- Re-mining and reclamation of the Revloc sites using coal refuse ash has substantially improved the quality of the South Branch Blacklick Creek by reducing the concentration of acid mine drainage parameters in the stream.

The Maple Coal Company Colver Site

Site Characterization/Setting

The Maple Coal Company Colver Site is located just to the southwest of the village of Colver where Elk Creek flows beneath State Route 4002. The main pile at the Colver site included approximately 4 million yds³ of refuse when the site was permitted (permit number 11900201) in January 1993 (Maple Coal Company, 1990). In 1997, Maple Coal Company permitted a second site (11970201), known as the Rail Yard Site, located adjacent to the southern edge of the Colver Site. The Rail Yard Site has yet to be activated, so will not be discussed in detail here.

Re-mining and reclamation of the Colver Site began in the spring of 1995, although some refuse from other locations was brought to the site prior to that and was stockpiled for future use. The refuse from the Colver site is the major fuel source for the adjacent Inter-Power/Ahlcon Partners, L.P. 110 megawatt FBC power plant. The plant is located at the northern end of the Colver Site. The plant consumes about 600,000 tons of waste coal per year, not all of it from the Colver Site.

Maple Coal has re-mined and reclaimed less than one-half of the Colver Site to date. According to figures reported to DEP, between 1998 and 2002 about 2.2 million tons of FBC ash had been returned to the site, or an average of about 432,000 tons per year.

Mining began on the northeastern end of the site and proceeded to the west along the northern edge. During 2002-2003, the company mined the western edge of the pile, and is now mining eastward through the central part of the pile. Operations on this site are conducted very much as was previously described for the Revloc sites. The final reclamation plan calls for either retrieving topsoil from beneath the refuse pile or, where not enough soil can be obtained, direct seeding onto an ash/reject mixture.

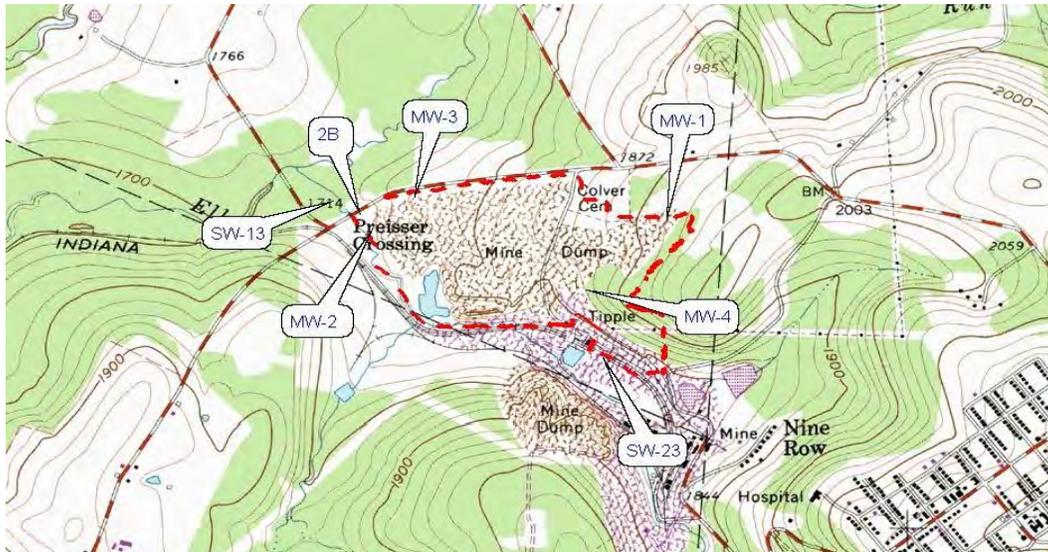


Figure 7. Map of the Colver site and associated monitoring points.

Monitoring Results to Date at the Colver Site

MW-2 is a downgradient well located at the southern tip of the site, between the site and Elk Creek. Areas immediately up gradient of the well have been re-mined and reclaimed, but most of the remaining areas of the unaffected refuse are in the potential recharge area for the well. Figure 8 displays the available data for iron, acidity and sulfate concentrations at MW-2. Note the downward trends in the three parameters, especially for acidity and iron during 2002-03, the time period when re-mining and reclamation took place in the area directly up gradient from MW-2. The background concentrations of these parameters in MW-2 were extremely high, demonstrating the severity of the AMD associated with the abandoned coal refuse.

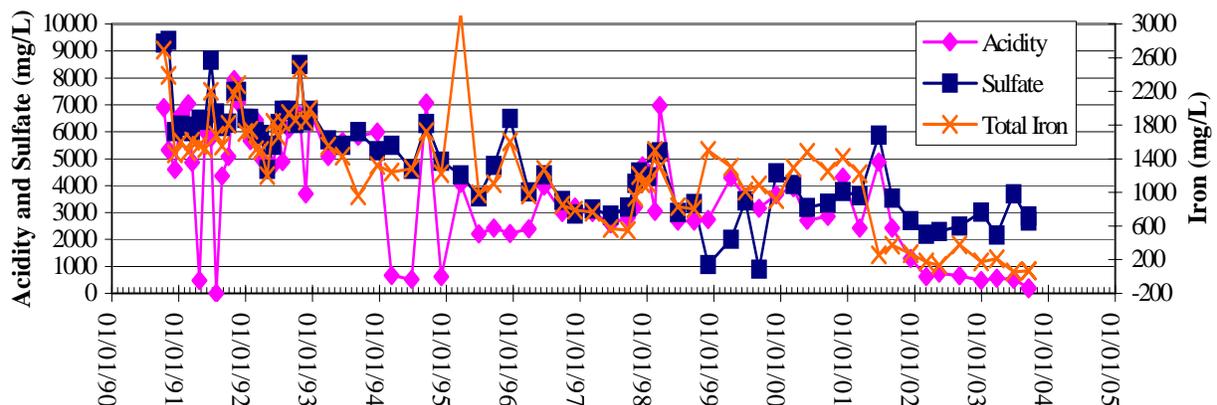


Figure 8. Iron, acidity and sulfate concentrations at Maple Coal downgradient well MW-2.

MW-3 is also a well located downgradient of the Colver site, along the southwestern side of the permit area. Figure 9 shows the available data for iron, acidity, and sulfate concentrations at MW-3. The data for MW-3 show similar trends to that seen for MW-2, although the magnitude of the change does not appear to be as great, with the possible exception of the iron concentrations.

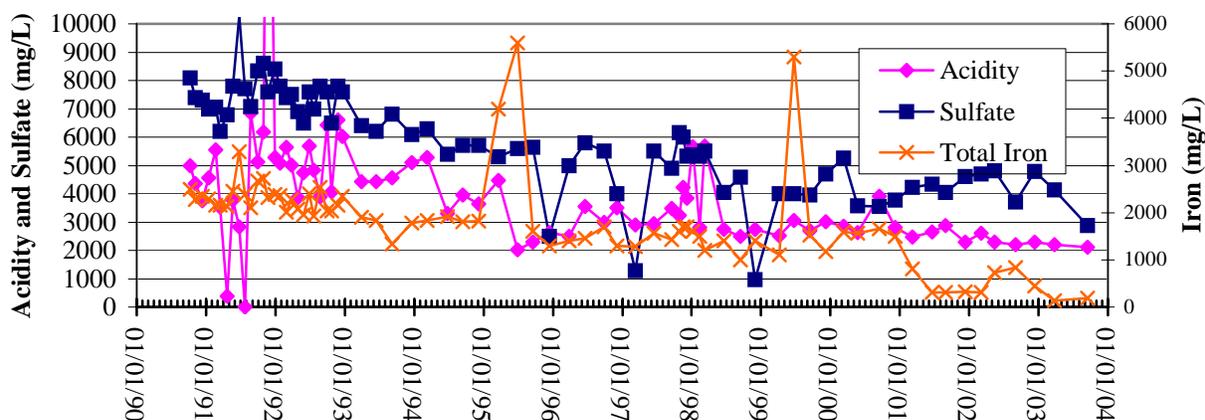


Figure 9. Acidity, sulfate and iron concentrations at downgradient well MW-3.

Table 5 displays trace metal concentrations in both wells MW-2 and MW-3. The numbers shown are median concentration values. In MW-2, most median trace metal concentrations have declined substantially after the placement of over 2.2 million tons of FBC ash directly upgradient. Note that trace metal concentrations for some parameters, especially in MW-2, were relatively high in the background, and in some cases were well in excess of the respective MCL's. It is clear from these data that the elevated trace metals were a result of the abandoned coal refuse. In MW-3, As, Cd, and Pb have decreased substantially, Cr, Cu and Se may have increased slightly, although the latter three all remain below their MCL's.

Table 5. Heavy metal concentrations at MW-1 and MW-3, Maple Coal. Median concentrations in mg/l; 1990-94 data compared to most-recent data.

MW-2	Al	As	Cd	Cr	Cu	Pb	Hg	Se	Zn
1990-94	430	0.930	0.060	0.320	1.020	0.170	*	0.003	2.17
1999-03	151	0.395	0.006	0.028	0.067	0.009	*	0.024	0.09
MW-3	Al	As	Cd	Cr	Cu	Pb	Hg	Se	Zn
1990-94	71	0.080	0.090	0.030	0.010	0.150	*	*	2.75
1999-03	66	0.016	0.004	0.038	0.040	0.001	*	0.008	2.03

MW-1 and MW-4 are monitoring wells located upgradient of the Colver Site. Background groundwater quality for the wells is alkaline with relatively low metals and other dissolved constituents. While wells 1 and 4 are upgradient of the Colver re-mining site, they are downgradient of the power plant and its associated fuel and ash handling areas. Both wells remain net alkaline with low metal concentrations, but sulfate levels have increased in each of them, to approximately 100 mg/l in well 4 and 200 mg/l in well 1.

Two polluting point source discharges were identified from the Colver Site prior to the Maple Coal re-mining operation, and these discharges have been monitored throughout the operation. The two discharges are known as 2B and 23. A portion of the water from 2B has been encountered and re-routed by the operation, so it now discharges at three separate points that are considered a single hydrologic unit in the permit for purposes of comparing current pollution loads to baseline. Table 6 presents a comparison of baseline data to past water year (September 30, 2002-October 31, 2003) for the two site discharges. There was an unusual amount of baseline data available in this case, N=110 for 2B and N=120 for 23. Twenty-four samples were collected from the points during the past water year.

2B was by the worst of the two points, and pollution loading rates have declined substantially since re-mining began. For instance the acid load from 2B has declined from 1305 lbs/day to 396 lbs/day and the iron load has declined from 682 lbs/day to 38 lbs/day. Discharge 23 has also improved, despite its flow having increased

substantially. The increased flow at 23 is most likely due to re-routing of water because of reconfiguration at the site.

Table 6. Comparison of background load data to recent load data from the two polluting point-source discharges emanating from the Maple Coal Colver Site.

Parameter	Total	2B Combo	23
Background Flow	79.3	30.8	48.4
Recent Flow	139	16	123
Background Acid Load	1323	1305	17.9
Recent Acid Load	396	396	0
Background Iron Load	683.3	682	1.3
Recent Iron Load	38.1	37.6	0.58
Background Mn Load	4.4	4.0	0.46
Recent Mn Load	1.49	.96	0.52
Background Al Load	98.1	97	1.1
Recent Al Load	45.9	45.1	0.8

Table 7 summarizes baseline data and recent pollutant concentration data from the downstream monitoring point on Elk Creek, SW-13. Reliable flow data are not available from this point. There have been large decreases in iron concentrations at the downstream monitoring point. Recall that iron had also decreased substantially in the down gradient monitoring wells and discharges. Aluminum and manganese concentrations have also improved in the stream. However, the reduction in acidity and increase in pH at SW-13 is modest, especially when compared to the large decline in acidity in the wells and discharges. There are additional sources of pollution to Elk Creek, not associated with the Colver site, that are contributing to the pollution at SW-13. For example, there is a discharge 4A from the adjacent Rail Yard Site that has yet to be affected by re-mining. Also, there is a tributary coming in from the south that is AMD contaminated, and another significant area of abandoned mine refuse.

Table 7. Comparison of median pollutant values at the Colver Site downstream monitoring point SW-13, baseline and recent data. N=15 for each data set.

	pH (su)	Acidity (mg/L)	Iron (mg/L)	Mn (mg/L)	Al (mg/L)
1989-94 Data	3.1	361	92.7	1.3	36.1
2000-03 Data	3.4	310	18.5	0.4	23.9
% Reduction		14	80	69	34

Conclusions Regarding the Colver Site

- When background data are compared to recent data, mine drainage pollution concentrations have decreased in downgradient monitoring wells MW-2 and MW-3.
- Concentrations of most trace metals have declined in MW-2 and MW-3.
- High concentrations of trace metals in the background data at MW-2 and MW-3, coupled with declines in trace metal concentrations after re-mining and coal ash placement commenced, show that the abandoned coal refuse is the source of trace metals at the site.
- Selenium concentrations in the monitoring wells have at times been elevated, but the median concentrations remain below the MCL.
- There have been large reductions in pollution loads from the two point-source polluting discharges from the Colver Site, when recent data are compared to baseline data.
- Downstream monitoring point SW-13 has improved in terms of concentrations of mine drainage pollutants when recent data are compared to baseline data. Iron has decreased the most, while the decrease in acidity is modest.

- The Colver Site is less than one-half completed; if current trends continue, additional improvement in water quality should occur as the site is completed.

Conclusion

While abandoned coal refuse piles represent a relatively small part of Pennsylvania's abandoned mine lands in terms of acres, they are a significant part of the AML problem. They are especially expensive to reclaim, and standard reclamation techniques do little to address the water quality problems the piles create. AMD from abandoned refuse piles is often quite severe and can include elevated levels of toxic trace metals in addition to mine drainage parameters.

The Blacklick Creek watershed in southwestern Pennsylvania is an example of a watershed that has been significantly impacted by AMD, some of it from abandoned refuse piles. A planned restoration of the watershed cannot occur without abatement of the AMD problem from these piles.

Re-mining of abandoned bituminous coal refuse piles, using the recoverable waste coal as fuel, and returning alkaline FBC ash to the sites to help reclaim the piles provides an alternative to traditional AML clean-up of the piles, using only private funding and preserving limited AML funds for the cleanup of other sites.

Two operations, the Ebensburg Power Company Revloc sites and the Maple Coal Company Colver site, have been re-mining and reclaiming large refuse piles in headwater areas of Blacklick Creek for several years. The Revloc sites are nearing completion, and monitoring data show substantial reductions in AMD pollutants leaving the sites as well as substantial improvements in stream quality. The Colver site is only about one-half completed to date, but water quality data thus far is encouraging with reductions in pollution loading from site discharges and improvement in stream quality. Background monitoring documented elevated levels of some trace metals emanating from both the Revloc and Colver piles prior to re-mining and reclamation with ash. Most trace metal concentrations, with the exception of selenium, have either remained unchanged or have declined during the monitoring period. If current trends continue, further substantial reductions in pollution loads from the sites can be anticipated as re-mining and reclamation is completed.

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- Ebensburg Power Company, Inc., 1988. Application to Pennsylvania Department of Environmental Protection for surface mining permit number 11880102, Ebensburg, PA.
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- Lighty, Robin, 2004, Personal Communication.
- Maple Coal Company, Inc., 1990, Application to Pennsylvania Department of Environmental Protection for surface mining permit number 11900201, Ebensburg, PA.

Tim Kania is the Permits Chief at the Cambria District Mining Office of the Pennsylvania Department of Environmental Protection (DEP). He has held that position for about 5 years. Prior to that Mr. Kania, who is a registered professional geologist, worked for 17 years as a hydrogeologist for DEP. In this capacity, he conducted numerous investigations into mine-drainage related problems, and reviewed many coal and industrial mineral permit applications, with an emphasis on predicting water quality and quantity impacts from the operations. Mr. Kania is a member of the Blacklick Creek Watershed Association. He holds a B.A. in journalism and a B.S. in earth sciences, both from Penn State University.

2004 UPDATE TO OCCURRENCE AND FATE OF SELECTED TRACE ELEMENTS IN CIRCULATING FLUIDIZED BED COMBUSTION BYPRODUCTS

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Abstract

Until 2000, a preponderance of the CFB ash placed in Pennsylvania had been generated at 13 plants that burn coal refuse as fuel in combination with limestone. During that year, the entire database of ash-testing at 12 of these plants was examined. In addition, sampling for independent laboratory testing was conducted at two of the plants as well as their ash-placement sites. Finally, the entire water monitoring database was examined at the 14 major ash-placement sites that constitute the bulk of the ash beneficial usage by these 12 plants. A total of 854 water samples were analyzed that were collected through 1999, over a period of 13 years at 66 monitoring points, located on or immediately adjacent to the ash placement permit areas. Three of the placement sites studied are alleged by the Clean Air Task Force to be damage cases. The study concluded that both laboratory and empirical field data clearly demonstrated that hazardous/toxic trace elements are strongly bound in the ash and are not available to the ground and surface water associated with the placement sites examined. This report updates the water quality testing conducted since 1999, which supports the conclusions reached in the 2000 study.

Introduction

Until 2000 a preponderance of the CFB ash placed in Pennsylvania had been generated at 13 plants that burn coal refuse as fuel in combination with limestone. During that year the entire database (1986 – 1999) of ash testing at 12 of these plants was examined. In addition, sampling for independent laboratory testing was conducted at two of the plants as well as their ash-placement sites. This testing, also featured in “Chemical Mobility of Selected Trace Elements in Ash from Plants Burning Coal Refuse by Circulating Fluidized Bed Combustion”, (Rose, Voigt, and Noll, 2000) exposed samples of ash, collected by this author, to sequential leaching with water and $MgCl_2$ solution and sulfuric acid at pH 2 and 3. Also performed were TCLP and SPLP leaching tests.

The findings led to the conclusion that only trace amounts of the subject trace elements were released and the concentrations of these elements in the leachate all met RCRA standards for non-hazardous waste as well as the 10X drinking water MCL standard the EPA uses as a screening test for potential hazards. In fact, nearly all the leaches met federal drinking water standards. Because the ettringite, CaO, anhydrite, and other calcium and sulfur-rich phases are easily leached by the water and $MgCl_2$ solution and the trace elements strongly resist dissolution, even when exposed to 1.6M nitric acid, another host must exist. X-ray diffraction studies of this ash revealed the presence of 10 to 30% aluminosilicate in the ash and, since the trace elements were found to be strongly bound, such a host would be capable of providing that type of bond. Other possible hosts are iron oxides and unburned coal particles.

To provide empirical evidence to support the laboratory findings the entire water monitoring database was examined at the 14 major ash-placement sites that constitute the bulk of the ash beneficial usage by these 12 plants. A total of 854 water samples were analyzed that were collected, through 1999, over a period of 13 years at 66 monitoring points, located on or immediately adjacent to the ash placement permit areas. The collection of these studies was memorialized in the report, titled “Occurrence and Fate of Selected Trace Elements in Circulating Fluidized Bed Combustion Byproducts” (Earthtech, Inc. and Arthur W. Rose, 2000). The study, submitted to the US EPA in September 2000, concluded that both laboratory and empirical field data clearly demonstrated that hazardous/toxic trace elements are strongly bound in the ash and are not available to the ground and surface water associated with

the placement sites examined. This report updates and expands upon the water quality analysis conducted for the 2000 study.

Discussion

As per the introduction the study of the water samples containing trace element analyses involved the examination of the results of 854 water-sampling events that occurred from 1986 to 1999. These sampling events took place at 66 monitoring points, located on or immediately adjacent to the ash placement sites. The availability of this data was due to the monitoring requirements of Module 25, a condition of one of the permits that provide regulatory environmental controls at the ash placement sites.

The Module 25 data, provided for the original study, included testing results for parameters other than the featured trace elements. These parameters, such as, but not limited to acidity, iron, manganese, sulfates, aluminum and other inorganic and organic parameters, are required to be tested quarterly in most cases and more frequently in some cases, such as during background sampling for Subchapter F protection. At the time of the original study analysis and discussion of these results was not possible in the time frame involved. However, due to the environmental scrutiny involving coal combustion byproducts, an examination of those results as well as the additional trace element and inorganic parameter results submitted to the PA DEP since 1999 has been authorized by those CFB ash generators placing ash on the sites examined. As a result a database for 17 CFB ash placement sites has been constructed, which includes results for selected trace and inorganic elements. This database can be compared to the original database in the following ways:

- A total of 17 CFB ash placement sites are now included, versus 14.
- Four (4) new sites have been added and one dropped, due to unavailability of data generated since 1999.
- Ten of the sites are located in the anthracite coal district and seven are located in the bituminous coal district versus seven in each district.
- Sampling dates from as early as 1986 to as late as 2003 are now included.
- A total of 4,728 sampling events are included versus 849
- Although the amount of placement sites increased the number of sampling points decreased from 66 to 55, due to the elimination of “up gradient” monitoring points.
- The CFB ash, placed at these 17 sites, constitutes at least 70% of the total amount of CFB ash generated during the period covered by the database.

Twelve of the 17 sites were visited by the author and the remaining five were evaluated through the examination of Module 25 maps, drill logs, well construction details, and interviews with company and DEP personnel. At the twelve sites visited all monitoring points were physically observed and compared to the data contained in the Module 25 and other permit submissions.

The sampling results were provided in numerous formats, including digital reports from the site operators and PA DEP as well as hard copies from both sources. Table 1 provides a breakdown of the sampling site types and analytical parameter totals.

TRACE ELEMENTS

SAMPLE ORIGIN	#SAMPS
WELLS	646
STREAMS	36
GROUND WATER DISCHARGES	68
MINE DISCHARGES	277
RUNOFF POND DISCHARGES	38
TOTAL SAMPLING EVENTS	1,065
SAMPLING EVENTS AT ANTHRACITE SITES	511
SAMPLING EVENTS AT BITUMINOUS SITES	554
NUMBER OF ANALYSES BY PARAMETER	
ARSENIC	1,056
CADMIUM	1,043

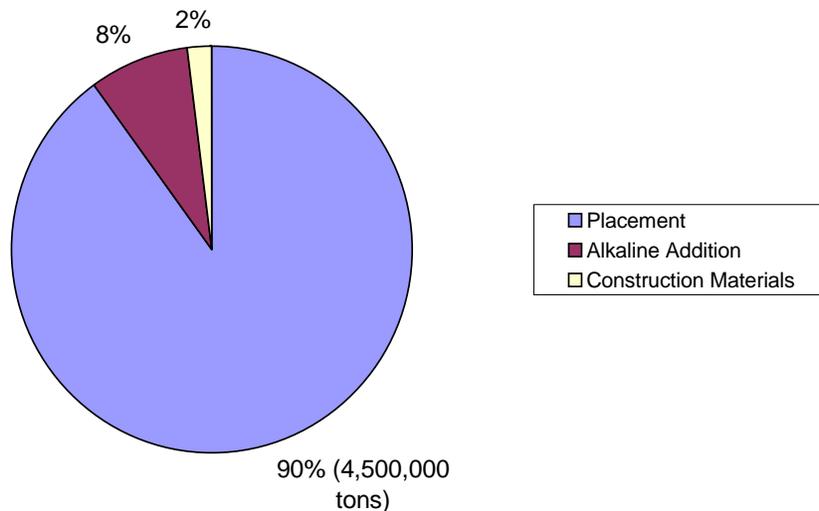
CHROMIUM	1,041
LEAD	1,041
MERCURY	1,060
NICKEL	785
SELENIUM	1,035
TOTAL NUMBER OF ANALYTICAL RESULTS REVIEWED	7,061

INORGANIC ELEMENTS

SAMPLE ORIGIN	#SAMPS
WELLS	1,037
STREAMS	180
GROUND WATER DISCHARGES	1,986
MINE DISCHARGES	203
RUNOFF POND DISCHARGES	253
TOTAL SAMPLING EVENTS	3,659
SAMPLING EVENTS AT ANTHRACITE SITES	915
SAMPLING EVENTS AT BITUMINOUS SITES	2,744
NUMBER OF ANALYSES BY PARAMETER	
ACIDITY	3,636
IRON	3,642
MANGANESE	3,639
SULFATES	3,634
ALUMINUM	2,625
TOTAL NUMBER OF ANALYTICAL RESULTS REVIEWED	17,176

With respect to the activity at the CFB ash placement sites, PA DEP's figures indicate that approximately 90% of the 5,000,000 tons of CFB ash generated is used for beneficiation at waste coal piles and abandoned pit backfilling. The remaining 10% is divided approximately 80/20 between alkaline addition for strip mine overburden/agricultural soil stabilization and building materials. (Personal communication: Al Dalberto, PA DEP, 2004). A breakdown of the sample analyses available for this study with respect to DEP's ash use figures certainly supports those use estimates.

PERCENTAGE OF YEARLY CFB ASH USAGE (5,000,000 TPY)



Methodology

Two databases were built, one for the trace elements arsenic, cadmium, chromium, lead, mercury, nickel, and selenium and the other for the inorganic parameters acidity, iron, manganese, sulfates, and aluminum. These particular inorganic parameters were selected because of their impact on aquatic life at a minimum and because all other inorganic parameters tested were either inconsequential or not tested often enough to be representative of the entire region examined. An example of an inconsequential parameter in this case is pH because of the fact that acidity has been included. Another example is total dissolved solids because of the fact that this measurement includes alkalinity and other “solids” not regarded as harmful unless at levels not normally found in this hydrologic setting. Total dissolved solids were also not tested often enough to warrant inclusion in this database.

Because of the intention to calculate medians for each parameter in the database the following manipulations were performed, prior to utilizing Microsoft Excel’s® median calculation formula:

- All values expressed as less than the detection limit were entered at 50% of the detection limit
- All alpha entries, representing non-testing, were converted to blank cells
- All zero’s, representing values less than the detection limit, were converted to 50% of the detection limits being employed by the labs at the time. This practice was particularly common at DEP in the late-1980’s and early 1990’s and, as a result the following detection limits were assumed: arsenic <5ppb, cadmium <5 ppb, chromium <50 ppb, nickel <40 ppb, selenium <5 ppb, lead <100 ppb, and mercury < 0.2 ppb. A zero for arsenic, cadmium or selenium was converted to 2.5 ppb and the other results were calculated accordingly. For the inorganic parameters zeros were converted to the detection limit appearing to be used at the time by the given laboratory and entered at 50% of that value. An exception to this was acidity in that a zero was entered when alkalinity was positive and acidity not entered or when acidity was expressed as a negative number. The following table lists the non-detects by parameter:

ANALYTE	TOTAL ANALYSES	# NON-DETECTS
Arsenic	1056	598
Cadmium	1043	623
Chromium	1041	680
Lead	1041	466
Mercury	1060	793
Nickel	785	193
Selenium	1035	619
Acidity	3636	525
ANALYTE	TOTAL ANALYSES	# NON-DETECTS
Iron	3642	70
Manganese	3639	93
Sulfates	3634	9
Aluminum	2625	288

- When an attempt to sample a point was not possible because it had dried up, due to ash placement, and the sampler had visited the point to confirm the status a zero was entered for each parameter. This was done for 136 of the 3,659 inorganic sampling events and 8 out of the 1065 trace element sampling events

Because of the large number of non-detects, which must be counted to maintain an accurate representation of the water quality at these sites, the results would necessarily be heavily skewed toward those values and may be bi-modal as well. For these reasons the median of the concentrations was used as opposed to the mean or mode as the most reasonable approximation of each datum population.

For each monitoring point the sample result dates were classified according to the up gradient activity. Six activity periods were identified and assigned as follows:

1. Waste coal piles not yet disturbed by mining or ash beneficiation

2. Waste coal piles actively being mined with no ash haul-back
3. Waste coal piles actively being mined with ash haul-back
4. Post-ash beneficiation of waste coal piles
5. Abandoned mine pits prior to backfilling with ash
6. Post-ash reclamation of mine pits

The test results at a given monitoring point can be affected over time by up to four of these activities e.g., #1, #2, #3, and #4. The dates for the activity periods assigned to the test results were generally the same as those for the initiation of the actual activity i.e., unless there was obvious evidence, no travel time calculations were employed to allow for delays between the date of the new activity and the date water from the location of that activity site would report to the monitoring point.

Findings

The database was used to build a series of charts that for all 17 sites as a whole compare the median concentrations for all of the sampling events grouped in the six activity periods described under Methodology. A separate chart for each element or parameter was utilized so that for example the median of the total arsenic concentrations for each activity period can be easily compared. The same follows for the additional 11 elements or parameters. Figures 1, 3, 5, 7, 9, 11, 13, and 15 through 19 express the results.

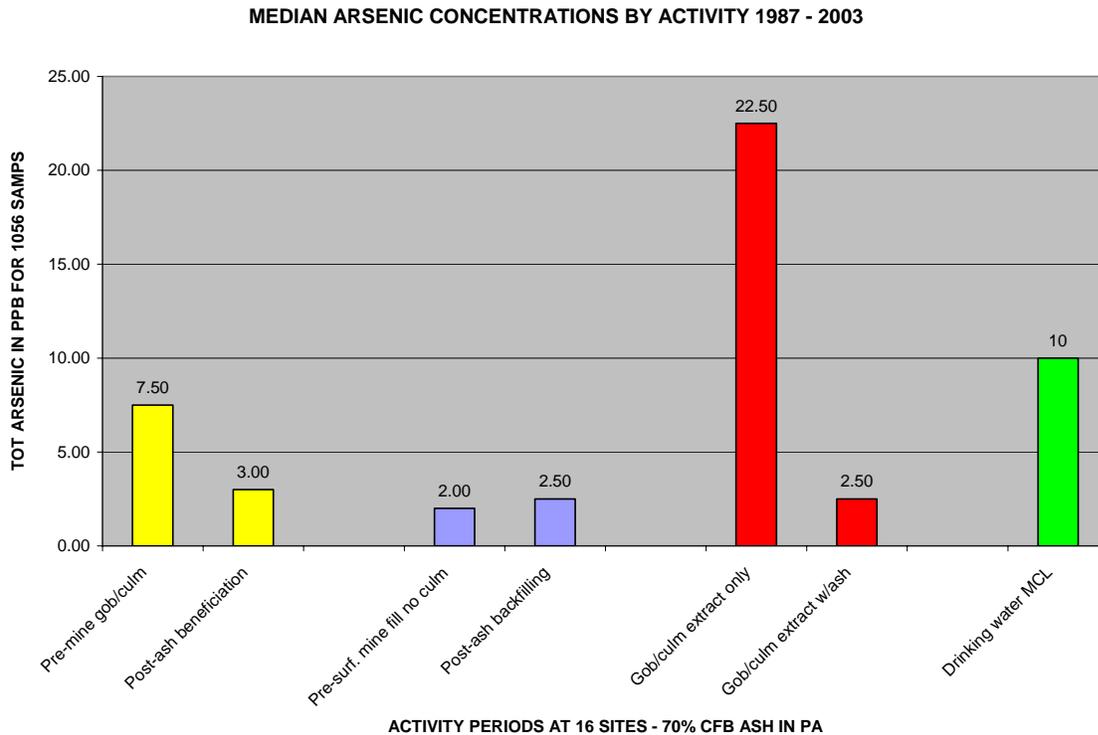


Figure 1.

MEDIAN CADMIUM CONCENTRATIONS BY ACTIVITY 1987 - 2003

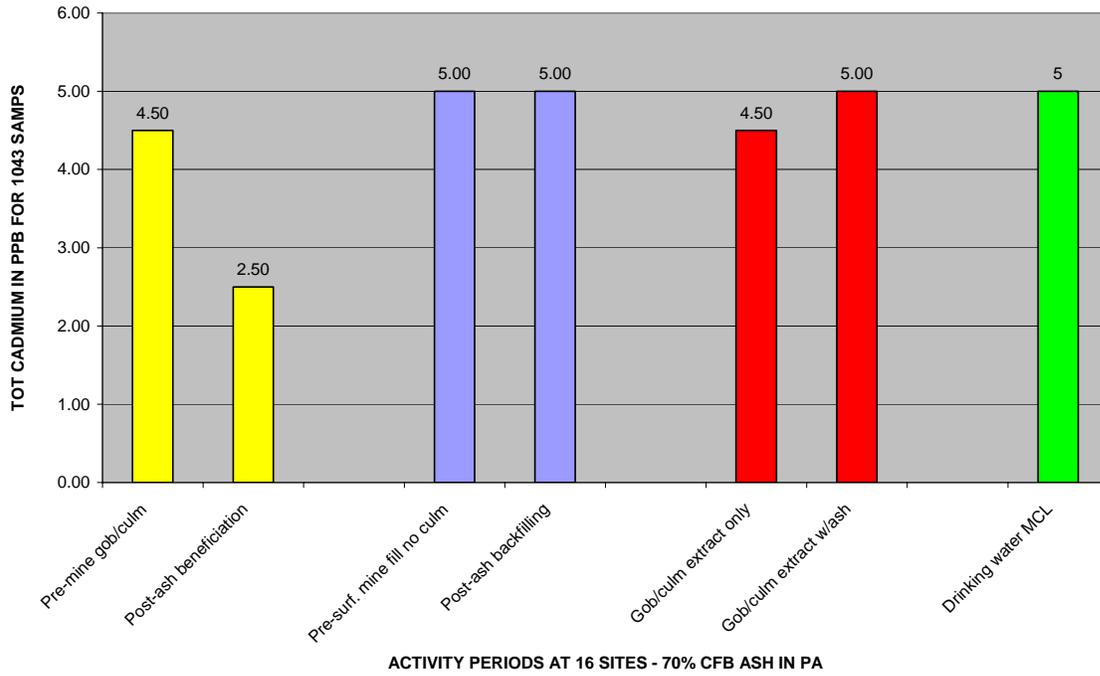


Figure 3.

MEDIAN CHROMIUM CONCENTRATIONS BY ACTIVITY 1987 - 2003

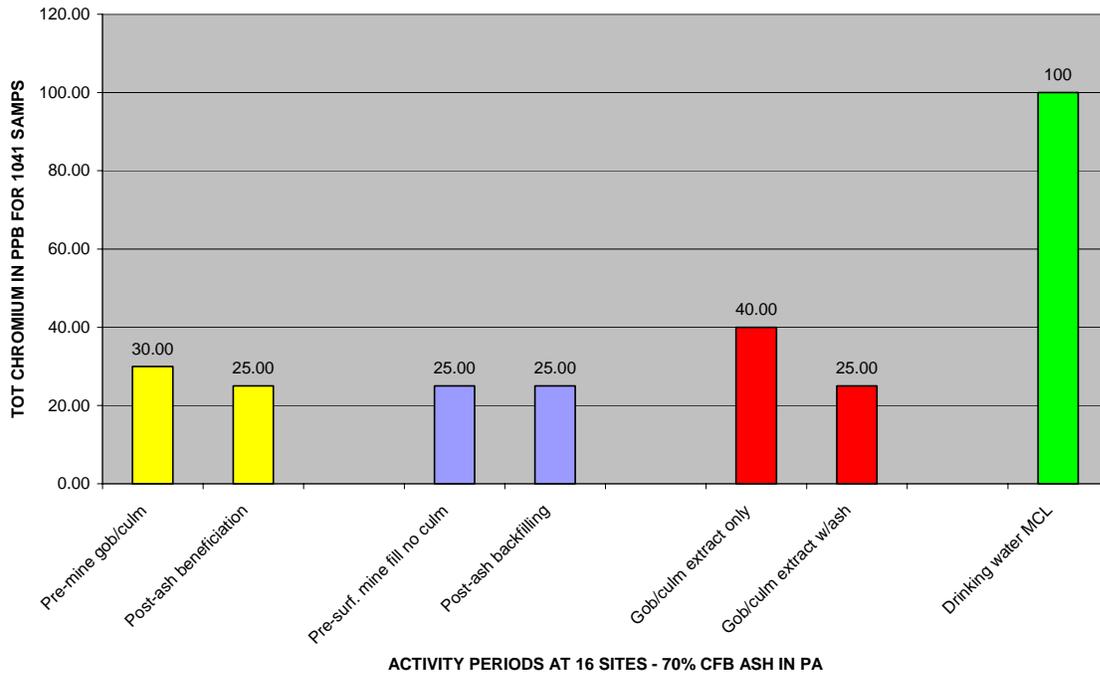


Figure 5.

MEDIAN LEAD CONCENTRATIONS BY ACTIVITY 1987 - 2003

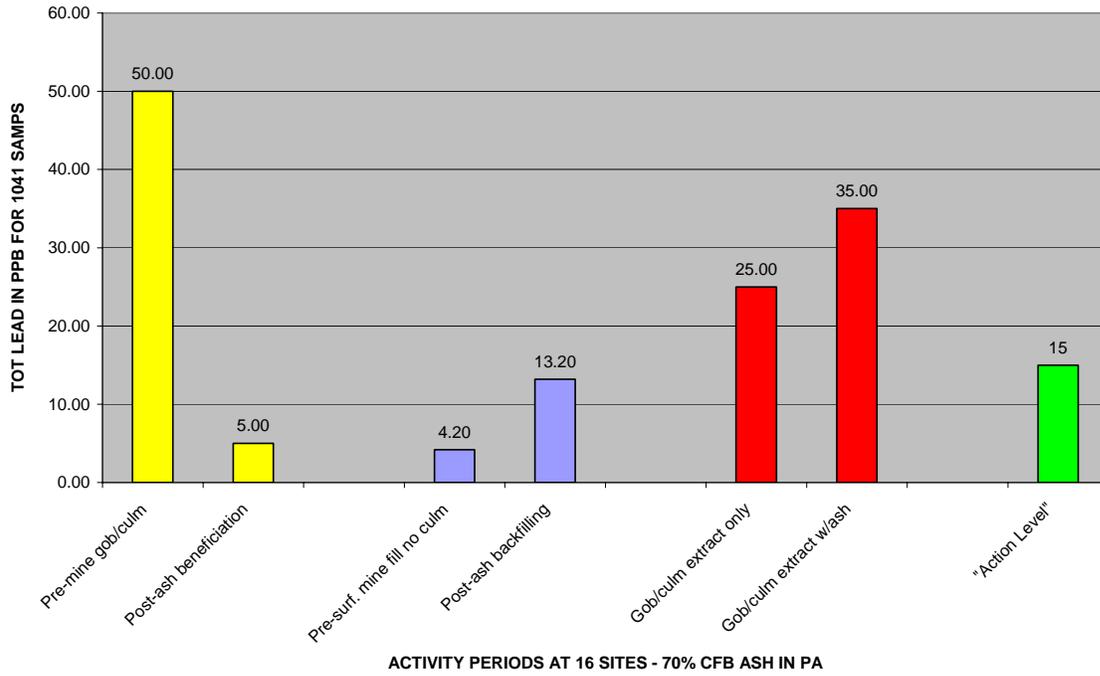


Figure 7.

MEDIAN MERCURY CONCENTRATIONS BY ACTIVITY 1987 - 2003

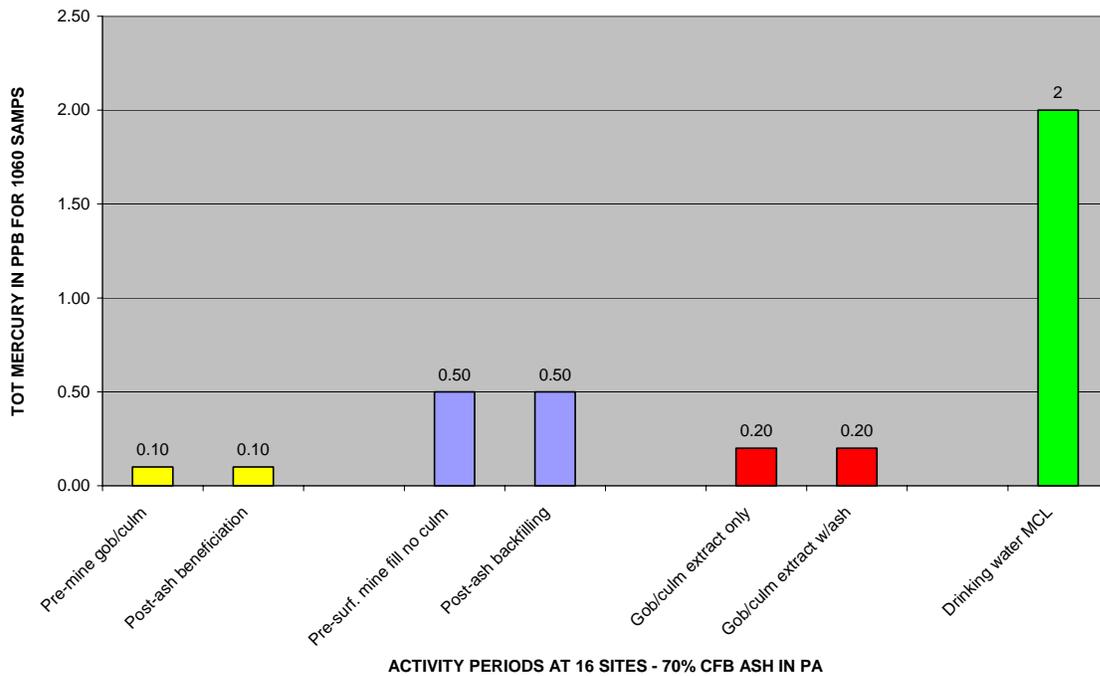


Figure 9.

MEDIAN NICKEL CONCENTRATIONS BY ACTIVITY 1987 - 2003

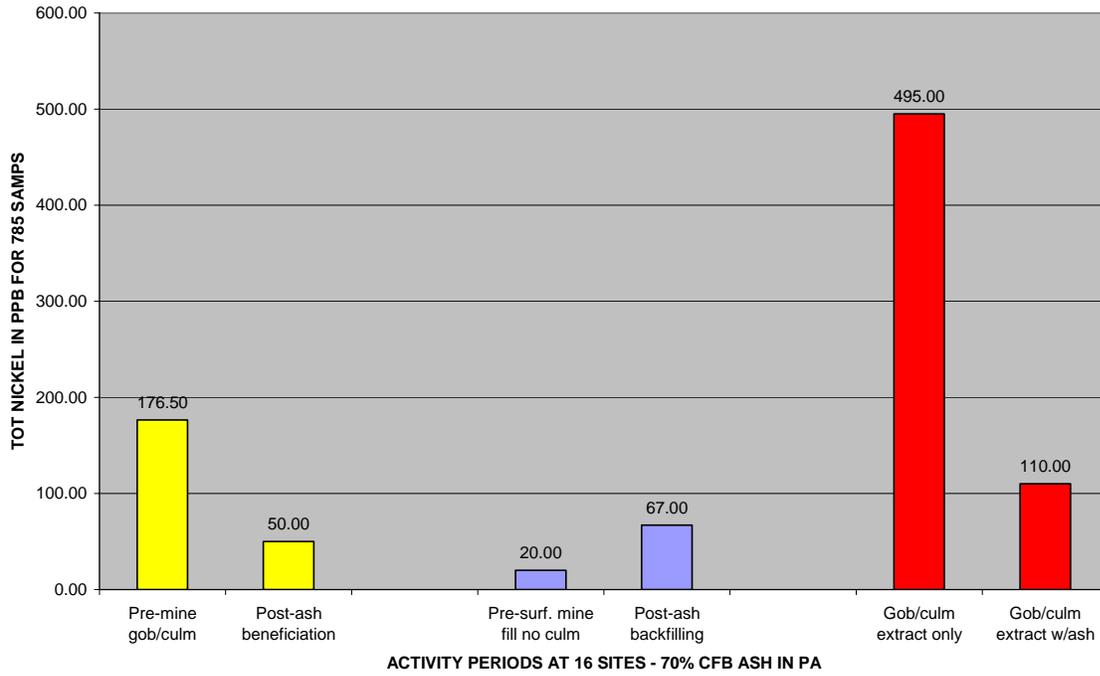


Figure 11.

MEDIAN SELENIUM CONCENTRATIONS BY ACTIVITY 1987 - 2003

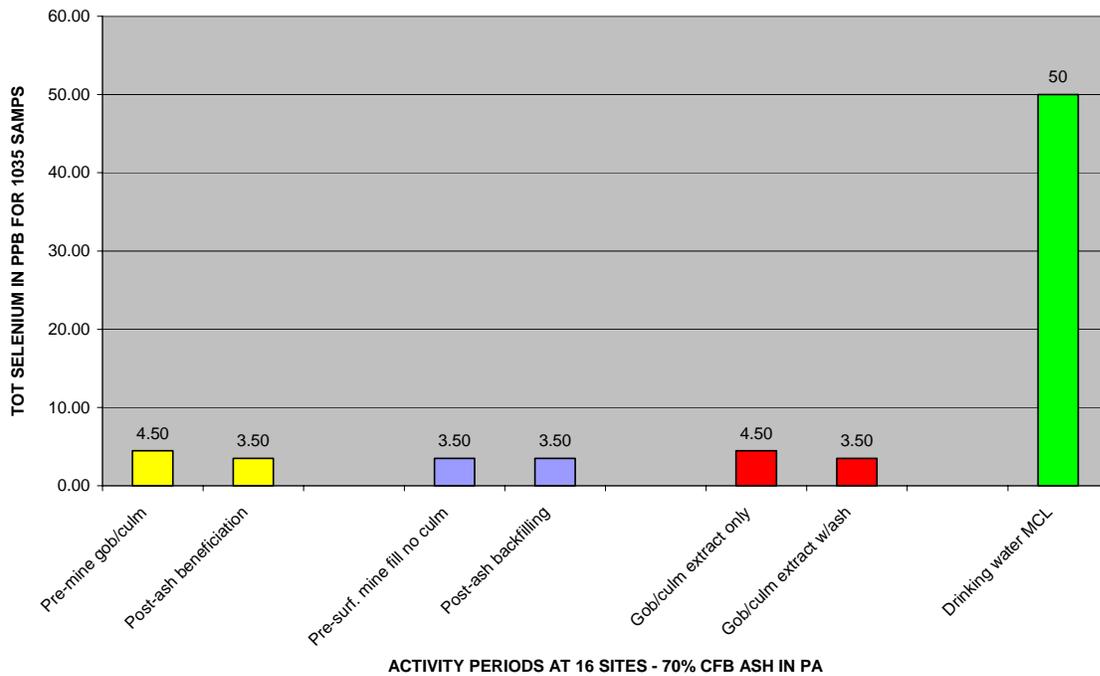


Figure 13.

MEDIAN ACIDITY CONC. BY ACTIVITY 1986 - 2004 (3636 SAMPLES)

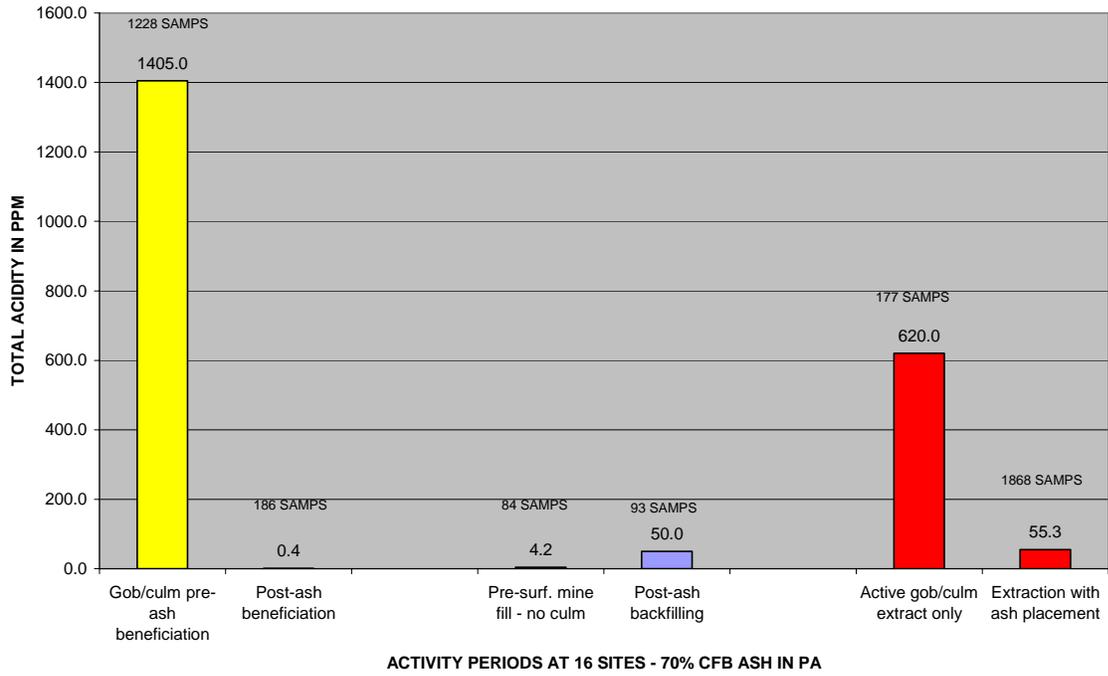


Figure 15.

MEDIAN IRON CONC. BY ACTIVITY 1986 - 2003 (3642 SAMPLES)

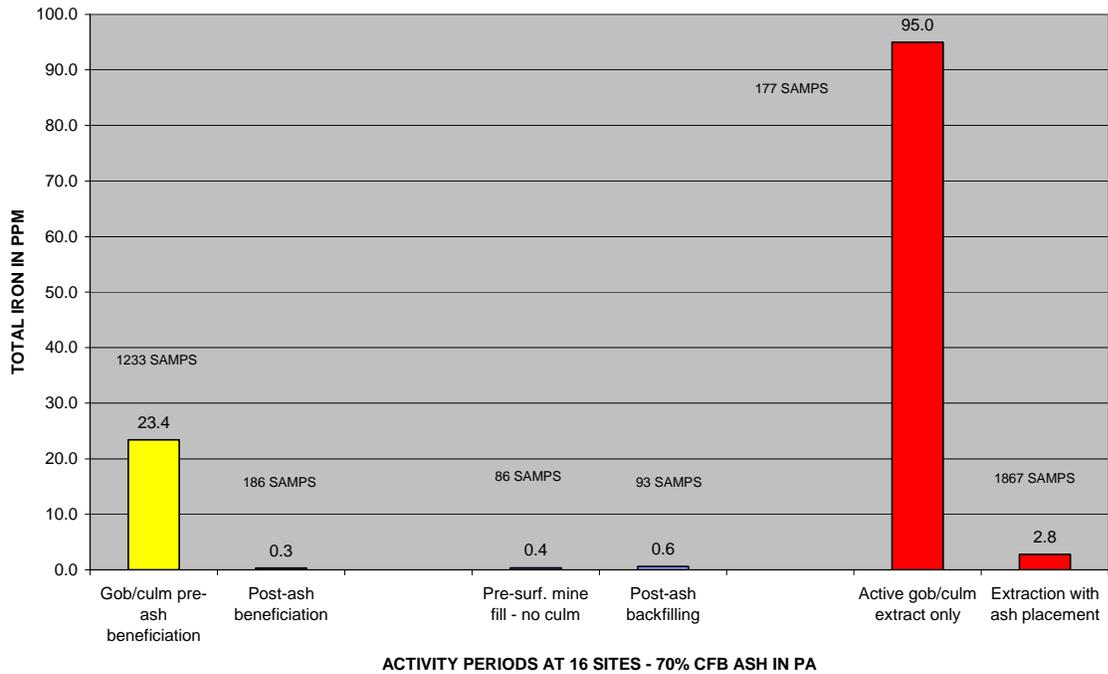


Figure 16.

MEDIAN MANGANESE CONC. BY ACTIVITY 1986 - 2004 (3639 SAMPLES)

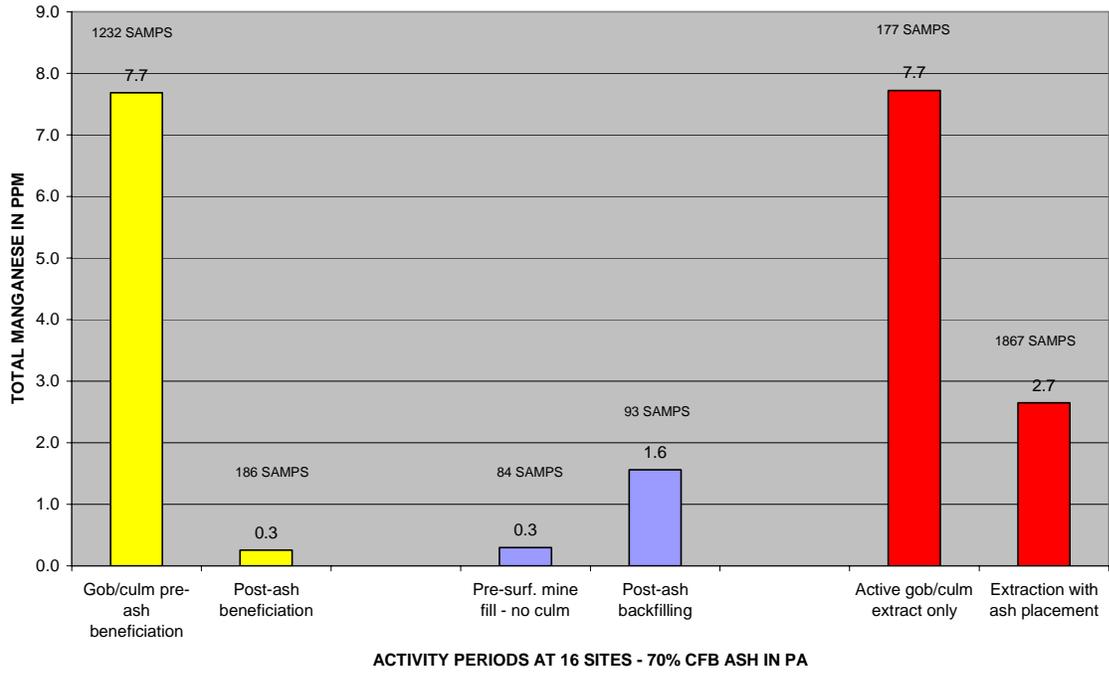


Figure 17.

MEDIAN SULFATE CONC. BY ACTIVITY 1986 - 2003 (3634 SAMPLES)

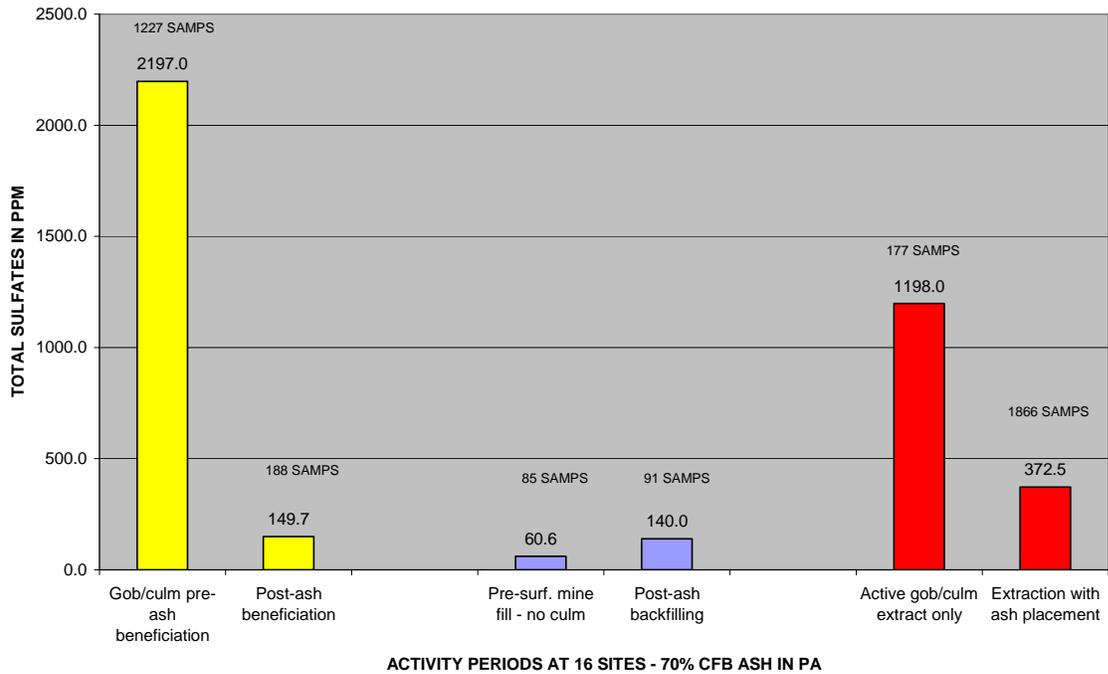


Figure 18.

MEDIAN ALUMINUM CONC. BY ACTIVITY 1986 - 2004 (2625 SAMPLES)

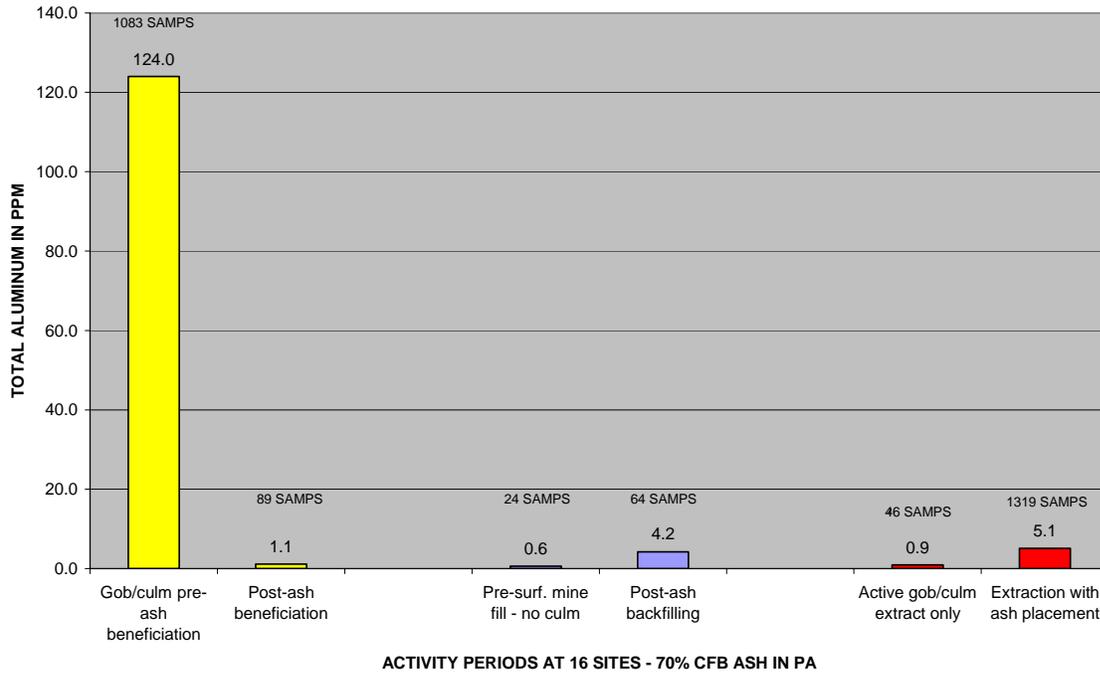


Figure 19.

In order to assure that comparison of the median values for all 17 sites as a whole does not mask problems at individual sites the actual values for each sampling event were reviewed by site and summarized with respect to occurrences when the concentration of the trace elements exceeded 10 times the “ground water parameter”. Please refer to figures 2, 4, 6, 8, 10, 12, and 14. This was not done for the inorganic parameters because in most cases there either is not a primary drinking water MCL established or, if it is, it is on the secondary list, related primarily to aesthetics. In Pennsylvania the “trigger levels” established for ash placement sites are based upon a concentration value of 25 times the “ground water parameter”. The ground water parameter (GWP) is generally the federal drinking water maximum contaminant level (MCL), established for public drinking water supplies. In the case of lead a term referred to as the “action level” is used because an MCL is established only for bottled water.

The use of 10X the ground water parameter in this study as an evaluation tool is derived from the modeling performed by EPA for monitoring sites located on or immediately adjacent to large landfills to indicate potentially hazardous conditions. Utilization of the 10X factor as does the 25X factor recognizes the dilution and attenuation affects of aquifers and soils located between the fill and any private or public water supply sources. The 25X factor in this case was not employed because occurrences of concentrations in excess of that standard are extremely rare for all of the elements studied.

The findings are expressed graphically on the following figures, described above.

ARSENIC >10X GWP OCCURRENCES BY ACTIVITY 1987 - 2003

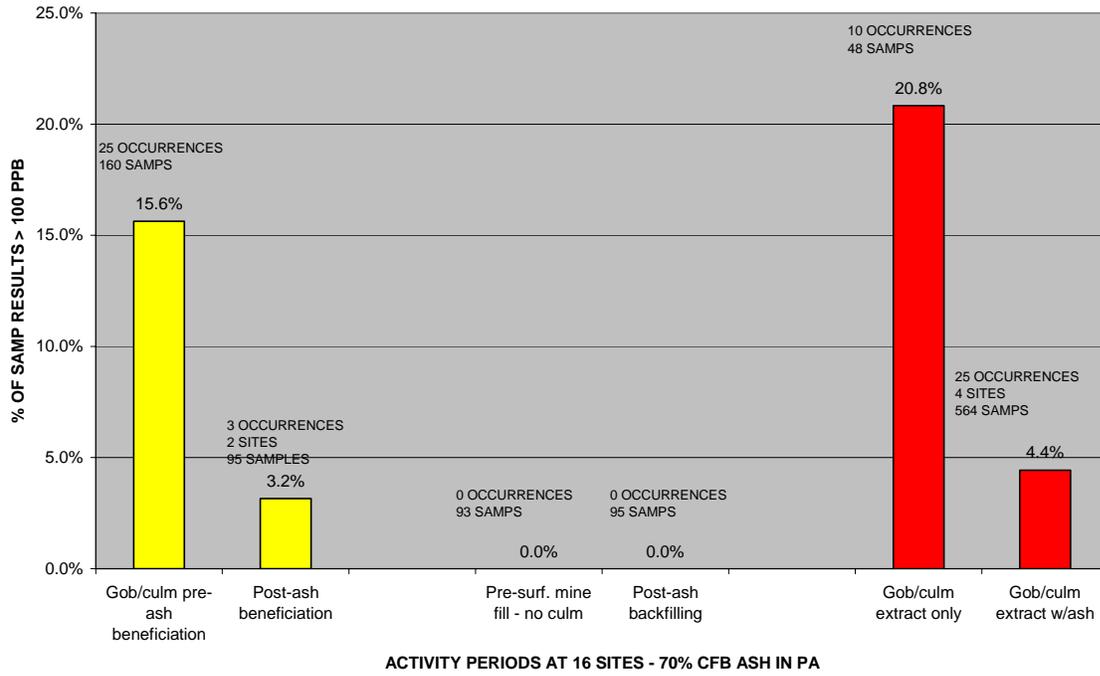


Figure 2.

CADMIUM >10X GWP OCCURRENCES BY ACTIVITY 1987 - 2003

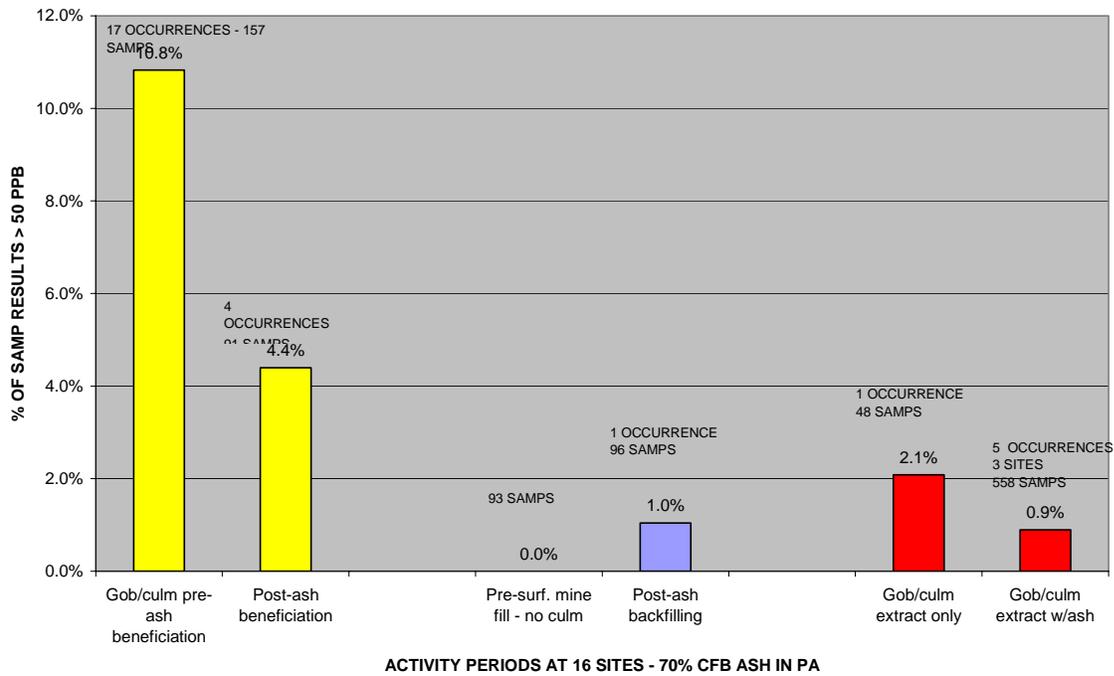


Figure 4.

CHROMIUM >10X GWP BY ACTIVITY 1987 - 2003

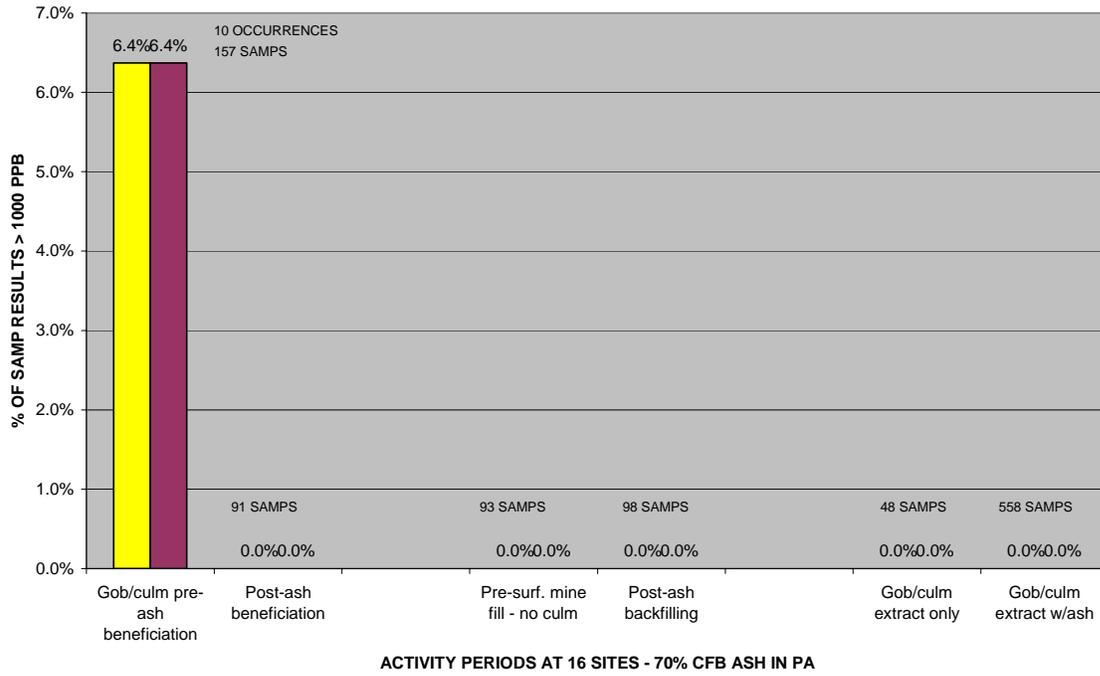


Figure 6.

LEAD >10X GWP OCCURRENCES BY ACTIVITY 1987 - 2003

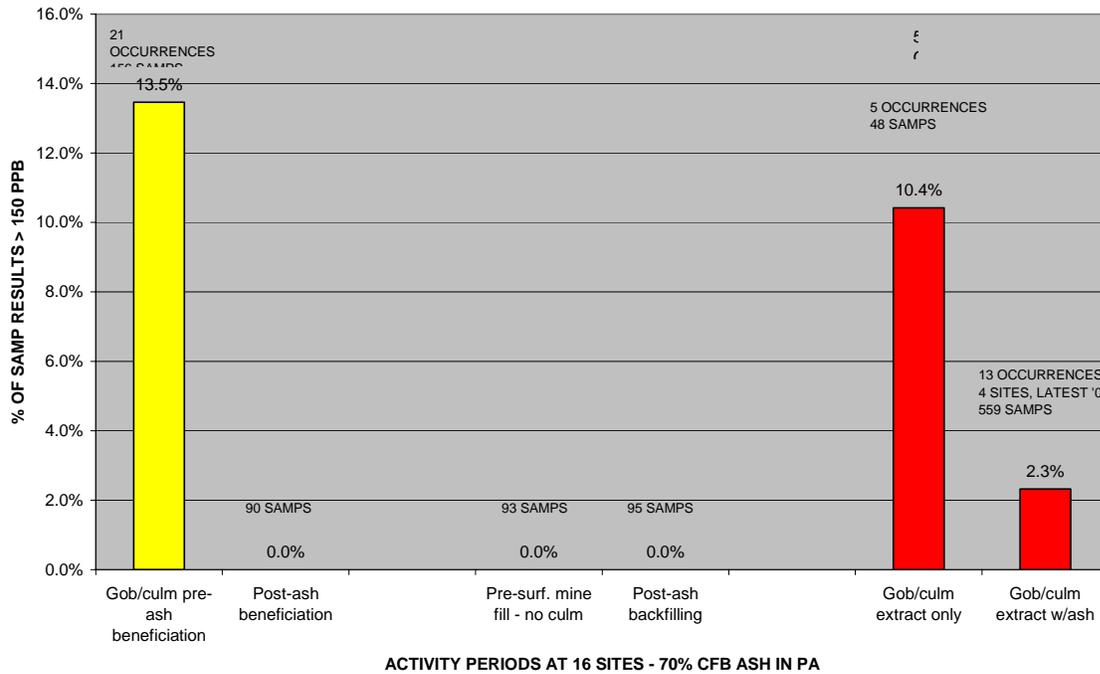


Figure 8.

MERCURY >10X GWP OCCURRENCES BY ACTIVITY 1987 - 2003

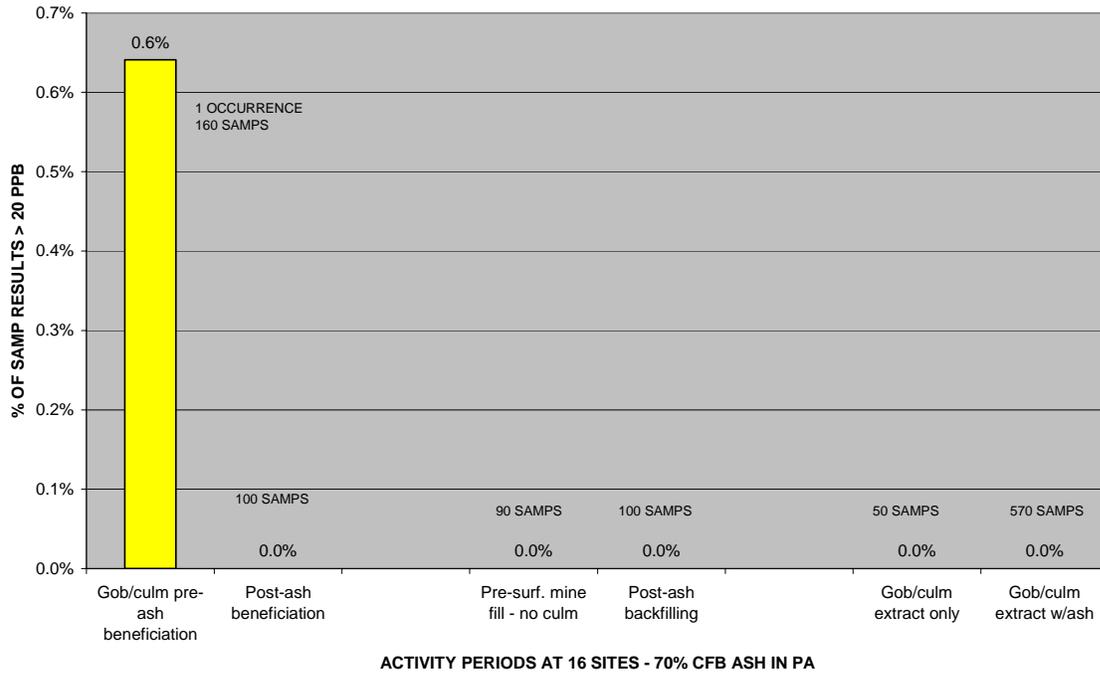


Figure 10.

NICKEL >10% GWP OCCURRENCES BY ACTIVITY 1987 - 2003

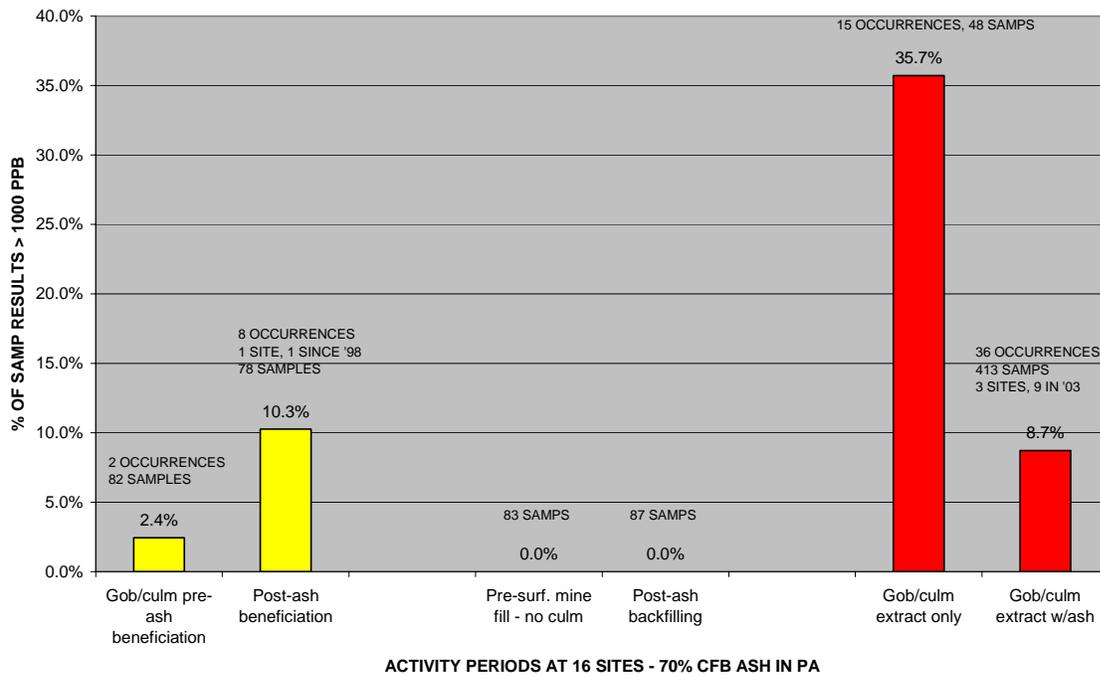


Figure 12.

SELENIUM >10X GWP OCCURRENCES BY ACTIVITY 1987 - 2003

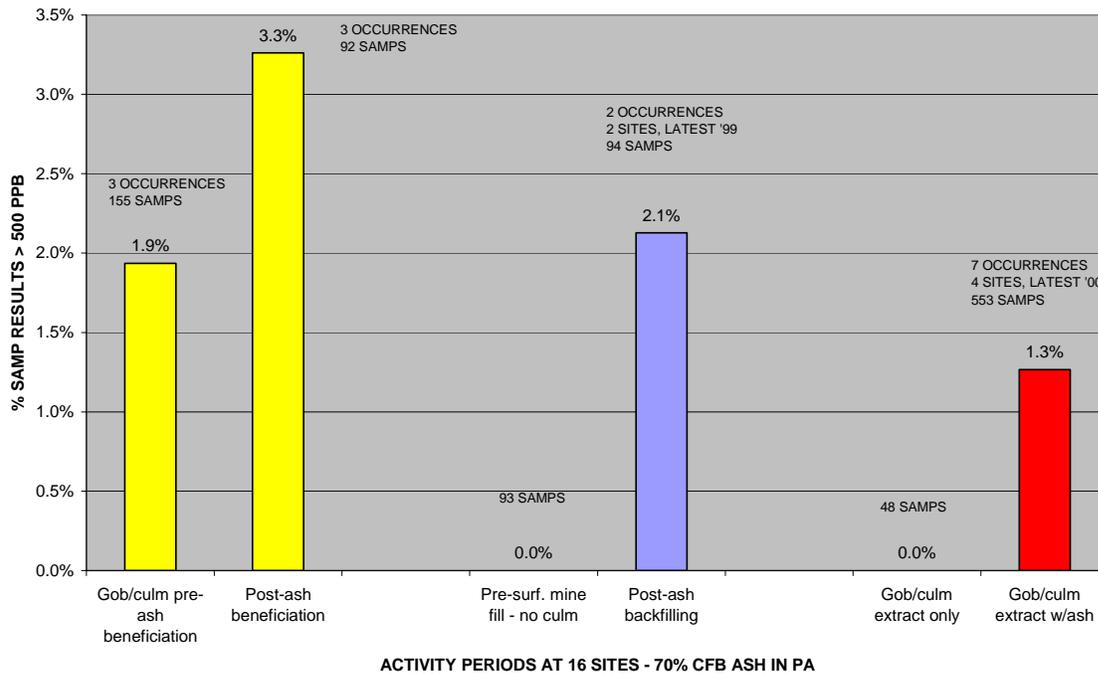


Figure 14.

Conclusions

1. For all of the trace elements and inorganic parameters studied the median concentrations for samples extracted from monitoring points located down gradient of waste coal piles unaffected by fuel extraction or ash beneficiation are significantly higher than those concentrations for samples extracted after ash beneficiation of waste coal piles is completed.
2. Except for cadmium and mercury the median concentrations for the rest of the parameters in samples extracted from monitoring points located down gradient of waste coal piles unaffected by fuel extraction or ash beneficiation are significantly higher than those concentrations for samples extracted during active waste coal mining with ash placement or haul-back. Cadmium and mercury are slightly higher for the active mining periods, but the median values for mercury are 10% of the drinking water MCL and there have been no values in 558 samples reported higher than 10X the GWP. The median value for cadmium during the active mining periods equals the drinking water MCL and there have been only five values in excess of 10X the GWP. These were out of 558 samples, distributed among three sites.
3. Except for cadmium, lead, and aluminum the median concentrations for the rest of the parameters in samples extracted during active waste coal mining with no ash-haul back are higher than those median concentrations for samples extracted during active waste coal mining with ash haul back. In the case of cadmium the difference is 0.5 ppb and the active mining with ash haul back concentration does not exceed the drinking water MCL. For lead the percentage of samples with occurrences higher than 10X the GWP is actually five time higher for active mining without ash haul back than with haul back and the occurrences during the ash haul back periods totaled 13, distributed among 4 sites and 558 samples with the latest occurrence taking place in 2002. The median concentration for aluminum is 5.9 ppm during active mining with ash haul back and 0.9 ppm during active mining without ash haul back.
4. For cadmium, chromium, mercury, and selenium the post mine void-filling median concentration results are less than or equal to pre-mine void filling median concentration results. The post-backfilling median for arsenic is slightly higher than the pre-fill scenario, but the result is 25% of the drinking water MCL and there have been no occurrences in either scenario that have exceeded 25X the GWP. Both nickel and lead median

concentrations are higher for the post-backfilling scenario, but neither scenario exceeds the drinking water MCL and neither scenario has experienced any occurrences in excess of 25X the GWP. The median concentration results for acidity, iron, manganese, sulfates, and aluminum are all higher for the post-backfill situations, but for sulfates, the one parameter with a secondary drinking water MCL related to human health, both pre- and post backfilling scenarios exhibit median concentrations significantly less than the drinking water MCL.

5. The total number of samples extracted for sites where backfilling of mine pits or voids takes place is 25% of the number of samples extracted for ash beneficiation of waste coal piles and 15% of the number of samples extracted during active waste coal mining.
6. Laboratory results, generated by 19 years of sample collection at 17 major CFB ash placement sites, indicate that for the trace elements and inorganic parameters studied the practice of CFB ash placement for beneficiation of waste coal piles and backfilling of mine voids is environmentally beneficial and safe.

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Dennis Noll is a Registered Professional Geologist in Pennsylvania and has practiced for a total of 35 years in the field of environmental geology. In 1980, he co-founded Earthtech, a firm largely dedicated to dealing with the environmental requirements of mining companies and energy producers. Since 1988, he has managed projects for Earthtech that involve the environmental aspects of coal combustion by-products. Most of these projects have dealt with those by-products generated by circulating fluidized bed boilers. In 2000, Earthtech was awarded a contract, managed by Noll, to produce a study for ARIPPA that characterized the occurrence and fate of trace elements contained in the CFB fly and bottom ash generated by all 12 of that organization’s member power producers in Pennsylvania. Since then, he has guided Earthtech’s environmental oversight work at the ash placement sites studied in that report.

Dr. Arthur W. Rose is Professor Emeritus of Geochemistry in the Department of Geosciences, Pennsylvania State University. He has experience in a wide range of problems involving geochemistry of natural and contaminated environments, especially on trace element behavior. Currently he continues research on passive treatment of acid mine drainage, and consults on problems related to coal, coal ash, landfills and groundwater. From 1964-67 he mapped geology and geochemistry for the State of Alaska before coming to Pennsylvania State University in 1967. At Penn State he taught exploration and environmental geochemistry, environmental geology, and economic geology, and conducted research on Pennsylvania mineral deposits including uranium, copper, zinc, and chromium. Other major research involved oil well brines, radon and uranium decay products in soils, and acid mine drainage. He received a PhD in Geology and Geochemistry from the California Institute of Technology in 1958 and joined Kennecott Copper Co as an exploration geologist in Arizona and later as research geochemist investigating trace element and mineral halos around porphyry copper deposits.

CHEMISTRY AND MINERALOGY OF COAL FLY ASH: BASIS FOR BENEFICIAL USE

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Abstract

Fly ash is the non-volatile, incombustible, thermally altered mineral matter that was contained in coal. The principal component of most fly ashes is altered clays reflecting prominence in the dominant mineral phase in coal. Optimum reuse of fly ash can be achieved if the chemical and physical nature of the ash is known, thus allowing its end use to be engineered to the proper end state. The mineralogical and bulk chemical composition of fly ash there are key fundamental properties that the engineer can manipulate. This presentations is being delivered as a tutorial which will highlight the characteristics of fly ash as they pertain to engineering applications and sound environmental utilization.

Introduction

The most significant environmental problem within the Commonwealth of Pennsylvania is the remnant minelands resulting from World War II coal mining. It is estimated by the Pennsylvania Department of Environmental Protection (PADEP) that 250,000 acres of lands remain impacted and recent stream surveys further suggest that in excess of 4,000 miles of surface streams have been compromised by acid discharge. The estimated dollar figure to return these impacted areas back to any semblance of their original ecological conditions is estimated to be in excess of \$2 billion dollars.

The financial enormity of the challenge far out strips the available resources that are available to the PADEP to address the reclamation. Because of the magnitude of the task, minelands are ranked in a priority which takes those with immediate public health and safety issues, such as fatalities, and places them first on the list to address with the available funding. Without additional resources, the majority of the disturbed minelands in Pennsylvania may never be addressed in our lifetimes.

One approach to address this issue is the development of a cooperative interaction between the Commonwealth and its industries. The purpose would be to take the cast-off materials from the industries, which do not pose an environmental threat, and utilize them in large volume applications to address the reclamation of the Commonwealth's disturbed minelands. The specific case that will be addressed is the use of coal combustion products. These materials are and should be considered a *mineral resource* available to the Commonwealth in the same manner that coal, iron, and water are resources. It is the objective of this paper to describe the chemical and mineralogical nature of this resource and to emphasize the fundamental properties of this class of material so that it can be best utilized in a sound and environmentally ethical manner.

Chemical and Mineralogical Characteristics of Fly Ash

The only coal combustion products considered in this discussion will be fly ash. Fly ash is the mineral matter in coal that has been thermally altered as it cycles through the combustion process. Of the one hundred plus minerals that have been reported in coal, the most abundant are: marcasite, pyrite, calcite, siderite, gypsum, anhydrite, quartz, and clays. Marcasite, pyrite, and siderite alter to form ferrite and hematite phases along with sulfur dioxide and carbon dioxide, respectively. Gypsum dehydrates to anhydrite and the calcium sulfate remains stable above 1400°C. Quartz passes through the combustion process unaffected chemically but does undergo an alpha/beta phase change at 573°C which can contribute to reducing the particle size of the individual grains. The remaining class of minerals is clays of various forms. These minerals will undergo dehydroxylation in a fluidized bed combustion facility and melt to form glass in a pulverized coal combustion unit.

Pulverized coal fly ashes are classified by ASTM by their aggregate alumina, silica, and ferric oxide content into Class F or C. Class F contains a total of at least 70% of these three oxides and Class C contains greater than 50% of the three oxides [ASTM C 618].

Class F coal ashes are generally produced by higher rank coals and typically have a lower calcium content. Table 1 summarizes the average bulk composition of a Class F ash based on some 45 analyses [Scheetz et al. 1997].

Table 1. Average Bulk Composition of a Class F Fly Ash

Oxide	Wt. %/STD
SiO ₂	52.5±9.6
Al ₂ O ₃	22.8±5.4
Fe ₂ O ₃	7.5±4.3
CaO	4.9±2.9
MgO	1.3±0.7
Na ₂ O	1.0±1.0
K ₂ O	1.3±0.8
SO ₃	0.6±0.5
Moisture	0.11±0.14
L.O.I.	2.6±2.4

The bulk chemical composition of Class F ash dictates the mineralogical constituents of the ash. Quartz, the ferrite phase, and mullite each average less than 10 weight percent of the bulk and in most cases the latter two phases average less than 5 weight percent of the bulk. The most abundant phase in a Class F fly ash is the glass that results from the melting of the clays and subsequent exsolution of mullite from the melt. This x-ray amorphous material is the pozzolan that can be exploited in chemical reactions of various alkaline chemicals. The glass is an aluminum substituted silicate glass which contains about 35 to 40 weight percent alumina. In an alkaline environment, where the pH exceeds 9.5, a rapid reaction occurs in which the aluminosilicate framework of the glass is dissolved and hydration products form as a result of the alkali or alkaline earth elements interacting with the fragmented glass forming network. If the elements are one of the alkali metals, the hydration products are proto-zeolitic phases; if alkaline earth metals, especially calcium, are present the hydration products are hydrous calcium aluminosilicates, C-A-S-H, equivalent to the hydration products of Portland cement.

Class C coal ashes are generally produced from the combustion of lower rank coals and contain considerably more calcium in their bulk chemical compositions. In fact, some CCPs derived from Chinese coals exhibit the bulk chemical composition of Portland cement. Table 2 details the average composition of 97 Class C coal ashes.

Table 2. Average bulk composition of a Class C fly ash.

Oxide	Wt. %/STD
SiO ₂	36.9±4.7
Al ₂ O ₃	17.6±2.7
Fe ₂ O ₃	6.2±1.1
CaO	25.2±2.8
MgO	5.1±1.0
Na ₂ O	1.7±1.2
K ₂ O	0.6±0.6
SO ₃	2.9±1.8
Moisture	0.06±0.06
L.O.I.	0.33±0.35

Because of the higher calcium content of this class of ash, the resulting mineralogical phase assemblage is quite different. Quartz, the ferrite phase, and mullite are present as in the Class F ash. The alteration of the clay content of the coal in the presence of calcium results not in large amounts of glass but rather in a suite of silicate and

aluminosilicate and oxide phases. A minor amount of glass is formed in the thermal alteration of this class of coals. The glass contains a rather high concentration of alumina and is, chemically, extremely reactive. Most notable among the dozen or so phases are lime, di-calcium silicate, and tri-calcium silicate. Additionally, periclase and anhydrite/gypsum can be found. These five mineral phases are all principal constituents in modern-day Portland cement. As a consequence, Class C fly ash is not pozzolanic and will, in the presence of moisture, hydrate autogenously.

Another combustion process, fluidized bed combustion, employs a circulating bed of crushed limestone into which coal or coal refuse is introduced and combusted. The operating temperature of this process is much lower than the combustion temperatures of pulverized coal combustion facilities; operating at approximately 800°C. The advantage of fluidized bed combustion is the ability of this process to react with the bed carrier material with oxides of sulfur to form anhydrite while maintaining emission standards for SO_x. The ash content of these facilities directly mimics the types of fuel that are being burned. Table 3 summarizes the bulk ash composition from coal refuse combustion in the anthracite and bituminous coal fields of Pennsylvania. The data show that anthracite ‘culm’ refuse is characteristically low in sulfur which is reflected in the reduced calcium content of the resulting ash. By contrast, the bituminous ‘gob’ derived ash has more than seven times more calcium content, which is also reflected in the SO₃ content. These differences in bulk chemistry directly reflect the chemical reactivity of the individual ashes. In general, the bituminous ash is cementitious and can result in hydrated solids with unconfined compressive strengths in excess of 4,000 psi. Although the mineralogies of the two ashes are similar, the relative percentages of the individual phases vary considerably. Clays in the coal refuse are the principal reactive phase.

Table 3. Typical Bulk Composition of a Fluidized Bed Combustion Fly Ash

	Anthracite	Bituminous
Oxide	Wt. %	Wt. %
SiO ₂	58	34
Al ₂ O ₃	20.4	2.2
Fe ₂ O ₃	5.7	5.9
CaO	4.1	30
MgO	0.62	0.62
Na ₂ O	0.59	0.11
K ₂ O	2.56	1.49
SO ₃	1.1	13.0
Moisture	+0.49	3.7
L.O.I.	3.3	10.0

In a fluidized bed combustion process, the temperature is hot enough to dehydroxylate the clays, a process in the ceramics industry that results in what is known to that industry as ‘meta-clay.’ Dehydroxylated clays are chemically very reactive, which in the presence of excess lime as in the gob-derived ashes, behaves in a similar manner to the glass in Class F fly ashes. The presence of the meta-clay, a source of aluminosilicates, and anhydrite results in the early term formation of the mineral ettringite. This phase, along with the conversion of anhydrite to gypsum, provides the early-term strength development in the ‘gob-derived’ ashes. With time the hydration in the presence of lime of the aluminosilicates prove long term stability.

‘Culm-derived’ ashes are much less reactive. The silica content in the form of quartz is higher and there is generally less lime. One can envision these ashes as being similar to the ‘gob-derived’ ashes, but diluted with an inert quartz phase. ‘Culm-derived’ ashes have been observed to form ettringite and thermodynamic calculations of the fluids in contact with these hydrating ashes have predicted the saturation with respect to 14 angstrom tobermorite, the crystalline equivalent of C-S-H in Portland cement [Loop, 2004].

Fluidized bed combustion ashes derived from the combustion of high BTU fuels are more similar to the ‘gob-derived’ ashes because bituminous coals are more often utilized. Table 4 proved the details of the bulk composition of one such ash. These data show that both the calcium and the sulfate reported are higher than even the ‘gob-derived’ ashes. The efficiency of sulfur oxides capture by this process allowed fluidized bed combustors to burn high-sulfur coals which could not be burned in pulverized facilities without blending with low-sulfur fuels, which offers an added cost savings on fuel.

Table 4. Typical Bulk Composition of a High BTU Bituminous Coal Fly Ash

Oxide	Wt. %
SiO ₂	24
Al ₂ O ₃	6.1
Fe ₂ O ₃	2.1
CaO	42
MgO	0.45
Na ₂ O	0.07
K ₂ O	0.51
SO ₃	20.8
Moisture	+0.25
L.O.I.	2.0

Environmental Issues

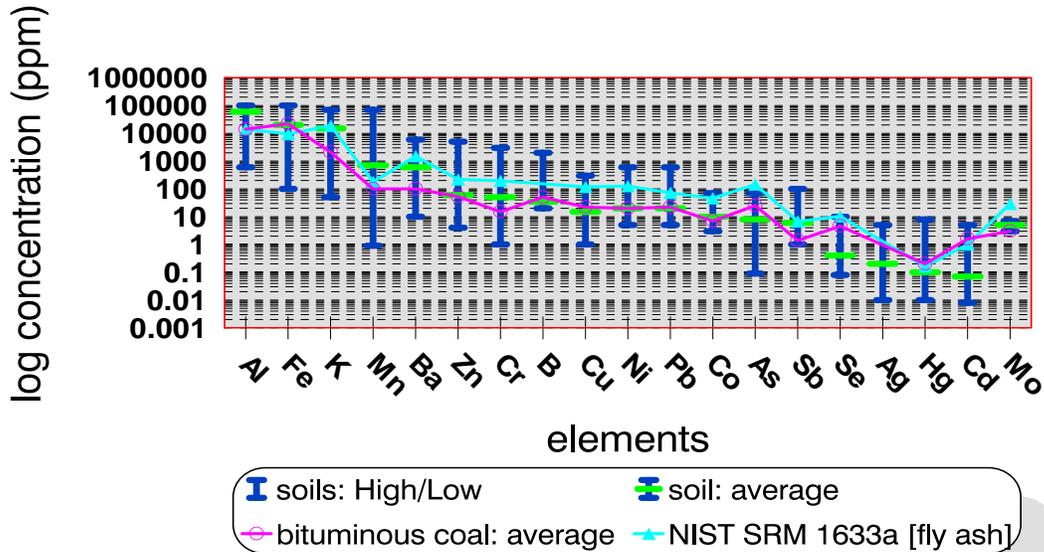
Coal is an organically derived carbon source that formed extensively across the eastern United States during the Pennsylvanian and early Permian between 250 and 330 million years ago. Vegetation grew in great swamps in the equatorial climate which accumulated the organic debris hundreds to thousands of feet in depth. Detrital minerals washed into the swamps and were incorporated as sediment into the accumulating organic remains of the great ferns and related plants. With time, the debris was buried and underwent anerobic decay and compaction from the overlying mass of cover. As a rule-of-thumb, it is considered that one foot of coal resulted from ten feet of original organic debris. These plants bio-accumulated heavy metals as part of their metabolism which was further concentrated during the coal formation process. Table 5 is a detailed summary of the trace elements in coal and pulverized coal derived fly ash and is presented for completeness [NRC, 1980].

To place the trace element distribution of fly ash into perspective, Figure 1 contrasts a selected number of elements for their average in soils from the United States to a bituminous fly ash and to a NIST reference fly ash. These data suggest that, for the most part, fly ashes have the composition of native soils. There are however a few elements that are present outside of the range of soils. Figure 2 graphically represents the RCRA elements, which are enriched with respect to the crust. In this group of enriched elements, mercury is currently receiving a great deal of attention. Mercury in pulverized coal fly ash ranges from 0.01 to 22 ppm [NRC, 1980] and in fluidized bed combustion ash from 0.005 to 0.76 ppm [Earthtech, 2000]. Rose reports attempts to extract mercury from this latter ash by leaching in a sulfuric acid solution and resulted in levels that would conform to the EPA drinking water standards.

Table 5. Detailed Range of Trace Elements in Coal and Fly Ash

Element	Coal (ppm)	Fly Ash (ppm)
Aluminum (Al)	**	11,500 - 144,000
Antimony (Sb)	**	0.8 - 1,000
Arsenic (As)	0.8 - 100	2.3 - 1,700
Barium (Ba)	70 - 120	90.0 - 13,900
Beryllium (Be)	0.24 - 2.60	1.0 - 1,000
Bismuth (Bi)	**	10 - 30
Boron (B)	5 - 125	<10 - 3,000
Bromine (Br)	0.7 - 28.0	0.3 - 670
Cadmium (Cd)	0.1 - 0.6	0.1 - 250.0
Calcium (Ca)	**	5,400 - 177,000
Cerium (Ce)	11 - 41	28 - 320
Cesium (Cs)	0.4 - 6.0	1.4 - 17.0
Chlorine (Cl)	**	13 - 25,000
Chromium (Cr)	10 - 90	11 - 7,400
Cobalt (Co)	1.5 - 34.0	6 - 1,500
Copper (Cu)	5 - 30	30 - 3,020
Dysprosium (Dy)	0.7 - 3.5	4.2 - 31.0
Europium (Eu)	0.17 - 0.90	1.0 - 16.7

Florine (F)	50 - 160	0.4 – 624.0
Gallium (Ga)	3 - 11	10 - 10,000
Germanium (Ge)	<10 - 11,000	
Gold (Au)	**	0.004 – 0.5
Hafnium (Hf)	0.6 - 2.2	1.4 - 11.0
Indium (In)	**	0.1 - 2.4
Iodine (I)	0.35 - 5.0	0.1 - 200
Iron (Fe)	**	7,800 – 289,000
Lanthanum (La)	6 - 24	17 - 270
Lead (Pb)	1 - 18	3.1 - 1,600
Lithium (Li)	**	77 - 120
Lutetium (Lu)	0.04 - 0.40	0.41 - 4.30
Magnesium (Mg)	4,900 - 60,800	
Manganese (Mn)	2.5 - 15.0	31 - 4,400
Mercury (Hg)	0.05 - 0.48	0.01 - 22.0
Molybdenum (Mo)	6.5 - 500	
Nickel (Ni)	6.5 - 29.0	1.8 - 8,000
Niobium (Nb)	**	16.0 - <22.0
Palladium (Pd)	**	0.2 - ---
Phosphorus (P)	15 - 1600	600 - 2,500
Platinum (Pt)	**	0.7 - ---
Potassium (K)	**	1,534 – 34,000
Rhodium (Rh)	**	0.02 - ---
Rubidium (Rb)	9 - 63	25 - 300
Samarium (Sm)	0.9 - 4.2	3.7 - 43.0
Selenium (Se)	1.75 - 9.0	1.2 - <500
Silicon (Si)	**	196,000 - 271,000
Silver (Ag)	0.01 - 0.06	1 – 50
Sodium (Na)	**	1,180 – 20,300
Sulfur (S)	**	0.11 - 0.25
Strontium (Sr)	30 - 550	40 - 9,600
Tantalum (Ta)	0.12 - 1.20	0.5 - 2.6
Tellurium (Te)	**	0.11 – 10.0
Thallium (Tl)	**	1.1 – 100.0
Thorium (Th)	1.8 - 9.0	1.8 - 68.0
Tin (Sn)	0.2 - 8.0	<3.0 - 4,250
Titanium (Ti)	**	400 - 15,900
Tungsten (W)	0.22 - 1.30	2.9 - 42.0
Uranium (U)	0.4 - 3.0	0.8 - 30.1
Vanadium (V)	15 - 70	20 - 1,180
Ytterbium (Yb)	0.18 - 1.50	1.7 - 23.0
Yttrium (Y)	**	21 - 800
Zinc (Zn)	2 - 125	14 - 13,000
Zirconium (Zr)	8 - 90	100 - 5,000



soils: USGS Team on Native Soils/coal: National Research Council/fly ash:NIST Standards

Figure 1. Comparison of selected trace elements in fly ashes to average soil content in the United States.

Enrichment Factors for RCRA Elements

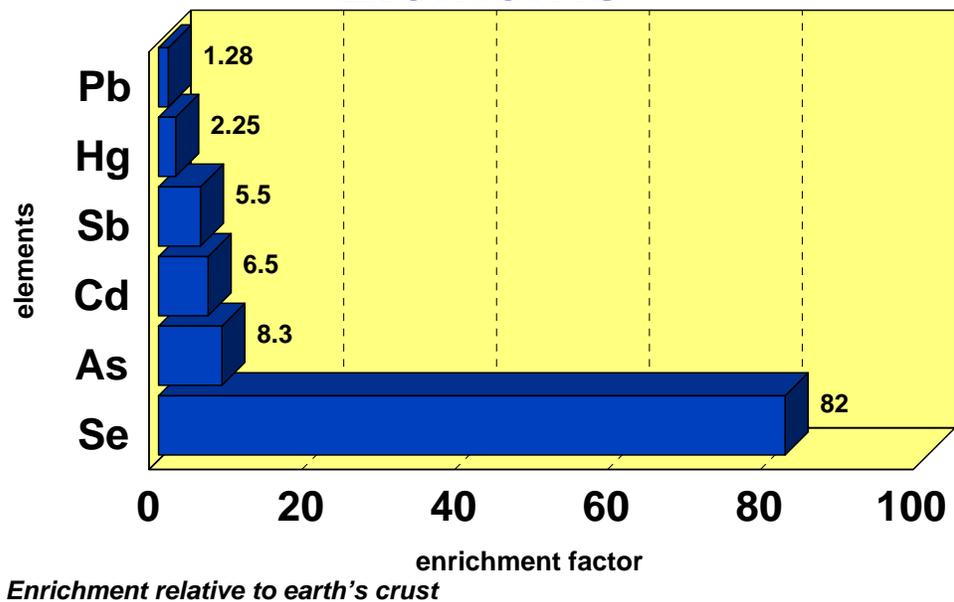


Figure 2. Enrichment of RCRA elements with respect to the earth crust.

Antimony, arsenic, and selenium are of interest because under the geochemical conditions of mineland usage, these three elements exist as anions. Anions are not typically exchanges of common clays and, as such, have little to inhibit their movements. Although selenium exhibits the most significant enrichment of all of the RCRA elements,

it is seldom encountered as a problematic element. The incorporation of these anions into the structure of ettringite appears to serve as a sink to mitigate the release of these elements.

An issue that has recently been raised addresses the amount of radioactivity that is present in fly ash. Some reports would have us believe “A coal plant releases about 74 pounds of uranium-235 each year, enough for two or more nuclear bombs.” (D. R. Francis, *The Christian Science Monitor*, 29 May 2001). When the natural abundance of ^{235}U is multiplied by the total concentrations of uranium in fly ash and by the total quantity of ash generated on an annual basis, indeed this quantity of fissile uranium may be present but it would mean that every bit of ash produced in the United States would have to be processed to extract it. Clearly, if fly ash were a viable source of fissile material, it would have been exploited during the Cold War arms race. In fact, the radioactivity in fly ash is comparable to background levels throughout the country and is no higher than the natural levels in the most widely encountered rocks. Figure 3 shows these comparisons. The USGS does not consider radioactive components of fly ash a health issue [USGS, fact sheet FS-163-97].

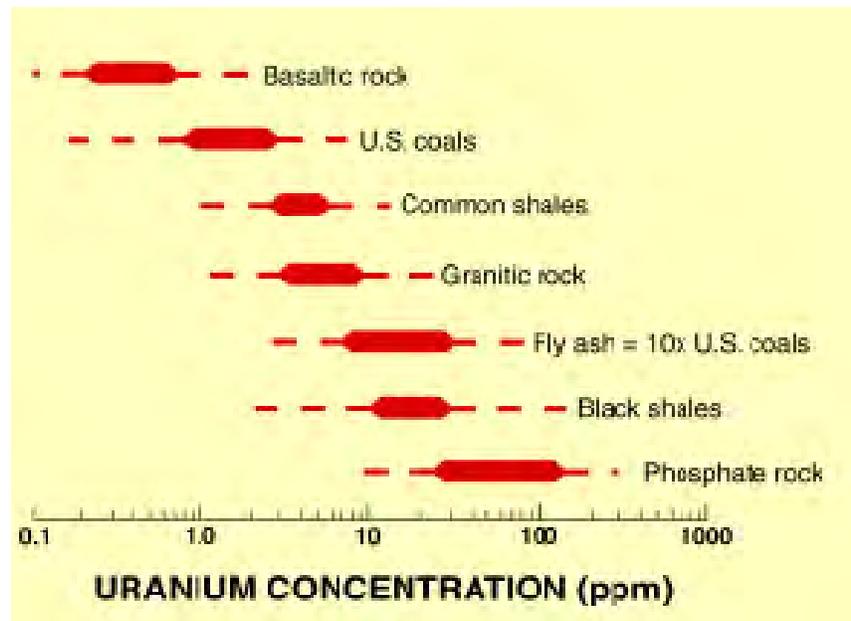


Figure 3. Comparison of uranium concentrations in coal and fly ash contrasted to common rock types.

Finally, one other issue that has been raised is the presence of PCB's associated with the presence of unburnt carbon and char in most fly ashes. Numerous studies have been conducted which tested for the individual PCB compounds. 210 individual analytes were tested in 11 pulverized coal ash samples and all compounds tested <40ng/Kg [EPRI 1998]. Comparable studies by PPL confirm these finding.

Conclusions

Coal combustion products should be considered a mineral resource. They have been successfully used as an engineering material for mineland reclamation and a wide variety of other applications. Fly ash must be considered as a 'material' in this sense. However, to understand the chemical and mineralogical characteristics of these materials will enable expanded and environmentally ethical applications. Like all mineral resources, prudent care should be exercised in the applications of fly ash to avoid potentially replacing one environmental problem with another.

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Dr. Barry E. Scheetz, Professor of Materials, Civil and Nuclear Engineering, is a faculty member at the Materials Research Institute at the Pennsylvania State University. His research deals with the chemistry of cementitious systems including environmental remediation by the use of industrial by-products, focusing on large-volume fly ash-based cementitious grouts. The program activities approach the utilization of these by-products from a materials aspect; that is, by knowing the mineralogy and the bulk chemistry of the by-product; hydrated end products with specific characteristics can be engineered. Field activities have focused on the long-term elimination of acid mine drainage and the elimination of public health and safety issues associated with abandoned mines. He is a member of The President's US-INDO Energy Consultation serving in the Coal Advisory Group. He also has been active in the Acid Drainage Technology Initiative where he serves as the program chairman of the Avoidance and Remediation Committee for the Coal Sector. He has also served on several National Research Council committees reviewing nuclear waste management activities at various National Laboratories. He received an M.S. in 1972 in Geochemistry from The Pennsylvania State University followed by a Ph.D. in Geochemistry and Mineralogy in 1976, also from The Pennsylvania State University. He has been an active research faculty member for 28 years.

LESSONS LEARNED FROM FULL-SCALE, NON-TRADITIONAL PLACEMENT OF FLY ASH

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Abstract

Reclaiming a surface mine pit or lake can have significant long-term benefits including the reduction of AMD, elimination of hazardous physical conditions, and the return of land to productive use. Fluidized bed combustion (FBC) ash from waste coal burning plants in the anthracite region contains dehydroxylated clays, calcium oxide, and gypsum which influence the chemistry of the water they come in contact with. If ash is distributed properly, the same minerals will act to strengthen and shrink pore spaces within the FBC ash monofill by encouraging the growth of minerals such as 14Å tobermorite, calcite, and ettringite. With proper placement and monitoring, alkaline FBC ash can be an important resource for eliminating abandoned mines, as shown in the cases of the Big Gorilla and Knickerbocker pits.

Introduction

In Pennsylvania, coal combustion by-product (CCB) ash is conventionally placed at least eight feet above the regional water table and four feet above any perched water table, so that it is not in contact with groundwater. This article is based on two full-scale demonstration projects in the anthracite region of Pennsylvania in which this was not the case. In the first project, dry fluidized bed combustion (FBC) ash was placed in standing mine water, and in the second project, FBC ash was slurried into a dry mine pit. A third demonstration, where FBC ash was to be slurried into standing mine water, has not yet taken place. The structural stability of the final ash monofill, basin to pore scale water chemistry, and long-term predictions are all key to a successful project and will be considered in this paper.

The need exists for a way to reclaim abandoned pit lakes that are common throughout the anthracite region of Pennsylvania. These lakes have been the location of drownings, cause further degradation of the regional water quality, and are an eyesore. Little clean fill exists near these sites in the anthracite region, as they were often deep-mined with minimal overburden removed. The cost of reclaiming the Shen Penn pit alone was estimated at \$20 to \$28 million, if a by-product material was not available (Dalberto et al., 2004). The goal of the reclamation projects discussed in this paper was to use FBC ash to create an inert, if not alkaline, plug of fly and bottom ash that fills the former surface mine. Once ash fills a pit lake, conventional reclamation using ash and clean fill can take place, finishing with a seeded area that conforms to the land's pre-mining contour.

At the site of the Big Gorilla project, where dry fly and bottom ash were being placed into standing mine water, the lake surface was not connected to a regional water table. The approximately 130 million gallon lake, called the "Big Gorilla," was a remnant of the mining of the Mammoth coal seam and is bordered to the south by the Centralia thrust fault. Over 3 million tons of ash was used to fill the pit lake over a period of approximately 6 years. The Northeastern Power Company (NEPCO) generates fly and bottom ash as it burns waste anthracite coal in a fluidized bed in which limestone is used to control sulfur emissions. The amount of limestone is adjusted to the sulfur content of the material, and the resulting fly ash is approximately 6.5 % wt. portlandite, 0.4% wt. calcite, and 0.5 % wt. gypsum. The four other major components of the FBC fly ash are dehydroxylated clays (63.6% wt.), quartz (20% wt.), mullite (7% wt.), and hematite (2% wt.). The mineralogy of FBC ashes resulting from the burning of waste culm piles in the anthracite region differs significantly from both class C and class F ash from pulverized coal plants. The mineralogy and placement technique have contributed to the structural integrity of the ash monofill.

Structural Stability

Ash leaves NEPCO through two silos, at which point it is transported by trucks in 35 ton loads to two platforms at the eastern end of the Big Gorilla pit. The upper and lower platforms were originally built of clean fill, until the practice became cost-prohibitive. Once the demonstration permit was issued, in 1997, ash was placed close to the edge of the platforms by a dumping truck and a bulldozer was used to push it into the water. The routine use of trucks and bulldozers was the only compaction that took place on the ash platforms. Initially, ash was only placed from the upper platform during the winter, due to the fear of slippery conditions closer to the water, but this concern proved unnecessary with time, and platforms were used without regard to the season.

Quantification and close control of the structural stability of the ash platforms should be a component of any plan for ash placement in water. The Proctor density test is a measure of the density of cohesive materials as a function of moisture content and serves as a baseline for field measurements of compaction. The recommended amount of moisture is added to the ash in the trucks at the plant; this process is called conditioning. Once the fly and bottom ash was placed on the platforms, a ^{137}Cs densitometer was used by the PA DEP to ensure the DEP requirements of 90% or greater of the laboratory measured Proctor density value. At times, the field density was over 100% of the theoretical value, but it was never below 90%. At the Big Gorilla site, both a field penetrometer and bearing capacity tests were used to determine the weight bearing capacity of the platforms. The soil penetrometer tests routinely met or exceeded a value of 3.0 tons per square foot (tsf) and the drilling tests determined that the bearing capacity was greater than 2 tsf.

At the Knickerbocker site, which was receiving a slurry of fly and bottom ash from the Schuylkill Energy Resources co-generation plant, Proctor density and bearing capacity tests were routinely conducted by the PA DEP, and the results deemed positive. One important goal of the slurry configuration was to minimize turbulence of the water as it flowed from the pipe and over the ash. In this regard, the pipe outlet was placed close to the ground, and only a very shallow slope was allowed for the slurry to flow over. Shortly after the slurry left the pipe, the fly and bottom ash mixture was stable enough to walk and drive a vehicle on. At one point, a smaller test was conducted to evaluate the effects of adding cement kiln dust (CKD) on the ash strength. In this test, the slurry line, containing additional CKD, was discharged into standing mine water. The ash behaved in the water in a manner similar to a delta formation, where ash settled out as it left the pipe and the water flowed to the far end of the cell. The additional CKD was found to contribute to the increased packing density of the ash fill.

The structural success of the ash plugs is not independent of the chemistry, as will be further discussed in the next section. It can be attributed to the chemical interaction of the components of the FBC ash, specifically, calcium hydroxide (CaO), the dehydroxylated clays, and gypsum. When CaO is exposed to moisture it becomes the mineral portlandite (CaOH)₂. The addition of CaO to water can raise the pH of the water to a theoretical value 12.45. This increase in pH, and concurrent increase in alkalinity, is required for the development of pozzolanic reactions in the ash. In the presence of CO_2 , the Ca will contribute to the precipitation of calcite. The heating process (to approximately 800°C at NEPCO) causes clays that were initially present in the waste coal to lose hydroxyl ions, in turn making them more reactive, especially in the presence of the high pH values that were found in the ash platform (11-12). C-S-H can be formed, and is thought of as the “glue” in Portland cement. The gypsum in the ash was also found to serve an important purpose in contributing to the formation of ettringite, a calcium, and aluminum sulfate mineral. Ettringite is a mineral present in Portland cement that can form in the pore spaces of the ash.

The mechanical compaction of the ash platform by the trucks and bulldozers, which minimizes density and the size of pore space is the first step to structural stability. The second step is the continued filling of existing spaces by the growth of minerals such as calcite, C-S-H, and ettringite. These processes are likely occurring on very different time scales.

Water Chemistry

Three water sources are especially important in the Silverbrook Basin. The first is the Big Gorilla water, which shows a significant change in chemical composition as ash is first placed in the pit lake. The two more long-term indicators of change in the basin are the pore water within the ash platform, which must be monitored by wells, and the basin outflow point. The pit lake water should be studied as the first field measured indication of the long-term

ash/water interaction. This is especially true for the first months of ash placement, as the water can change significantly in chemical composition with relatively small inputs of ash, especially in the case of highly alkaline FBC ash. The water in the surface mine pool should ideally be largely contained within the surface mine pool, unless beneficial aspects may be provided by mixing it with acidic water in the surrounding area. Since the goal of the ash placement is to fill the entire mine pool, the water in the pit lake is transient. Therefore, high pH and alkalinity values will not be of as great a concern for wildlife as if the altered lake water were to be a permanent feature.

Within the first 3 months of ash placement in the Big Gorilla, the pH value of the water rose from 3.6 to approximately 10. After the first winter hiatus in ash placement, the pH value was most commonly between 11 and 12. The alkalinity increased and decreased significantly in response to ash placement, with concentrations between 50 and 200 mg/L when no ash was added to the water. In addition to being controlled by carbonate reactions, the alkalinity in the mine pit lake was also governed by the silica and hydroxide content of the water, which fluctuated with ash input. When placing ash after a break in ash placement, the silica concentration initially increases, but with further ash placement, the silica concentration decreases and alkalinity increases, until the alkalinity is controlled primarily by the hydroxide content of the water. Carbonate reactions control the alkalinity when the ash is not actively contributed to the water. The substantial carbonate fraction of the ash also represents a long-term reservoir of alkalinity. A white rim was present on the banks of the Big Gorilla mine pool, demonstrating that in the zone that mixes with CO₂, calcite will precipitate. It is expected that in the high pH porewater, the pores that are open to CO₂ diffusion will eventually have calcite precipitated within them.

Because aluminum (Al) is amphoteric, it can be present in toxic concentrations at high or low pH values. When the ash was initially added to the water and the pH became neutral, most of the aluminum in the water precipitated as a hydroxide and was covered by successive layers of ash. This same process alters iron and manganese aqueous concentrations. However, the dissolution of dehydroxylated clays and mullite at high pH values would also contribute to the aluminum concentration in water within the mine lake and the pore spaces. Geochemical modeling of the Big Gorilla ash pore waters and scanning electron microscope (SEM) analysis of the ash from the Knickerbocker site indicate the presence of ettringite. When enough sulfate and calcium are available, the aluminum content of the water at higher pH values (above 11.5) can be controlled by ettringite (Loop, 2003). An additional benefit of the presence of ettringite is that it can bind within its structure more toxic elements such as Cr, As, and Cd (Gougar et al., 1996).

Geochemical modeling of the waters sampled from a borehole in the ash platform indicated that the water was supersaturated with respect to 14Å tobermorite. Assuming thermodynamic equilibrium, PHREEQC (Parkhurst, 1995) was used to speciate the water. The concentrations of Ca and Si in the water encourage cementitious reactions to take place, a primary indication of which is the presence of tobermorite. 14Å tobermorite is very similar in composition to C-S-H, an important binding component of Portland cement.

Thus, the FBC ash has CaO and CaSO₄ from the addition of limestone to capture sulfate, as well as reactive clays containing more easily dissolved Si and some Al. In non-FBC ash, amendments with alkaline material, for example cement kiln dust or lime kiln dust, may be necessary to achieve these results.

The changes throughout the basin can be measured by monitoring wells, if they intersect major flowpaths. In the case of the Silverbrook basin, where the Big Gorilla is located, only one well showed altered chemistry as a result of ash placement in the Big Gorilla. The well was within 100 feet of the pit lake and registered a change in chemistry approximately 2 years after the first ash was put into the pit. The chemical change only lasted for about a year, at which time the connection between the lake water and the well was sealed.

More useful than monitoring wells, in areas where preferential flow occurs, can be a mine outflow point, analogous to a spring in limestone bedrock. The Silverbrook outflow drains the entire basin, including the deep mines, and is a large-scale monitor for processes occurring throughout the basin. No chemical signatures in the outflow could be isolated as impacts from the Big Gorilla, however there were indications of change in the basin. Calcium, iron, and sulfate concentrations have all increased, but began prior to ash placement in the surface mine pool. The increase in calcium concentration began in 1992, and can be traced to land reclamation in the basin near areas that may have contained preferential pathways to the deeper, Buck Mountain coal seams, such as cropfalls. The calcium concentration is a reflection of the dissolution of CaO from the ash, which contributes to the alkalinity in the basin, but has yet to affect the alkalinity measured at the outfall. Iron and sulfate concentrations in the Silverbrook outfall

have been increasing as culm has been collected and stored onsite. This unfortunate change is a consequence of the moving of the culm from onsite and beyond, which exposes fresh surfaces of the pyrite to weathering, and in turn generates more acid mine drainage. However, because the waste coal is being burned, after this process, the pyrite is being removed permanently, as opposed to waste coal piles that remain sources for acidic drainage for decades. Seasonal signatures of sodium and chloride can also be detected in the water. Thus, no direct effects of ash placement in the Big Gorilla mine pool can be distinguished in the Silverbrook outflow, after seven years of ash placement.

Long-Term Predictions

Current indications of the long-term effect of ash placement in the Big Gorilla mine pool and the Knickerbocker pit point to an inert material with very low values of hydraulic conductivity. It had been initially hoped that alkalinity from the FBC ash placed in the Big Gorilla would enter the deep mines below and have a beneficial effect on the basin's water. In fact, this has not occurred. Only very limited amounts of water have left the Big Gorilla, and the water that has must have become acidic during interaction with pyritic material and mixing with acidic water in the subsurface. This does, however, leave a large reservoir of alkaline material in the pore spaces of the ash monofill, which will contribute to high pH values of the pore water, and will in turn increase the stability of the ash and decrease the permeability of the ash fill. Permanent monitoring wells will be located in the ash placed at these locations and will continue to be monitored for changes in chemistry of the waters. The permit for ash placement on the NEPCO site will soon revert to a conventional land reclamation permit, as the ash will be above the water table. At both NEPCO and at the Knickerbocker site, ash will be used to return the land to the estimated pre-mining contour, which will eliminate the highwalls. Clean fill will be placed on the ash, and vegetated. Water flow into the deep mines through the Big Gorilla and Knickerbocker areas will be largely eliminated due to the low permeability of the ash in the subsurface.

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FUEL AND ASH MANAGEMENT FOR A TYPICAL CFBC POWER PLANT

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Abstract

The legacy of mining coal in Pennsylvania spans over a 100 year period starting at the beginning of the 20th century. Coal mined in Pennsylvania helped fuel the industrial revolution, established the mass production of steel, and served to propel the United States to victory in two World Wars. During the industrial boom period, few people recognized the environmental consequences that would follow in the years ahead. Large waste coal piles continued to grow as the coal mining industry increased production year after year. The resulting environmental damage, caused by elemental sulfur in the waste coal, is known today as Acid Mine Drainage (AMD). The acidification of groundwater and streams is considered to be the greatest environmental problem in Pennsylvania.

Until recently, the solution to the Acid Mine Drainage problem was nearly hopeless since costs associated with traditional methods of remediation are simply prohibitive. The importation of the Circulating Fluidized Bed Combustion (CFBC) boiler technology from Europe, in the early 1980's, changed that. The CFB boiler design offers the first glimmer of hope due to its ability to burn large amounts of waste coal, thereby, eliminating the waste coal from the environment. In addition to providing useful electricity and steam from waste coal, the resulting ash product was determined to have beneficial value in reclaiming waste coal contaminated land due to its alkaline properties. Simply removing the waste coal from the environment would not completely solve the problem because the acidity in the surrounding soil and residual sulfur bearing waste coal would remain unabated. Mixing the alkaline ash product with underlying soil and residual waste coal serves to neutralize the remaining acidity in the vicinity of the waste coal piles, thus returning the area to its pre-mining condition.

The Circulating Fluidized Bed power plant serves the dual purpose of producing valuable electricity and steam while simultaneously reclaiming the land occupied by environmentally damaging waste coal. This reclamation is accomplished at no cost to the tax payers of Pennsylvania and has a remarkable record of success.

The CFBC Boiler

In the 1980's, American companies began to explore the possibility of using a new technology to combust fuels with low energy value. The Circulating Fluidized Bed Combustion (CFBC) boiler design, although new in North America, was used with great success in Europe. In Scandinavia, for example, these boilers were used to burn peat and wood wastes. This was attractive from a commercial standpoint due to the abundance of these low grade fuels compared with the high costs of using conventional fuels. During this same time period, energy regulations were changing in the United States, making it commercially feasible for independent power producers to employ the CFBC boiler technology. Pennsylvania proved to be an ideal State for the application of this technology with its abundance of waste coal for fuel. At last, the possibility of reversing the most serious environmental hazard in the State passed from wishful thinking to reality. During the initial pilot plant studies, many design changes were made to adapt the CFBC boiler for waste coal combustion. These plants have proven to be very reliable with many having on-stream performances of greater than 90%.

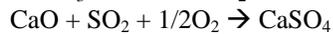
A close look at the CFBC boiler design shows significant differences when compared to a typical pulverized coal fired boiler (figure 1). The following discussion of CFBC boiler operation uses the Cambria Cogeneration unit, located in Ebensburg, Pennsylvania as a model. The waste coal fuel is fed directly into the combustor at four locations and burns approximately 35 – 40 tons/hour. The fuel feed is sized to about 1/4" and is introduced into the bottom of the combustor. The combustor operates at about 1,650 °F and the waste coal ignition temperature is about 900 °F.

A critical aspect of CFBC boiler operation is the method of introducing air for combustion. In addition to staging air for proper combustion, the primary air system must also serve the purpose of fluidizing the bed material for

circulation. The primary air fan supplies most of its air to the under-bed grid nozzles. The floor of the combustor consists of thousands of air nozzles that act to suspend the bed material in a fluidized state throughout the combustor. The bed material consists of waste coal, ash, limestone, unburned carbon, and products of the limestone reaction. This hot circulating bed provides the heat required to generate steam in the boiler water wall tubes which comprise the sides of the boiler for its entire height. Interestingly, a significant amount of the heat transfer is accomplished via conduction as the hot bed material slides down the walls of the water wall tubes. The suspension of the bed material in the combustor also provides a well mixed environment for the combustion and limestone reactions. As expected, the bed density is greatest at the bottom of the combustor and decreases as the bed rises to the top and approaches the cross-over duct to the cyclones.

Limestone is used to remove sulfur dioxide that forms from the elemental sulfur that is present in the waste coal. The limestone is introduced with the fuel at the four fuel feed locations and is no small amount. For every pound of fuel fed to the combustor, approximately 0.20 pounds of limestone is required for SO₂ control. Typical sulfur values in waste coal range from 2% – 4 % by weight. The limestone product used in this particular unit contains 83% CaCO₃ by weight.

Calcium carbonate does not react with SO₂ directly. Sulfur capture is accomplished by CaO (lime) that forms when limestone is introduced to the combustor at 1,650 °F. The limestone and sulfur capture reactions are described as follows:



From the reactions we can see that SO₂ reacts with CaO to form CaSO₄. The calcium sulfate is a solid that makes up a portion of the bed inventory. The environmental air permit requires that 92% of the SO₂ be removed from the flue gas. The CFB design allows these units to operate with much lower SO₂ emissions when compared with typical coal fired boilers operating without flue gas desulfurization.

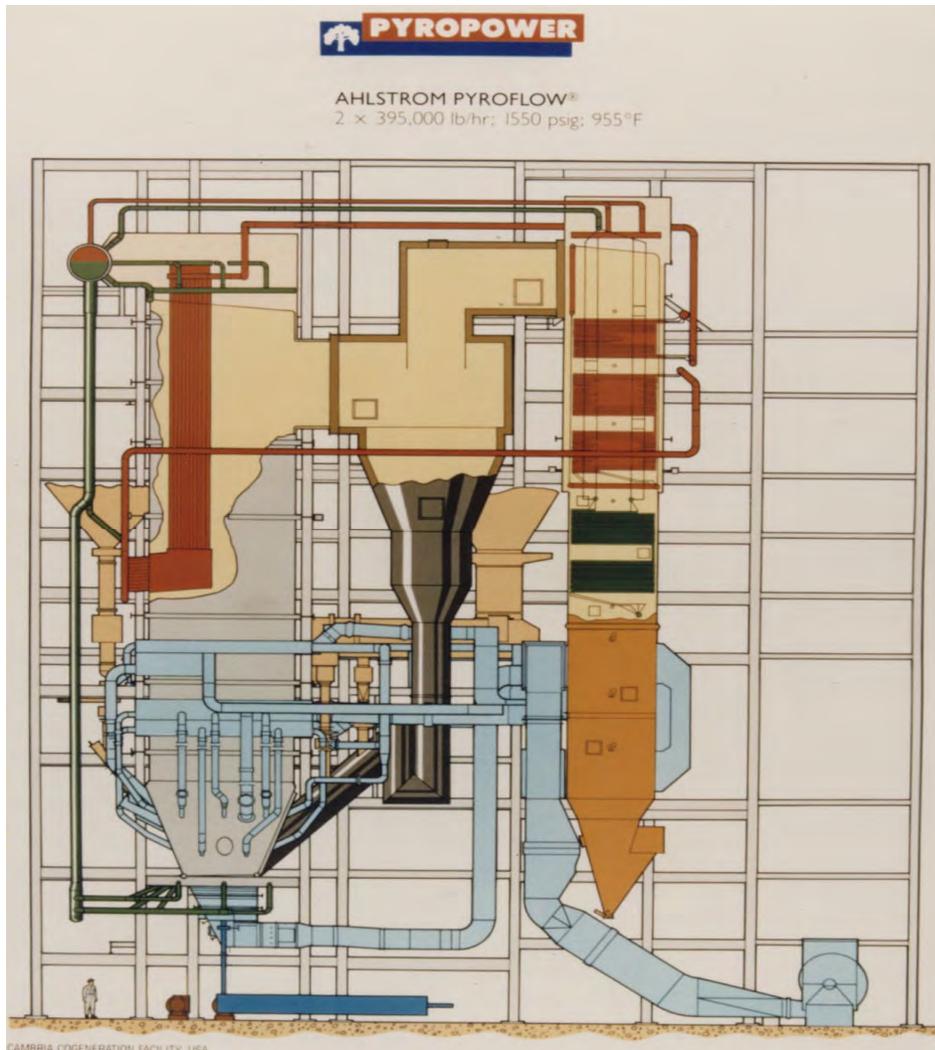


Figure 1. Cutaway View of CFBC Boiler

Another unique feature of the CFBC boiler design is the cyclone section that is attached to the combustor. In addition to the primary and secondary air fans that force air into the combustor, an induced draft or ID fan is used to pull flue gas and fine particulates through the heat recovery section of the boiler. These fans are balanced to set up a pressure differential that allows for the proper circulation of the fluidized bed and flow of the flue gas stream. Each boiler is equipped with two cyclone separators that allow the heavier particles, including unburned carbon, to recirculate into the combustor section for complete combustion. At the top of the boiler, the bed material and flue gas pass through a crossover duct and into the cyclone separators. As stated, the heavier particles fall into a down-comer duct where they are reintroduced into the combustor. The lower portion of the down-comer duct consists of a trap or seal arrangement that prevents the combustion gases and bed material from “short circuiting” the cyclone separators. Fluidizing nozzles are employed at the base of the seal to allow the bed material to flow back into the lower combustor section.

The CFB boiler generates two different ash products. The heavier ash particles remain in the bottom of the combustor and are removed using bottom ash screw coolers. Each boiler uses four of these units and they are designed to maintain proper solids loading at full operating conditions. The bottom ash coolers consist of hollow, water filled screws that transfer hot bed ash at approximately 1,650 °F to a conveyor where the exit temperature is about 375 °F. Likewise, the cooling water flowing through the hollow screws consists of condensate that absorbs heat from the ash to increase the steam cycle efficiency of the plant. Bottom ash is ultimately conditioned with water to improve its handling characteristics before transferring it to final disposal via belt conveyor.

The fly ash is carried through the heat recovery components or convection section of the boiler with the flue gas. The fly ash and flue gas are cooled in the convection section prior to entering the baghouse portion of the boiler. The baghouse consists of ten compartments, each equipped with 240 filter bags. Mechanical filtration is used to separate the fly ash from the flue gas. Fly ash is collected in the compartment fly ash hopper and conveyed pneumatically to a silo for storage. The fly ash is mixed with water to allow conveyance via a belt conveyor. Although the bottom and fly ash products are initially treated separately, they are ultimately recombined on the same transfer belt for eventual use as a soil amendment at the active waste coal excavation site.

Waste coal combustion ash is often referred to as “fly ash” when, in fact, the ash product is a mixture of both bottom and fly ash. In a typical CFBC boiler, the fly ash/bottom ash split is commonly 60% fly ash and 40% bottom ash by weight. The Cambria facility is somewhat unusual given that the split is 40% fly ash and 60% bottom ash on a mass basis.

Waste Bituminous Coal

The origins of the waste coal used by the modern CFBC boiler can be traced back to the turn of the century, in the early 1900’s, when coal was used as a fuel to power the industrial revolution. Later, the coal industry supplied energy for electrical power generation and steel production. Since coal is mined in seams due to its stratification among native rock, miners developed machines and methods to remove the virgin coal while leaving the surrounding rock zones in place. During the mining operation it is inevitable that portions of the rock zone at the coal/rock interface are mined with the virgin coal in an attempt to remove as much usable coal as possible. Once on the surface, the impure coal is classified and prepared for quality. Coal that is mined at the strata interfaces contains a mixture of rock and coal that cannot be separated. It also has no useful value and was deposited, over many years of operation, into large waste piles. This is where the environmental problems caused by waste coal begin. Waste bituminous coal in western Pennsylvania is commonly called GOB. The acidification of groundwater and streams, caused by residual sulfur in the waste coal, has caused profound environmental damage throughout the State of Pennsylvania. This type of environmental damage is commonly referred to as Acid Mine Drainage or AMD. Acid mine discharges also occur in abandoned deep mines when water, flowing through the mined areas, eventually discharges at the existing mine entry openings. It is the greatest environmental problem in Pennsylvania, encompassing 43 of the State’s 67 counties and contaminating over 2,400 miles of stream water.

With this as a backdrop, many regulators, environmental groups, and concerned citizens struggled with the nearly impossible task of determining a solution to these AMD problems. AMD is associated with the roughly 250,000 acres of abandoned mine lands and hundreds of millions of tons of waste coal scattered throughout the State. Employing the CFBC boiler technology to remove the waste coal is only part of the solution. Removing the source of the acidic and metals contamination does not complete the environmental remediation. Using the alkaline CFBC ash to neutralize the acidity and bind the offending metals completes the picture. What was once considered an unusable waste with severe environmental impact is now converted to useful electricity and steam. In addition, the abandoned waste coal sites are reclaimed, with the oversight of the Pennsylvania Department of Environmental Protection, using sound beneficial use practices and carefully developed mining plans.

The characteristics of waste bituminous coal such as carbon content, ash content, and sulfur concentration vary widely. The CFBC boiler design is well suited for burning fuels with low BTU value and highly variable composition. Even with its inherent design flexibility, the CFBC boiler has limitations. It cannot be used to burn waste coal materials with unlimited sulfur and ash composition. In addition, the sulfur concentration is limited by the facility environmental permit regarding sulfur dioxide capture requirements. Despite these limitations, the CFBC boiler technology is far more capable of handling fuel variability than conventional pulverized coal fired units.

Typical fuel properties for the Cambria facility consist of waste coal with an upper limit for ash of about 45% – 50% and sulfur composition of 2% – 4%. The fuel source is currently obtained from the existing waste coal site in Indiana, Pennsylvania, formerly owned by the Rochester and Pittsburgh Coal Company. This site is located in the borough of Ernest and resides on property owned by the Cambria CoGen Company. The current waste coal specifications, using averaged values, are 7,300 Btu/lb, 45% ash, and 3.2% sulfur.

The basic composition of waste coal can be roughly described as a 50/50 mixture of coal and rock. The presence of sulfur and moisture make up the remaining balance. Sulfur is the main culprit associated with acidic contamination

of surface and ground water. Sulfurous acid is formed from the sulfur and sulfur oxides that make up the waste coal. The ash product that is removed from the CFBC boiler is the unburned rock that enters with the waste coal. The CFBC ash product also contains byproducts from the limestone reactions that occur in the combustor, primarily calcium sulfate. A more detailed elemental analysis of the waste coal indicates the presence of various metals. These metals are also found in native soils. A discussion of the fate of these metals is provided in more detail in subsequent sections of this paper. The metals in the waste coal are environmentally problematic because they are particularly mobile. The acidic conditions of the existing waste coal produce a situation that allows the various metals to leach into the ground and surface water.

The Fate of CFBC Solids

The CFBC boiler is uniquely designed to burn fuels with low heating value and can handle relatively high variability in fuel quality. To understand the disposition of solids into and out of the process it is instructive to think of the boiler as a chemical reactor. The standard combustion reaction of burning carbon in the fuel to generate heat is typical of any fuel fired boiler. The CFBC boiler can burn waste coal with low carbon content. The Cambria units currently burn waste coal with approximately 45% carbon. The balance of the incoming fuel consists of ash (45%), moisture, and sulfur. Prior to combustion, the waste coal is sized, crushed, and dried using equipment specifically designed for this purpose. The as-received moisture content of the waste coal varies according to the time of year. Material that is excavated during the cold weather months contains more moisture than during the warm weather periods. The dried waste coal typically has a moisture content of about 3% – 4% by weight. With the exception of carbon and a small amount of sulfur, the solids that enter the combustion process pass through the boiler in the solid state. A simplified mass balance (Figure 2) shows the overall mass of materials entering and leaving the CFBC boiler. This is not a detailed mass balance and is not intended to account for the materials that change state. The basis for this simplified mass balance was calendar year 2003, however, any time basis can be used to generate a material balance. According to this material balance, the added limestone contributed an additional 28% of solids to the boiler. Typical limestone addition for the Cambria units is more commonly in the 20% – 30% range, but as stated earlier, fuel and operational variability can cause changes. The limestone significantly changes the characteristics of the ash during the combustion process. There is nothing inherently unique about the ash that enters with the waste coal. It is simply rock consisting primarily of silicon oxide. The limestone contributes to the ash inventory by reacting, ultimately, to form calcium sulfate. In addition, there are also some unreacted species of calcium oxide (lime) and calcium hydroxides. The byproducts of the calcium reaction give the CFB ash product its alkaline properties. The process also requires a significant amount of water addition to condition the ash for improved handling characteristics. In this case, the total mass reduction of solids leaving the system is 10% relative to the mass of solids entering the system. This is also a unique feature of the CFBC boiler. The large amount of ash entering with the fuel combined with the added limestone produces an ash stream that is quite large relative to the amount of fuel fed to the boiler. The Cambria units typically do not operate with ash loading as high as the conditions shown in Figure 2. These values simply reflect calendar year 2003 operating conditions and are typically closer to a 20% solids reduction.

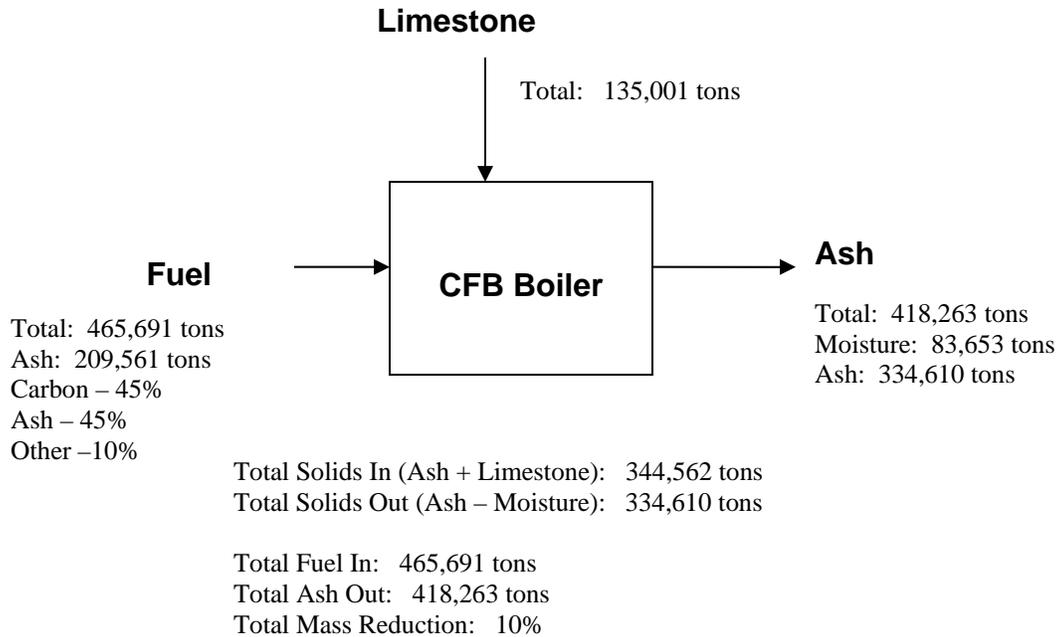


Figure 2. Simplified Mass Balance for Total Solids

As previously discussed, the waste coal contains various metals that occur naturally in the native rock zones from which the material was originally mined. The metals in the waste coal pass through the boiler, unaffected by the process, and remain in the ash. Since the amount of ash produced in the CFBC boiler is high relative to the mass of waste coal fired, there is also very little concentrating effect of the metals in the ash. The end result is that the concentration of metals is relatively unaffected and the alkaline properties of the ash make them much less mobile when compared to their mobility in waste coal.

CFBC Ash as a Soil Amendment

The obvious goal during reclamation is to remove the waste coal from the environment. In practice, this is not that simple. The Mining Plan requires careful consideration of the impact caused by excavation on surface and groundwater monitoring wells. Experience has shown that metals concentrations, measured at the monitoring wells, sometimes increase in concentration as a result of disturbing the waste coal area. This result is both temporary and expected. The Mining Plan is used as a tool to minimize the impact of metals mobility during the excavation of the waste coal. In addition, a small amount of waste coal residue remains on the ground and the acidity of the soil area is significant due to many years of exposure to sulfurous acid runoff. The alkaline CFBC ash is returned to the site and mixed with acidic surface soil as a neutralizing agent. The neutralization of acidic soil with alkaline ash is the main benefit in combating the Acid Mine Drainage problem. The other significant benefit is more subtle. The elements in the waste coal, such as various naturally occurring metals, are also present in the ash. Since the waste coal consists of roughly 45% ash, these elements are returned to the site in the alkaline ash mixture. A simple material balance indicates that for every pound of waste coal removed, approximately 0.8 pounds of ash is returned to the reclamation site. The difference is primarily the carbon burned during combustion and the limestone added to control sulfur dioxide removal. The concentration of metals is theoretically slightly higher in the ash product but is still within the variability of concentrations found in native soils. There is an important distinction between the properties of the waste coal and the CFBC ash product concerning the mobility of metals.

An analysis of the Cambria CFBC ash was conducted in 1998 using the USEPA Test Method No. 1312, “Synthetic Precipitation Leaching Procedure.” This same procedure was applied to three native soil samples local to the Borough of Ernest. The three native soil samples were tested for comparison with the Cambria CFBC ash. The first soil sample was from a drill boring collected near the Ernest Borough garage. This sample provided the opportunity to examine soils which exist several feet below the ground surface. A second sample was taken from a nearby cornfield for comparison with cultivated soils. The third sample was collected from the ground surface in the

vicinity of the waste coal permit area to compare the ash sample results with surface soils from the wooded areas surrounding the Borough. Additional studies of the CFBC ash have shown that trace elements and acidity in waste coal samples exceed those in the ash product. This result was similarly demonstrated by Rose, Sheetz, *et al.*, in additional studies that provide the physical mechanism for this result. In qualitative terms, the ash demonstrates a superior ability to bind the metals in a more stable matrix relative to that exhibited by the waste coal. As a result, the levels of trace elements in the CFB ash product are comparable to those found in native soils, and in some cases, are actually lower than native soils.

The results from the 1998 Synthetic Precipitation Leaching Procedure test are given in Table 1 and Figures 3 – 6. The graphs were prepared to provide a visual comparison for each element that was analyzed. In addition, the Pennsylvania Department of Environmental Protection (PADEP) Standard was added to provide a comparison of the test results with the State upper concentration limits. The graphs also use a logarithmic scale on the Y-axis since the PADEP standard is often orders of magnitude higher than the test results for many of the elements. In some cases, the PADEP does not have an upper limit for some of the elements such as aluminum, cobalt, molybdenum, and potassium. The results of the leachate tests show another interesting result. In many cases, the concentrations of the samples are the same for each of the various elements analyzed. For example, the test results for cadmium show concentrations of 0.04 mg/l for each sample. The PADEP standard is 0.125 mg/l. This result is likely due to the sample concentrations being at or below the detectable limit of the analyzer or test for that particular element.

The overall result of the leachate test shows that the elemental analysis for the Cambria CFB ash is well within the range of concentration typically found in native soils. None of the element concentrations of the ash exceed the PADEP standard. Interestingly, some of the elements found in native soils exceed the PADEP standard. This result is observed for iron and manganese.

Table 1. Comparison of Leachate Results for CFBC Ash versus Native Soils

Metal	Cambria CoGen CFB Ash (mg/l)	Ernest Drilling Tailings (mg/l)	Ernest Area Corn Field (mg/l)	Ernest Area Top Soil (mg/l)	PaDEP Standard (mg/l)
Aluminum	0.32	0.62	36.3	13.1	NA
Antimony	0.041	0.003	0.003	0.003	0.15
Arsenic	0.004	0.003	0.008	0.006	1.25
Barium	1.25	0.2	0.2	0.2	50
Boron	0.119	0.1	0.1	0.1	78
Cadmium	0.04	0.04	0.04	0.04	0.125
Chromium	0.071	0.04	0.06	0.04	2.5
Cobalt	0.15	0.1	0.1	0.1	NA
Copper	0.053	0.04	0.07	0.04	32.5
Iron	0.21	0.27	30.5	14.3	7.5
Lead	0.2	0.2	0.12	0.12	0.375
Manganese	0.038	0.1	0.8	0.42	0.05
Mercury	0.0008	0.0004	0.0004	0.0004	0.1
Molybdenum	0.15	0.01	0.01	0.01	NA
Nickel	0.125	0.04	0.04	0.04	5
Potassium	6.87	1.9	11.3	3.6	NA
Selenium	0.022	0.002	0.002	0.002	1.25
Silver	0.04	0.04	0.04	0.04	1.25
Zinc	0.04	0.04	0.16	0.09	1.25

CFB Ash and Ernest Soils Soluble Metal Comparison

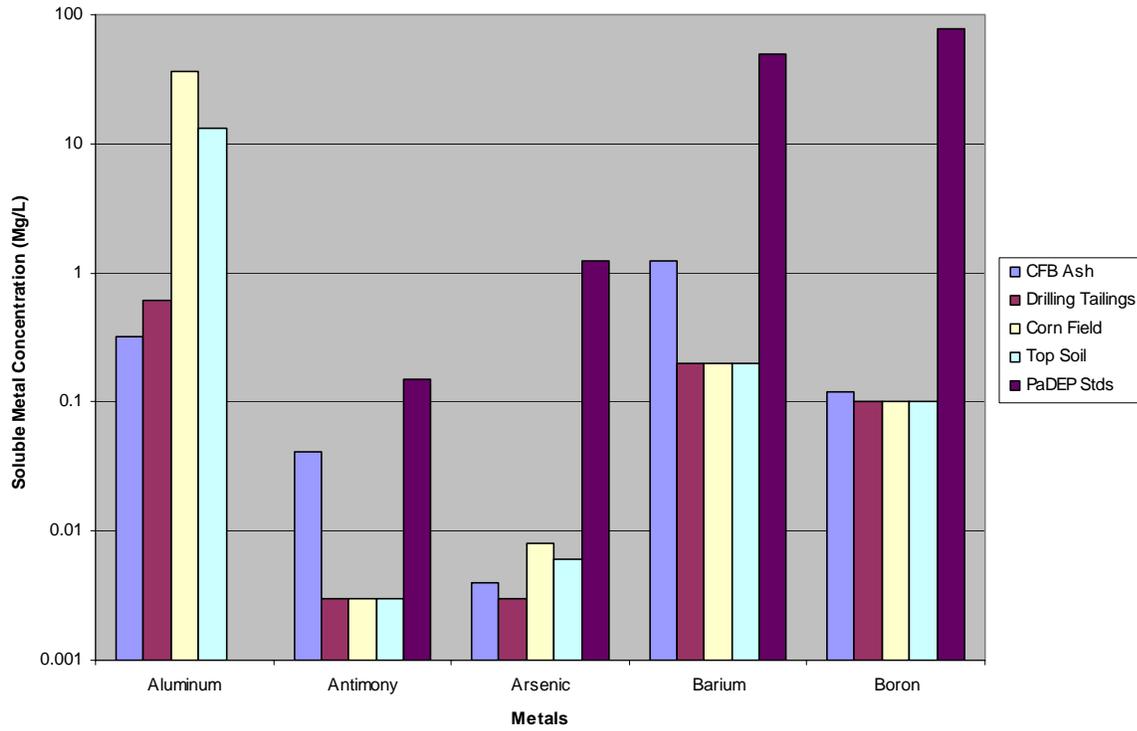


Figure 3. Soluble Metal Concentrations for CFB Ash versus Native Soils

CFB Ash and Ernest Soils Soluble Metal Comparison

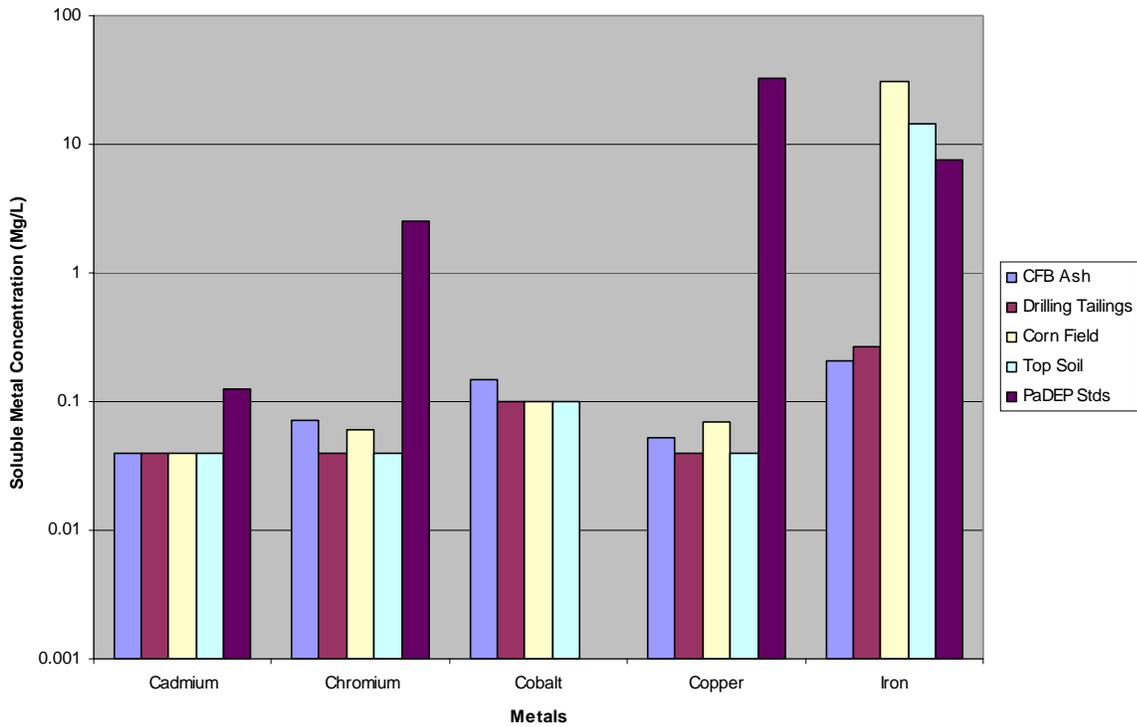


Figure 4. Soluble Metal Concentrations for CFB Ash versus Native Soils

CFB Ash and Ernest Soils Soluble Metal Comparison

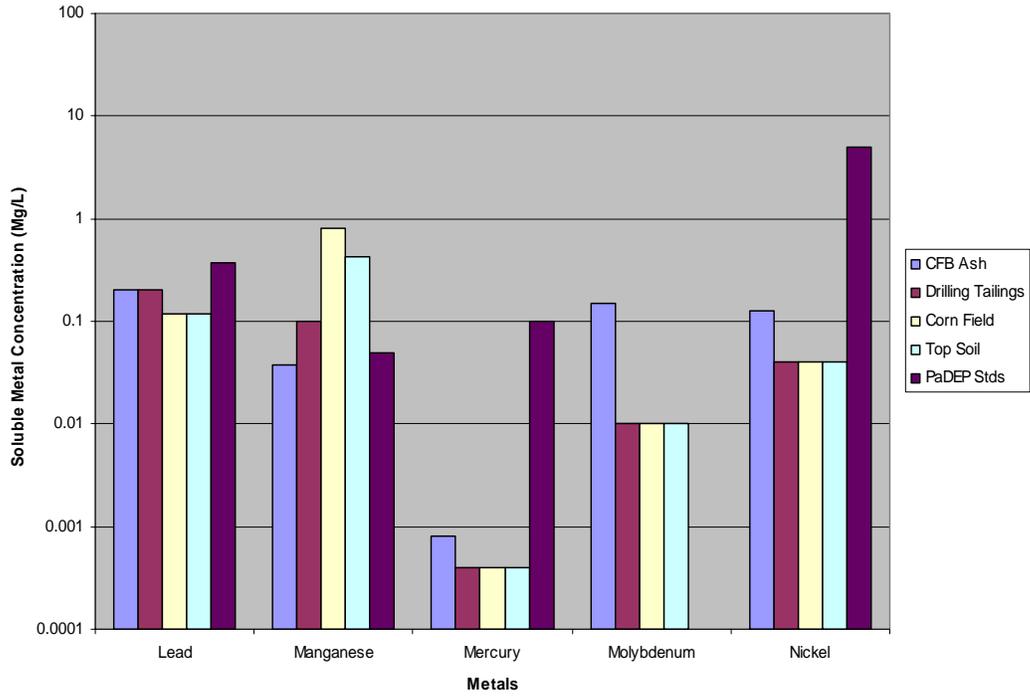


Figure 5. Soluble Metal Concentrations for CFB Ash versus Native Soils

CFB Ash and Ernest Soils Soluble Metal Comparison

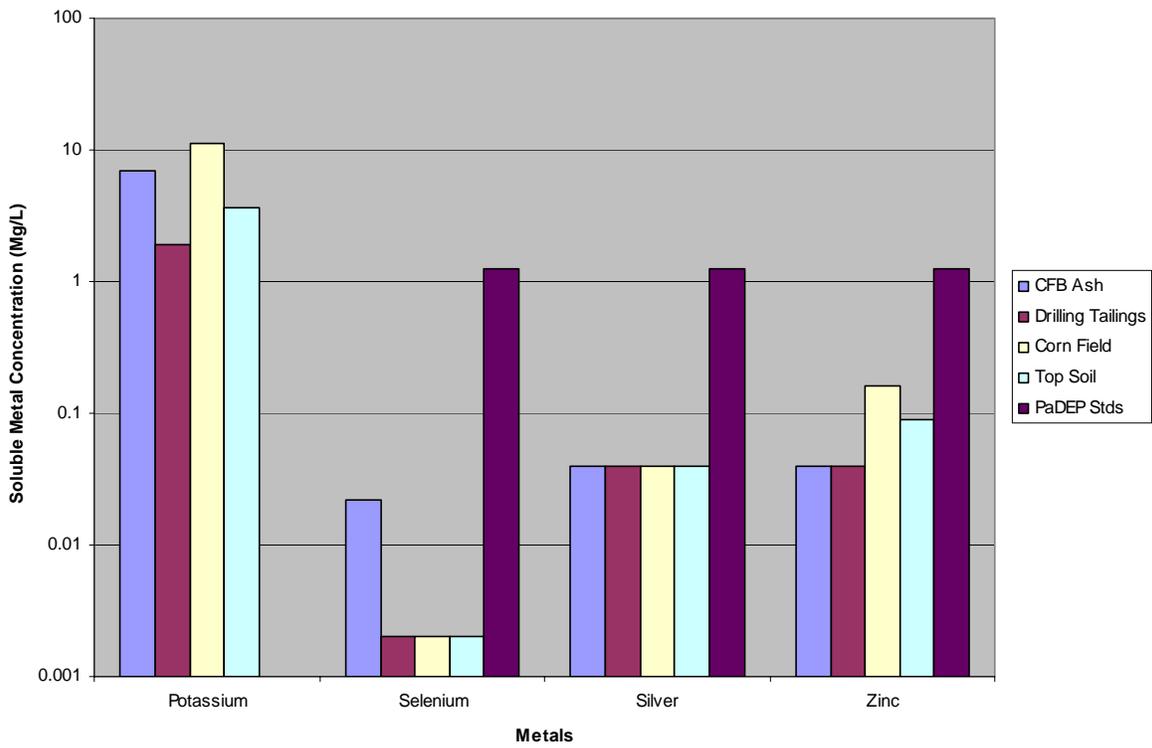


Figure 6. Soluble Metal Concentrations for CFB Ash versus Native Soils

Summary

The greatest environmental problem facing the State of Pennsylvania is Acid Mine Drainage and discharges. Decades of active coal mining resulted in the generation of millions of tons of acid bearing coal waste. Current estimates indicate approximately 250,000 acres of abandoned mine lands covering 43 of the 67 counties and adversely affecting over 2,400 miles of streams. The CFBC boiler technology has been successfully used by independent power producers to produce valuable electricity and steam while simultaneously removing the mountains of hazardous waste coal scattered throughout the state. In addition to removing the waste coal, the sites are reclaimed beneficially by using alkaline ash to neutralize the residual acidity of the waste coal surface areas. The ability of CFBC ash to bind the metals contained in the waste coal results in decreased mobility of these metals into the environment. Reclamation is completed by using topsoil as a cover and planting with native vegetation to restore the areas to a condition similar to premining conditions. The environmental remediation and site reclamation are conducted with tremendous savings to the citizens of Pennsylvania. The environmental improvements accomplished by Pennsylvania's waste coal plants are well documented and should be embraced by both Federal and State regulatory agencies.

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Session 1

USE OF FLUIDIZED BED COMBUSTION (FBC) MATERIALS IN MINING RECLAMATION

Session Chairperson:

Alfred Dalberto

Pennsylvania Department of Environmental Protection

Bureau of Mining & Reclamation

Harrisburg, Pennsylvania

Overview of the Fluidized Bed Combustion Process and Material

Dr. Francois Botha, Illinois Clean Coal Institute, Carterville, Illinois

Re-Mining and Reclamation of Large Abandoned Bituminous Coal Refuse Piles Using FBC Ash: A Watershed Restoration Perspective

Timothy Kania, Pennsylvania DEP, Ebensburg, Pennsylvania

2004 Update to ‘Occurrence and Fate of Selected Trace Elements in Circulating Fluidized Bed Combustion By-Products’

Dennis Noll, Earthtech, Johnstown, Pennsylvania & Arthur Rose, PhD,

Pennsylvania State University, University Park, Pennsylvania

Chemistry and Mineralogy of Coal Fly Ash: Basis for Beneficial Use

Dr. Barry Scheetz, Pennsylvania State University, University Park, Pennsylvania

Lessons Learned from Full-scale, Non-Traditional Placement of Fly Ash

Dr. Caroline Loop, Pennsylvania State University, University Park, Pennsylvania

Fuel and Ash Management for a Typical CFBC Facility

Stephen Skidd, Cambria CoGen, Evansburg, Pennsylvania

PERMITTING ISSUES ASSOCIATED WITH WET FGD PLACEMENT AT THE METTIKI MINE IN MARYLAND

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Mettiki Coal, LLC
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Abstract

Electric utility response to certain amendments of the Clean Air Act has resulted in the production of several types of alkaline coal combustion byproducts. Alkaline combustion byproducts are gaining increasing usage for acid mine drainage mitigation as research leads to a better understanding of their beneficial applications.

Since January of 1997, Mettiki Coal, LLC has been injecting alkaline flue gas desulfurization material from Dominion Resources Mt. Storm Power Stations' wet limestone scrubbers into abandoned portions of the active Mettiki mine. This paper provides an overview of the key regulatory, environmental, design, and placement issues faced during the project.

Introduction

Electricity constitutes a crucial input in sustaining the Nation's economic growth and development. Coal combustion has historically accounted for the bulk of electrical energy production in the United States, accounting for over 51% of the total net generation of electricity in 2003 according to the Energy Information Administration (National Energy Information Center, 2004). One of the concerns of fossil-fueled combustion is the emission of sulfur dioxide (SO₂) during the combustion process. Title IV of the Clean Air Act Amendments of 1990 was enacted to reduce the emissions of SO₂ in two phases. Phase I, running from 1995 through 1999, affected approximately 435 generating units while Phase II, which is more stringent than Phase I, began in the year 2000 and affected more than 2000 generating units. Though fuel switching had become the Phase I compliance method chosen by most utilities to meet these reduction requirements, flue gas scrubber systems have been installed on 192 generating units at seventy nine utilities and has accounted for 28 percent of the 1995 SO₂ emission reductions, the second largest share after fuel switching (Energy Information Administration, March, 2000).

All scrubbing units utilize a chemical reaction with a sorbent to remove SO₂ from combustion gases and are classified as either "wet" or "dry." In the most widely used wet scrubber systems, combustion gases are contacted with a sorbent liquid which results in the formation of a wet solid byproduct.

Additional Key Words: Coal combustion byproducts, flue gas desulfurization, hydroclones, underground injection, NPDES.

Most scrubber systems utilize an alkaline limestone sorbent, resulting in an alkaline calcium sulfite and / or calcium sulfate sludge byproduct. Approximately 25 million tons of these flue gas desulfurization (FGD) byproducts are being produced per year in the United States (EPRI, 1999). As increased cost of disposal and heightened regulations make disposal less desirable, alternatives to disposal are being investigated. Alkaline FGD byproducts are finding increased uses in environmental applications as extensive research provides a more comprehensive understanding of their benefits and behavior.

In November of 1994, Mettiki Coal, LLC (Mettiki) made application to the State of Maryland Department of the Environment (MDE), Industrial Permits Division for an underground injection permit modification to inject FGD material into abandoned sections of its underground mining operation in southwestern Garrett County. Material available for injection at that time was available from Virginia Power's Mt. Storm Power Station Unit #3 wet scrubber located approximately 17 miles away in Mt. Storm, West Virginia. In 2001, Units #1 and #2 were also "scrubbed" providing additional alkaline material availability.

Mt. Storm Unit #3 Scrubber

The Mt. Storm Unit #3 forced oxidation wet limestone scrubber is a General Electric Environmental Systems unit placed in operation in October, 1994. The SO₂ laden flue gas from Unit #3 enters an absorber vessel down stream of the precipitators and flows up through a spray of limestone (CaCO₃) slurry. The SO₂ is contacted by the spray and falls into a reaction tank below. The initial collection of SO₂ is primarily with water, but once the slurry falls into the reaction tank, the SO₂ reacts with excess calcium to produce calcium sulfite. Additional oxygen is provided to the reaction tank by oxidation air blowers resulting in a conversion of calcium sulfite to calcium sulfate (gypsum) (Figure 1). The reaction tank provides suction for the recycle slurry pumps, which continually pump slurry to the spray headers in the absorber vessel. For Mt. Storm Unit 3, approximately 100 gallons of slurry is sprayed into the absorber vessel for every 1000 ACFM of flue gas. As the larger gypsum particles settle in the reaction tank, they are pumped by the absorber bleed pumps to the waste dewatering system which consists of a bank of hydroclones and a drum vacuum filter. The hydroclones separate the gypsum slurry into two streams. The overflow stream, containing less than 5% solids, flows into a filtrate tank for recirculation back into the scrubber. The underflow stream, containing approximately 50% solids, is fed to the drum vacuum filters. The vacuum filters further concentrate the solids to approximately 80% solids with the resultant water also being recycled back into the scrubber. The byproduct solids are then temporally stored in an enclosed building sized to hold a 3 day supply of product where it is loaded into trucks for transportation to Mettiki for injection. Production averages approximately 400 tons per day (Figure 2).

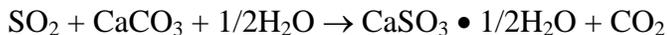


Figure 1.

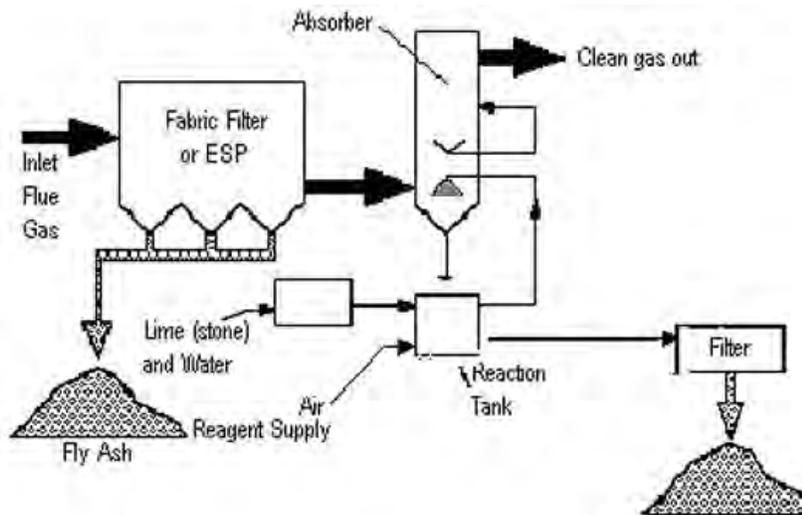


Figure 2.

MDE Regulatory Issues

In 1993, the Environmental Protection Agency (EPA) issued its final regulatory determination on FGD residues. They were deemed to be non-hazardous and therefore, regulated under Subtitle D of the Resource Conservation and Recovery Act (RCRA). This determination gave individual State's regulatory authority which can vary from State to State.

Based on available research data at the time, it was felt that FGD addition would assist Mettiki in maintaining an alkaline environment in its underground mine pool at closure and aid in preventing acid generation. Since 1987, Mettiki has been injecting alkaline metal hydroxide sludge from its 10 million gallon per day mine drainage treatment facility along with thickener underflow from its coal preparation plant under a MDE Underground Injection Control (UIC) permit. Though permitted under the Maryland UIC program, compliance monitoring and environmental impact assessment is managed through a National Pollution Discharge Elimination Systems (NPDES) permit.

Being the only permittee in the State to request the ability to inject FGD material - coupled with the fact that the material was not available in the State of Maryland at the time - added a level of complexity to permitting. Part of the problem faced by Mettiki was that coal combustion byproducts are not covered by any single regulatory unit within the State of Maryland.

In January of 1995, shortly after the preliminary discussions with MDE and the formal submittal of the permit modification, MDE requested a meeting of section heads from Solid Waste Management, Hazardous Waste, Underground Injection Control, and Mettiki to discuss which single department would regulate the injection and maintain compliance oversight and how that would best be accomplished.

Alkaline coal combustion byproducts are not considered hazardous materials under Maryland law. FDG in particular has its own line item hazardous material exclusion ((Code of Maryland Regulations 26.13.02.04-1.A(4)) and does not fail any of the required RCRA tests used to determine if it is a hazardous waste (Table 1). This excluded the material from MDE Hazardous Waste oversight.

Table 1. - Chemical Analysis - Mt. Storm FGD (mg/L)

TCLP DATA ¹		SOLIDS DATA	
Arsenic	<0.10	Calcium	186,000
Selenium	<0.20	Magnesium	685
Barium	0.15	Iron	273
Cadmium	<0.01	Aluminum	229
Chromium	<0.03	Potassium	<500
Lead	<0.10	Sodium	<50
Silver	<0.02	Zinc	<10
Mercury	<0.002	Chloride	6000
		Moisture	39.7 %
		pH	7.88

¹ Averaged analytical data. Tests performed with standard TCLP extraction fluid, raw mine water, and dilute sulfuric acid.

The fact that the material would not be landfilled, outside of the existing mining program laws and regulations, excluded it from Solid Waste oversight.

Since the material was chemically and physically similar to Mettiki's current injected materials, it was agreed within MDE that oversight could be best handled under Mettiki's existing UIC discharge Permit. A modification of the existing NPDES permit was also required to address and monitor what MDE felt could be a potential, despite TCLP testing, for dissolution of the material in the underground mine pool. Of particular concern to MDE were chloride levels. Accordingly, discharge limits based upon US Fish and Wildlife Service trout recommendations were set at 230 mg/L quarterly average and 850 mg/L quarterly maximum. Given MCC's cooperative trout rearing facility location and potential impacts to trout production, Mettiki agreed to the limitations and additional monitoring. Table 2 shows pre and post injection raw mine water analysis indicating negligible dissolution impacts and beneficial AMD reduction impacts to date. Figure 3 shows chloride levels in the mine pool in relation to injected tons of FGD.

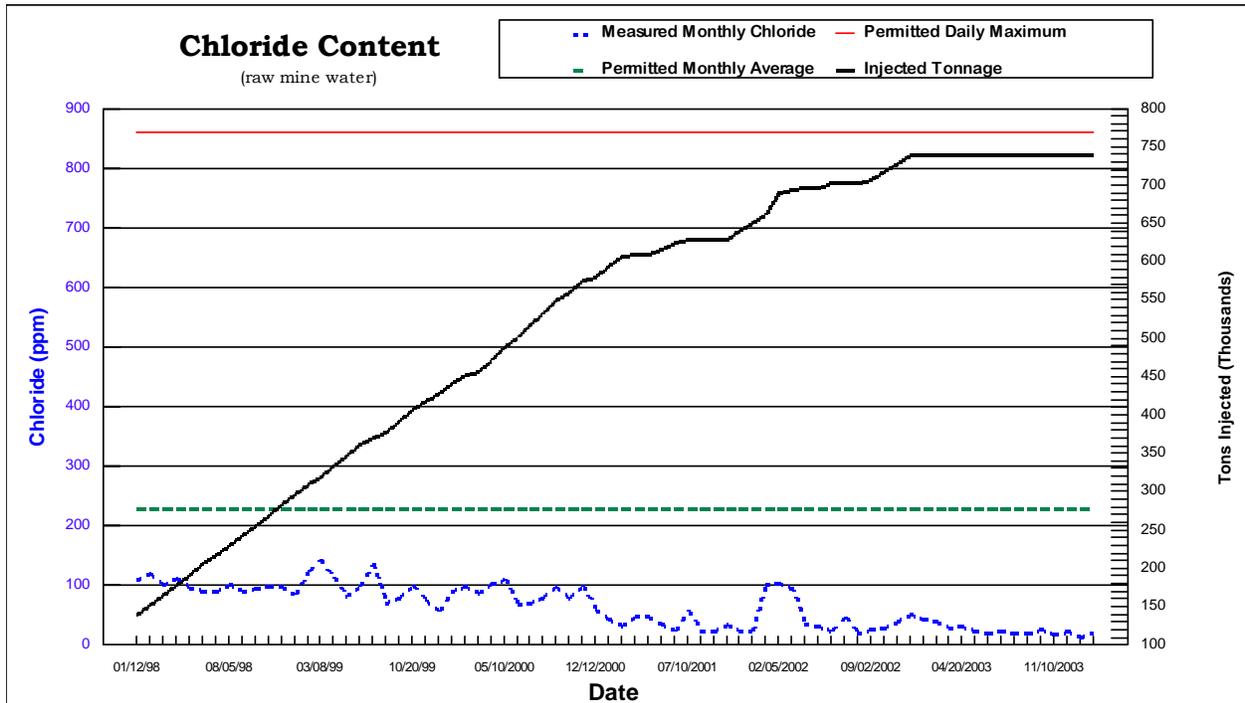


Figure 3. - Raw Mine Water Chloride Levels

Table 2. - Raw Mine Water Analysis (mg/L)

	06/13/96	07/10/97	04/18/00	02/04/03
pH	5.98	6.20	6.61	6.68
Al	0.4	1.32	0.81	0.331
Arsenic	<.025	<.025	.005	<.01
Antimony	<.05	<.05	.0084	<.02
Barium	0	0.033	0.0367	0.05
Berillium	<.0025	<.0025	<.0011	<.005
Boron	0.065	0.937		
Cadmium	<.0025	<.0025	<.00081	<.005
Calcium	224	541	533	226
Chromium	<.0075	<.0075	<.0075	0.01
Cobalt	0.1	0.137	0.0853	0.0566
Copper	0	0.0095	<.0029	<.01
Iron	37.8	34.4	31.8	1.32
Lead	<.025	<.025	<.0079	<.020
Magnesium	49.5	83.7	87.6	59.3
Manganese	2.72	4.8	2.95	2.47
Mercury			<.00010	<.00020
Nickel	0.139	0.195	0.132	0.0908
Potassium	7.43	10.2	8.97	7
Selenium			0.0085	<.010
Silver	<.0025	<.0025	<.0056	
Sodium	77.2	79.2	85.2	78.7
Sulfate	830	1345.7	1710	818
Thallium	<.13	<.13	na	<.020
Vanadium	<.0050	<.0050	<.0050	<.005
Zinc	na	0.266	0.227	0.05

The permitting process was fairly straightforward once the information, test results, and proper applications were submitted to MDE. MDE issued tentative determination in late January, 1996 and scheduled a public hearing for March. At the hearing, sixteen citizens appeared to voice concerns that injection of CCB's would cause subsidence and that any heavy metals bound up chemically in the material would automatically leach out and contaminate drinking water supplies, both surface and groundwater.

The public hearing lasted approximately two hours and no amount of technical information or detail seemed to allay the public perception of the material. Final permit issuance occurred in May 1996, approximately 19 months after initial application.

Bureau of Mines MDE Regulatory Issues

Concurrent with MDE UIC permitting, additional regulatory requirements under the Maryland mining laws enforced and permitted by the Maryland Bureau of Mines (BOM) division of MDE were pursued. A meeting with BOM personnel shortly before the UIC permit modification submission resulted in their request that Mettiki first submit a permit application to the Industrial Permits Section of MDE then immediately submit the required separate application to the BOM. Dual permits were submitted to each agency with similar but varying data needs supplied to each. In addition to the UIC data requirements, the BOM requested additional chemical information along with detailed information on the injection system design and monitoring program. The BOM application concentrated on the source material, type of facility and product, chemical characteristics and utilization plan. Additionally, the potential use of FGD in surface areas around the mine site and for backfill stability was addressed, along with how contamination of surface and groundwater would be monitored and prevented.

Underground Injection

To handle the additional injection material, Mettiki modified an injection system upgrade occurring at the time designed to carry Mettiki through the life of the current D mine reserves. To accommodate the delivery of the material to the site, Mettiki constructed an unloading facility with slurry water conveyed from existing deep well turbine pumps at the water treatment plant.

Once slurried at approximately 15% percent solids content - controlled by a nuclear densometer and Allen Bradley SLC 503 programmable logic controller - the material is pumped in buried high density polyethylene pipelines to a disposal surge tank at the water treatment plant. Tank level controls cycle two Warmen 10 x 12 discharge pumps arranged in series. Line velocities and the potential to sand out the line over the 14,000 foot distance to our B mine injection point required the high pressure, high volume pumps. Design capacity is 2,500 gallons per minute at 260 psi at the pumps. Vertical elevation difference between the pumps and the highest point in the disposal line is 250 feet with approximately 150 feet of elevation to work with in the mine voids. Ultimate placement is 600 to 750 feet below surface elevations. Storage capacity within B mine at current peak solid injection rates is 13 years.

Mine Pool Impacts

Water that pools underground is either stage pumped through the mine in Mettikis' active works or flows by gravity in the inactive portions (including the decant solution from the injection) to an underground sump and is then conveyed to the surface via a combination of one 800 hp Layne, one 800 hp Goulds, or two Peabody Floway 800 hp deep well turbine pumps and treated at the water treatment plant. Under normal conditions, flow rates of from 2000 to 12,000 gal/min. are maintained depending upon what pump or combination of pumps are placed in operation. Treatment options consists of one High Density Sludge treatment system, capable of treating 20,000 gal/min. and one Techniflo in line aeration system presently capable of treating 4000 gal/min.

Raw mine water enters the neutralization tank initially and is mixed with a hydrated lime slurry. The slurry is made from clarifier sludge and or clear water taken from the clarifier. Lime addition is controlled by dual Great Lakes pH probes located within the neutralization tank. The neutralized water is aerated in the same tank through two 8 inch PVC pipes using 15 hp Roots blowers in the HDS system. The aerated water then discharges through a sluice-way where polymer is added prior to entering the 200 foot Eimco clarifier for

precipitation of the hydroxide sludge.

The in-line aeration system differs from the above in the oxidization step. Oxidization is accomplished by an air inductor that entrains air by a venturi device which is powered solely by the pressure of the raw water pump. Post aeration treatment involves anionic polymer addition to aid flocculation of the metal hydroxides and clarification in a concrete 115 ft. by 14 ft. circular classifier.

Metallic hydroxide accumulation in the bottom of the clarifiers is raked and suctioned to the combined sludge disposal tank via two Warmen sludge pumps or by a centerwell pump in the circular clarifier. Final sludge disposal into old underground workings is accomplished by the two Warmen 10 x 12, 400 hp disposal pumps.

Conclusion

This project, though complex in implementation, is intended to quantify the benefits of CCB utilization and affords a unique opportunity to provide real-life data on CCB interactions with acid producing mine waters. The fact that there are no exits to the environment other than the deep well pumps and through MCCs' treatment facility offers a controlled environment to observe those interactions and potential benefits.

Public concern regarding management of coal combustion byproducts is founded in a belief that small quantity toxic constituents contained within the material could potentially damage human health and the environment. Public opposition can create major obstacles to beneficially using coal combustion byproducts for acid mine drainage mitigation and should not be underestimated. Though potentially toxic elements may be present in some materials and, at certain concentrations, these elements may have toxic effects, with approximately 750,000 tons of FGD material injected thus far, raw water heavy metals chemistry has remained similar to pre-injection conditions or improved. Continued monitoring to determine if FGD injection produces the expected long-term improvements in water quality is planned.

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Jim Ashby is the Manager of Environmental Affairs for Mettiki Coal, LLC, and Maryland's largest coal producer. He holds Bachelor Degrees in Biology and Chemistry from West Virginia Wesleyan College. For the past 20 years, he has focused on coal, water, and environmental quality issues in various capacities in the Utility, Mining, Laboratory, and Consulting fields.

THE USE OF DREDGED MATERIAL AND ALKALINE ACTIVATED COAL ASH IN ABANDONED MINE RECLAMATION: FINAL REPORT OF THE BARK CAMP DEMONSTRATION PROJECT

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Abstract

The human and environmental impacts of Abandoned Mine Lands are well known; falling, drowning and asphyxiation hazards, mine void collapses, stream and groundwater diversion, and the acidification of thousands of miles of waterways. They are the number one cause of water pollution in many States. Surface mines despoil hundreds of thousands of acres while underground mine fires burn for decades. The reclamation of America's 560,000 abandoned mine features will require billions of tons of fill. At the same time, environmental regulations seeking cleaner air and water are redirecting millions of tons of bulk materials to the land. Cleaner-burning coal means adding tons of lime during combustion, and collecting more of the ash that used to go up the stack. Mud dredged from navigation channels, dams, and reservoirs may no longer be disposed of in the ocean. This country annually produces 130 million tons of coal combustion products and 500 million tons of dredged materials. This project set out to demonstrate that dredged materials amended with alkaline activated ash wastes to undergo a cementitious reaction can be used beneficially to replace the devastated geology of Abandoned Mine Lands, removing physical hazards, returning surface waters to their watersheds, preventing the formation of Acid Mine Drainage, and restoring natural vegetation and habitat, without harm to the environment. Nearly one million tons of amended fill was used to return a double high wall along 11,000 feet of a hillside in western Pennsylvania to its original contours. Over 100,000 surface and groundwater tests have been performed since 1998. The permitting, monitoring, operations, and results of this project will be summarized.

Introduction

This report documents the successful evaluation of the beneficial use of nearly half a million cubic yards of dredged materials to reclaim an abandoned coal mine in central Pennsylvania. The project was undertaken to evaluate whether sediments from standard navigational maintenance dredging operations, containing metals and organic contaminants within regulatory limits, can be processed with alkaline activated coal ash to form a low permeability cementitious fill for mine reclamation with exclusively positive environmental benefits. It also demonstrates the feasibility of this application on a practical basis; the material can be handled, processed, treated, transported, and emplaced while keeping up with the production capacity of dredging operations. Port economies, which handle 95% of U.S. International trade, generate and spend billions of dollars to sustain these waterways. This project sought to join port economies, and the by-products of coal combustion and dredging, with the vast fill requirements of dangerous abandoned mine land features, using the rail lines that had originally brought coal to the ports.

Bark Camp Demonstration Project

Initiated in 1995, the Bark Camp Demonstration Project is a public-private partnership among the Pennsylvania Department of Environmental Protection, the New York/New Jersey Clean Ocean And Shore Trust (a bi-state marine resources commission), and Clean Earth Dredging Technologies, Inc (a Pennsylvania environmental contracting and recycling firm). The project sought to join port economies, the need to dredge navigation channels, and the would-be waste products of coal combustion and dredging with the vast fill requirements of dangerous abandoned mine land features.

The Pennsylvania Department of Environmental Protection (PADEP) has designated over 5,600 abandoned mine features as human health hazards in need of remediation. These features, responsible for several fatalities each year, include dangerous shafts, high-walls and submerged pits, 36 underground mine fires, 800 annual incidents of land subsidence over deep mines, 250,000 acres of unreclaimed mine lands, and the production of Acid Mine Drainage

(AMD) that impacts 3,000 miles of Pennsylvania's rivers and streams. Pennsylvania has individual strip mine features with fill requirements estimated at up to one billion cubic yards, while the total cost of this reclamation is estimated to be \$15 billion. Twenty-nine (29) US States and Tribal lands have catalogued over 560,000 abandoned mine land features. While many mine sites have millions of tons of coal wastes in local piles (overburden and rock originally surrounding the coal seams and removed during mining) much of these wastes are acid generating and should be replaced as encapsulated fill to prevent acid drainage. Other mines have had their spoil widely scattered, while some, like vertical outcrops, are massive voids that have no spoil associated with them at all.

An issue of similar scale is the disposal of the 500 million tons of sediments dredged on average each year from the nation's navigational channels, lakes, harbors and marinas. These volumes, which are mostly composed of water, silt, sand, and clays, were typically disposed of in off-shore waters, and generally contain trace amounts of contaminants from agricultural and industrial sources. They should not be confused with the minority of highly contaminated or hazardous materials associated with 'environmental dredging' operations from industrial sites that require decontamination, and which make up a small percentage of the total volumes dredged nationally. There is however legitimate concern over the disposal of even trace-contaminated materials in open water because loose sediments in aquatic habitats are literally consumed by bottom dwelling creatures, which over time may bioaccumulate contaminants and could provide their entry into the food chain. For this reason, dredged materials are increasingly being used world-wide in upland applications where they are solidified and used as structural fills with no threat to the environment.

While many upland uses of dredged materials have been developed, from the construction of port islands to airport runways and use in landfill closures, sufficient portside areas are not available for the volumes involved. The cost of dredging has skyrocketed, threatening to rearrange shipping patterns with dire consequences for traffic, employment, and consumer prices in the entire mid-Atlantic region. In other instances, shoreline placements of dredged materials have smothered valuable wetlands. On the Delaware River, the long standing practice has been to hydraulically pump dredged materials onto what were originally wetland areas of Delaware, New Jersey, and Pennsylvania, covering thousands of acres. The proposed deepening of the Delaware channel would generate 93 million yards of sediments, mostly destined for existing facilities in New Jersey, as well as an additional 670 acres of existing wetlands.

Another large scale problem is the US annual production of 130 million tons of coal ash, largely from coal burning power plants, which provide over 50% of the nation's electricity. These ashes are the mineral residue of coal and lime (added during combustion to reduce acid rain production) and the captured 'fly ash' that formerly exited from smokestacks. They have cementitious (or *pozzolonic*) properties and are similar to volcanic ashes used to perfect Roman concretes twenty centuries ago. Under alkaline conditions, these mineral products of burning react to form cementitious bonds. Modern coal ashes have been used in US construction since 1948, and 20% of the total ash generated is used in the manufacture of Portland cement. But the remainder is largely land filled or stockpiled. Their sheer volume and the portion placed in unconsolidated stockpiles are themselves causes of environmental concern.

Historic proposals for the use of abandoned mines often focused on the planned dumping of bulk materials simply as a means of disposal, in the absence of present day environmental sensibilities. However there is a real need for safe mine reclamation fill materials where such materials are not economically available, and for funding mechanisms to address that \$15 billion problem. Along with their ability to form cements, the contaminant binding properties of alkaline activated coal ashes are well established, making them an appropriate binder for dredged sediments, so as to form an extremely low permeability manufactured fill for the replacement of the underlying rock of stripped mine lands.

This project sought to demonstrate the potential for the combined beneficial use of these wastes, or byproducts of US shipping and power generation while leveraging the economies of scale of each of the problems addressed and directing their financial resources towards mine reclamation. The high cost of dredged material management along with ever increasing volumes of coal ash generated to reduce air emissions, have made their beneficial use for mine reclamation economically feasible. Over \$19 million was provided to accomplish this project, of which about \$7.5 million were spent at Bark Camp. The determination to be made was whether it was truly beneficial without negative consequences for the environment.

The Project

In 1995, the NY/NJ Clean Ocean And Shore Trust, a bistate marine resources commission known as COAST, approached the Pennsylvania Department of Environmental Protection (PADEP) with a proposal to test the feasibility of using dredged materials from the Hudson and Raritan estuaries and the Delaware River in abandoned mine reclamation. Facing the legacy of 300 years of coal mining, PADEP has been a national leader in mine reclamation research and applications. The department determined that in spite of negative public perceptions of dredged materials, the levels of contaminants involved were not excessive and well within their regulatory experience. In order to ensure that only acceptable materials were used, the Bureau of Land Recycling and Waste Management applied regulatory limits for contaminant levels from existing programs and forbade the use of any hazardous materials whatsoever.

The Bark Camp Mine Reclamation Laboratory

PADEP identified the existing Bark Camp Mine Reclamation Laboratory in Clearfield County as a good candidate for the demonstration site. It was an abandoned mine in a State Forest owned by the Commonwealth, with the responsibility for its reclamation having reverted to the State.

Bark Camp as a whole is an abandoned coal mining complex that included abandoned surface and underground mines, preparation facilities, and operating equipment, all with their attendant residual problems. Given that the facilities operated from the 1950s through the early 1980s, substances commonly used included electrical transformers, a variety of fuels, solvents, and other materials no longer in use today. Several reclamation and acid mine drainage abatement methods had been and continue to be tested and evaluated there, including the use of coal ash and municipal waste incinerator ash grouts.

The permitting process was begun and discussions were entered into with the nearby community of Penfield in Huston Township. The township governing board formed an Environmental Committee to work with PADEP as full partners to inspect and monitor site activities. In June 1997, PADEP approved an amendment to the existing permit for ongoing reclamation activities at Bark Camp for the inclusion of dredged materials.

Bark Camp Run is a small stream that runs northward down a narrow headwater stream valley for about a mile and a half before entering a six foot diameter culvert under a former coal processing facility. It emerges and continues another 1.75 miles to where it joins a larger stream. The entire inside rim of the valley upstream of the culverted area was strip mined about 50 feet above the valley floor to reach thin layers of coal outcropping there. A bench was cut about 100 feet into the hillside creating a vertical cliff or highwall. The pyritic overburden was simply dumped directly downhill, and has since become overgrown in trees and shrubs. The project area is that bench and highwall along the west side of the stream which stretches from near the stream's headwaters almost to the culverted area. Approximately one third of this stretch of highwall has a second bench above the first one at the downstream end. The reclamation achieved in this project involved replacing the stripped out overburden with the manufactured fill, placed in lifts until the original contours of the hillside were restored, then covered with topsoil and planted.

Since the stripped pyritic overburden ringing the valley lies below the project level and was not remediated, it has and will continue to impact the upper portion of the stream with acid drainage contaminants. Prior to the project, surface and groundwater in the project area exceeded Pennsylvania Chapter 93 water quality criteria and/or drinking water standards for lead, cadmium, aluminum, Iron, manganese, sulfate, pH, and phenols. The area's coal beds, exposed on the valley walls by the strip mine, tilt down northwestward at twice the pitch of the descending streambed, intersecting the valley floor at the processing area and continuing underground beyond it. Two deep mines had been historically tunneled into the hills on either side of the valley at the processing site, and were now flooded and generating 180,000 gallons of acid mine drainage daily into the stream below the former entrances, heavily impacting the downstream portion of Bark Camp Run. The use of materials with minimal contamination during the demonstration was never likely to further degrade the already impacted stream.

During the course of the project, many interested people visited the site and discussed the various activities and their merits. They represented a wide range of backgrounds and interests that included State and Federal agencies both within and outside of Pennsylvania, news media, elected officials (both municipal and state representatives), national and State environmental organizations and watershed groups, academic researchers, private business concerns, and

local citizens.

Analyses were regularly posted and available on the PADEP website during the course of the project.

Permitting, Sampling, and Analysis

Every step of the process, beginning prior to dredging through the processing and placement of dredged materials, and including all components of the fill materials used, was regulated by permits and approvals issued by the relevant State and Federal authorities. Dredged materials and all additives had to meet specific bulk chemistry and leachate testing standards to gain approval for their use. Prior to the acceptance of any sediments, core samples from proposed dredging projects were analyzed for bulk chemistry, volatile, and semi-volatile organic compounds, pesticides, PCBs, dioxins, and metals before being approved for use in the demonstration. A single proposed dredging project was rejected for exceeding allowable contaminant limits. All admixture elements were also required to meet regulatory standards.

In addition to the minimum testing standards for inclusion of any materials in the project, a series of confirmatory tests were also required to ensure that no unauthorized materials were being included anywhere along the processing line. Samples were regularly obtained at the port-side dredged material processing facility after pre-amendment and just prior to shipping and random samples were also obtained from rail cars upon arriving at Bark Camp. Samples were taken of the final pugmill mixture as well, before emplacement in the highwall. These samples were subjected to the entire chemical testing protocol for all analytes.

Over 50 surface and ground water monitoring points have been established in various places at the mine reclamation laboratory over the past decade. These include the monitoring of a mine pool, multiple acid drainage seeps, and a collection system beneath a lined area of municipal waste incinerator ash grout. Several of these were established for, or continued to be monitored during, this project. Surface water monitoring points were established in Bark Camp Run and its tributaries above, along, and downstream of the demonstration project, and 6 monitoring wells were drilled along its length at the toe of the lower highwall (rather than the standard 100 feet away) to monitor groundwater. A small ravine separated the project near its middle into two phases, with three of the wells along each phase. Additionally, domestic wells in the general vicinity of the off-loading site on the rail siding were monitored to detect any changes during operations. Water samples were taken at the monitoring points at first monthly, and then quarterly, beginning prior to operations on the site. The samples were tested for a comprehensive suite of organic compounds and metals, which are listed in the report appendix.

Operations

Coal Ash Amended Dredged Material

Dredging was accomplished by using clam shell buckets mounted on cranes. The sediments are grabbed from the bottom of the waterway and placed in hopper scows for transfer to Clean Earth Dredging Technology's (CTI) port side processing facility. Dredged sediments are over 60% water, and must be stabilized for shipping. After decanting excess water rising to the surface of the scows, the material is screened to remove debris and blended with approximately 15% coal ash (by volume) before being loaded into 110 ton gondola railcars and covered with tarpaulins for transport to the mine site. This addition of coal ash is sufficient to bind any free water in the material. CTI processed and shipped up to 4,300 tons of amended dredged materials per one-shift day.

Although Bark Camp was an appropriate site for demonstration purposes, it was never logistically ideal, requiring a long haul from the port and multiple re-handling of materials. On arrival at Bark Camp, the railcars were unloaded into off-road trucks for the nearly two mile trip from the rail siding to the processing pad at the mine site. The now pre-amended dredged material was further mixed with coal ash and lime kiln dust in proportions necessary to initiate cementitious reactions, taken to the high walls and placed in piles while they began the curing process. The material was then spread in one to two foot lifts and roller compacted. In this project, the fill was engineered to achieve: a minimum compressive strength of 35 pounds per square inch within 28 days (enough to support construction machinery); a permeability of less than 1×10^{-6} centimeters per second (nearly the low permeability of a clay cap) and; to withstand freezing and thawing cycles.

Each lift was placed in a width slightly narrower than the one immediately below it to reconstruct the original contours of the hillside. When that was accomplished, the surface was covered in approximately 18 - 20 inches of manufactured topsoil made from local shale, paper fiber cellulose, organic material from a vegetable tannery, coal ash, and lime. The surface was planted in a mixture of grasses.

The first manufactured fill to be placed was a total of 40,000 tons of coal-ash amended dredged material in May 1998. Placed on the upper bench, its side slope was covered and planted but the entire top surface was left uncovered to monitor the effects of weathering. The site was monitored for two years from initiation before additional materials were brought to the project. In two years of surface and groundwater monitoring, there was not a single detection of a volatile or semi-volatile organic compound, pesticide, PCB, or dioxin. No metals were detected other than those previously present due to mine drainage. Yet the flat expanse of bare shale and the severe drop of the highwall were diminishing through the reclamation efforts. The planted slope had established a lush growth, and there was no detectable change in the stream below the length of the project. A negligible amount of chloride (34 mg/L) from the salt water (sodium chloride) in the material was detected in the lowest elevation monitoring well, indicating that project impacts were being captured by the monitoring system. However, no chlorides were detected beyond background levels in the surface water monitoring point below the site.

Manufactured fill placement resumed in July, 2000. In May 2001, three years after the project began, and after the emplacement of over a quarter million tons of dredged sediments, a biological survey by the Pennsylvania Fish and Boat Commission showed over-wintering trout in the stream below and along the site. In spite of the fact that the entire Commonwealth is under a fish consumption advisory due to contaminants like mercury and PCBs, the fish sampled from Bark Camp Run met the standard for unlimited consumption. A total of 435,000 cubic yards of dredged materials were amended and emplaced by spring 2002, with a three month winter hiatus in operations.

Municipal Waste Incinerator Ash

As previously noted, there were ongoing reclamation projects on the site before this demonstration was proposed, including projects seeking low cost fill materials in significant volumes. Permits had been issued for the use of coal ash, manufactured soil and finally, municipal waste incinerator ashes (MWIA) as a cementitious fill material. MWIA went through a full permitting process in 1996 and was shown to exhibit low permeability and pozzolonic activation. Regional competition for qualifying, bidding, and contracting for dredged material delayed completion of this project long enough that MWIA began arriving at the site in April, 2001. CTI began amending the alkaline activated ash and dredged material mixtures with MWIA in May, placing most of that material in Phase Two and at another site away from the project. For several months in 2002 after final placement of dredged materials, MWIA grouts were used to complete the final sections of the Phase Two highwalls.

Physical and Chemical Processes

The factor that makes this application safe and beneficial for mine reclamation is the physical and chemical changes undergone by the constituent materials. More than 20 years of increasingly sophisticated scientific investigations have established an understanding of the long known ability of alkaline activated bituminous coal ashes to form very strong cementitious bonds. Cementitious properties occur among the mineral fraction remaining after coal combustion because they are converted by the heat of combustion, in the presence of lime injected to reduce the generation of acid rain, into highly reactive compounds with stored chemical energy. When mixed with water, they initiate cementitious reactions breaking their chemical bonds and forming new ones. Most of the water present (H₂O) becomes divided into oxygen (O) and hydroxyl (OH), and is chemically bound into the newly formed minerals of the hardening cement matrix. Both coal ashes and dredged sediments (sand, silt, and clay) are mostly composed of silicon, aluminum, and oxygen compounds, which along with calcium, are the core constituents of cementitious reactions.

Cement mineral structures or matrices are extremely tight and leave only tiny pore spaces that may contain some liquid water of a pH in excess of 12. Because of chemical reactions with calcium, this pore water remains extremely alkaline, and maintains a very strong buffering capacity, being able to neutralize acids over long periods of time. Several things happen to metals and organic compounds present during these reactions, including:

Precipitation: most metals that are in soluble form are transformed in the high pH environment into insoluble

forms that precipitate out of solution, just as iron dissolved by acid drainage precipitates out in streams when the mine drainage is diluted and neutralized by fresh water. Since the pore water has the capacity to buffer acids, these substances remain immobilized and less vulnerable to leaching out of the matrix.

Isomorphic Substitution: contaminants may be chemically incorporated into the new compounds formed as the solid mineral phase develops from the slurry, they may also stick to the surface of the new compounds or be absorbed into their three dimensional structure (*adsorption* and *absorption*).

Physical Encapsulation: contaminants are surrounded by a strongly bonded matrix from which they can not escape.

Metal and organic contaminants are therefore physically and chemically bound at the molecular level and are not released when the concrete matrix is broken up. Once a contaminant is incorporated into the mineral phase of the cement, that matrix must be chemically destroyed to release them. The specific regulatory testing that the manufactured fill in this demonstration must pass is the Toxicity Characteristic Leaching Procedure (TCLP) testing, where a sample is pulverized, then tumbled in acidic solution for 18 hours, the equivalent to an extremely long duration of exposure to acid rain attack.

Aside from these complex mineral interactions at work, a more commonplace effect is taking place as well: the massive reduction in surface area. When disposed of in the ocean, dredged materials are literally consumed in large quantities over time by mud-dwelling marine organisms; their digestive systems processing every grain of sediment, and any contaminants thus available. But when solidified into a low permeability monolithic cemented mass, the surface area that is available to chemical attack is infinitely reduced, in relation to the massive volume enclosed. The fine particle sizes of both ashes and sediments make for very low permeability materials when compacted, and water can only move through them at a very minute rate. When considering the ability of these mixes to buffer naturally mildly acidic rainfall, along with their extremely low hydraulic conductivity, they are calculated to remain stable through geological time.

Results and Conclusions

1. Analysis of sediments before dredging and along the processing train showed that trace contaminants within permitted levels were present in the fill material prior to placement. Yet in the more than five years of monitoring ground and surface water impacts after placement began, the substances of public health concern - PCB's, pesticides, volatile and semi- volatile organic compounds, dioxins, and furans- were not detected in any of the surface and groundwater monitoring points. Similarly, metals remained at the levels present before the project and were not impacted by the manufactured fill. No hazardous levels of substances were ever detected in regular confirmatory and random sampling of transported materials.
2. The demonstrated effects were predicted by an extensive body of research and are due to: (1) the well established physical and chemical binding properties of pozzolonic materials; (2) the low permeability of the fill; (3) a relatively low level of commonplace contaminants in the manufactured fill constituents; and (4) the small surface area to volume ratio of the restoration. Correctly proportioned blends of dredged sediments, coal combustion ash and kiln dusts, properly applied, will not leach contaminants to ground or surface waters due to their inherent physical characteristics and the chemical bonds formed upon their proper blending.
3. This demonstration has proven the feasibility of this application on a practical basis; the material can be handled, processed, treated, transported and emplaced while keeping up with the production capacity of dredging operations.
4. The only statistically significant water monitoring impact detected over the course of the entire project was the appearance of chlorides from common salt (sodium chloride) in the marine dredged materials, which fluctuated in relation to project activities and demonstrated the effectiveness of the water monitoring plan. Chlorides were expected due to the release of some salt water from marine dredged materials during handling, and a period of surface washing off the hardened material. But elevated chlorides were correlated with the use of municipal waste incinerator ash (MWIA) as a pozzolonic amendment in the later stages of the project and its placement as a grout, its use having been permitted prior to this project.
Much of the material dredged in the United States comes from fresh water sources, this single outstanding effect would not be a consideration in the use of that material.
5. At the height of activity during exclusive placement of dredged material fill, chloride levels in the area of Bark

Camp Run affected only by the project (and not a source of drinking water) briefly reached 44 mg/L, well below impact levels for fish and other aquatic organisms. During placement of MWIA grout, chlorides briefly exceeded the EPA suggested drinking water standard (for taste and not health reasons) of 250 mg/L in a single round of testing (282 mg/L) and then declined. While such projects are typically done near acid drainage impacted waters that require time to recover after remediation, a short period of elevated chlorides would be inconsequential. However, projects in potentially sensitive freshwater areas must be designed and managed to take this phenomenon into account, employing appropriate sediment and runoff management. Careful mix design and project management can reduce the amount of free water remaining un-bound by hydrating reactions in the cured material, thereby reducing any mobilization of chlorides. PADEP will continue to closely monitor the project to quantify this trend.

6. Analysis of Domestic Wells in the vicinity of the rail siding where materials were off-loaded indicate that, removed from the project and the site, there is a source of contamination originating at some far distance away from the siding and migrating toward it. The affected wells are all within the influence of multiple residential sewage discharges and several are within the influence of a large farm field that has had contamination issues in the past. Further, the wells closest to the railroad, indeed the one directly below and adjacent to the unloading area, have lower values of detected elements.
7. A dangerous high wall in a State forest, adjacent to State game lands was eliminated. Water is now flowing overland to the stream rather than back into and along the highwall. Flat expanses of bare shale and pyritic rock have been restored to a meadow habitat frequented by bear, deer, elk, bobcat and turkey. The survey of Bark Camp Run by the Pennsylvania Fish and Boat Commission in May 2001, three years after the project began, cited significant water quality improvements with increasing numbers of macroinvertebrate taxa and some common fishes at a downstream station in Bark Camp Run which was formerly sterile due to the mine drainage impacts left behind by the bankrupt mining operation.
8. The survey further reported over wintering trout in the upper section of Bark camp Run, directly below the fill project area. Pennsylvania has a State-wide precautionary one meal per week fish consumption advisory due to the prevalence of trace contaminants in the environment. And while there is a one meal per month advisory for PCB contaminated fish and a two meal per month advisory for mercury contaminated fish, all the fish tissue samples from Bark Camp met the standards for unrestricted consumption, including for mercury and PCBs.
9. Community outreach and participation were vital to the success of this project. Local communities must be worked with closely and included in the projects to dispel misconceptions and build cooperation. As would any major reclamation project, this work also provided significant employment and financial resources to the host area.
10. The appearance of moderately elevated chlorides during this demonstration corresponds with the introduction of municipal waste incinerator ash (MWIA) into the manufactured fill as a pozzolonic material. About 253,000 tons of MWIA was placed in Phase 2 of this demonstration, and an area separate and distinct from the dredge demonstration project. MWIA ash is known to contain sodium chloride and as such, the manufactured fill would be expected to release a finite amount of chlorides as well. Chlorides are commonly released as brines from gas wells in this portion of the State, (which is underlain by a vast salt-dome) and are widely applied during winter road-salting operations, and a period of elevated chlorides does not pose any problems at Bark Camp. The extent and degree to which the chloride levels have increased in various monitoring points at Bark Camp over time with the placement of material containing MWIA, indicates a clear need for caution in the use of this material in a similar project. PADEP, therefore, has decided that Municipal Waste Incinerator Ash will not be considered for use in mine reclamation projects. Any other potential use of this material would require a more extensive review and separate examination with the appropriate permitting agency which is not within the scope of this report.
11. The key to the successful use of coal ash amended dredged materials is thoroughness. The capabilities of properly made ash mixes were utilized in ancient times and over the last 70 years. Over 80% of the surface and groundwater analytes tested for, at significant cost, were reported as undetected. The proper characterization of raw materials, and the imposition and monitoring of appropriate performance criteria for compressive strength and low permeability, along with sound project design and operations, are more important than continually analyzing bulk chemistry for contaminants during operations.
12. The use of dredged materials, amended with alkaline activated coal ash, is found to be feasible, safe, and beneficial for use in abandoned mine reclamation.

This report was compiled from the work of many individuals over several years. The full, referenced report is available online at www.nynjcoast.org.

Andrew Voros has been the Director of a bi-State, marine and coastal resources commission between the State legislatures of New York and New Jersey for the past 7 years. This commission proposed and coordinates the project using dredged materials for the reclamation of abandoned mines in the Commonwealth of Pennsylvania. He has degrees in biology and psychology from Rutgers University in New Jersey and earned a Federal Law Enforcement Commission from the Dept. of the Interior, where he worked as a Park Ranger in Yosemite National Park. He served two years as a Peace Corps Volunteer and spent 8 years working in tropical rainforest conservation in West Africa.

ALTERNATIVE METHODS OF ASH PLACEMENT AT A NON-COAL MINE SITE IN OKLAHOMA

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Abstract

In 1991, the Oklahoma legislature enacted Title 45 § 940 of the Oklahoma Statutes. This legislation stated that “*All fly ash, bottom ash or any other such material produced by coal combustion generating facilities shall be exempt from all solid waste permitting requirements pursuant to Title 63 of the Oklahoma Statutes, provided that such ash is constructively reutilized, or disposed of in any active or inactive coal or noncoal mining operation subject to the provisions contained in Title 45 of the Oklahoma Statutes.*” Authority for regulating this unfunded program was given to the Oklahoma Department of Mines thus giving the operator the option of utilizing coal combustion by-products in their mining operation.

In 2001, the Oklahoma legislature appropriated funds to the Oklahoma Department of Mines (ODM) to regulate mine sites that were utilizing coal combustion by-products for reclamation purposes. The primary focus of the Coal Combustion By-Product (CCB) Program in Oklahoma, which includes the placement of ash and cement kiln dust, is the reclamation of mine sites. These sites are issued a mining permit under the Minerals Division of ODM and reclaimed by having CCB placed in designated portions of the site. Despite having the funds available, there were still no specific rules established to regulate the placement of CCB into mines sites.

In 2003, this changed with the adoption of the Coal Combustion By-Product Standards (OAC 460:30). These new rules set forth guidelines for the use of CCBs in reclamation. Operators must also adhere to the previously established Non-Coal Rules and Regulations (OAC 460:10). All mining and reclamation applications are technically reviewed for environmental design prior to the permit being issued to ensure the compliance with all applicable rules and regulations protecting public health and the environment. In addition, regular inspections are conducted and periodic water and ash analysis are required in every permit.

Site Descriptions

Permit L.E.-1857 was issued to Making Money Having Fun (M.M.H.F.) in April 2001 and utilizes what we consider an alternative method to ash placement because the ash is not directly placed in the pit. At this site, the ash is mixed with water in a “Portable Fly Ash Hopper” that is up gradient to the pit and the mixture is gravity-fed through an open ditch down to the pit. The following discussion further elaborates on the processes involved in this operation and the environmental effects this operation has had to date.

Generalizations will be made throughout the course of this discussion and comparisons made between permitted sites and aspects of various other sites are highlighted for reference purposes. As an example of a typical site, even though the placement methods are considered an “alternative method” more attention will be paid in this paper to the permitted site and how it is regulated under the current CCB Standards (OAC 460:30). M.M.H.F. #2 (L.E.-1857), on which we will focus, is the second fly ash placement site to be operated by M.M.H.F., L.L.C. M.M.H.F. #1 (L.E.-1685) is not currently accepting ash and the one active cell on this permit area has received all required cover and is currently establishing vegetation. A partial bond release has already been conducted on this site.

In Oklahoma, all mines that accept ash are permitted as “Non-Coal” mine sites no matter what the history of the property. All permitted CCB sites have been issued life expectancy permits, but are renewed annually and constantly being reviewed and revised as needed. ODM currently permits ten (10) sites across Oklahoma that accepts CCBs and each one was issued a mining permit with the Minerals Division of ODM. The CCB Placement, i.e. Fly Ash, Program is operated under the auspices of the Minerals Division. Nine (9) of these sites are permitted to receive ash and one (1) of these sites is permitted to receive Cement Kiln Dust (CKD). L.E.-1857 has been issued

a permit to mine clay and shale. However, clay and shale have never been mined under this permit and as soon as the permit was issued, the site immediately began reclamation with the acceptance of ash.

L.E.-1857, consisting of 333.0 permitted acres and 149 bonded acres, is located south of the town of Bokoshe in Sections 16, 17, 20 and 21 Township 8 North Range 24E LeFlore County, Oklahoma and was originally permitted in April 2001 to receive fly ash and bottom ash produced at the AES Shady Point Power Plant near Panama, Oklahoma through fluidized bed combustion. Panama is approximately 10 miles east of this permit area. This site has an estimated life expectancy of 25 to 35 years and was estimated in the original permit to accept between 500 to 1500 cubic yards of ash on a daily basis. The original permit established that ash would be placed in an area an average of 520 feet wide, 100 feet deep, have an average length of 4,800 feet, with a total volume of 9.24 million cubic yards and reach a final elevation of 550 feet above sea level. The site is divided in the permit into three (3) ash disposal cells (ADCs). ADC #1 and ADC #2-#3 are currently separated by a large dike made of ash and ash is currently being placed in ADC #1. A floating pump is currently located in ADC #2-#3 and water in ADC #2/#3 is pumped to the portable fly ash hopper for the mixing process.

Alternative Methods

The method in which ash is received at L.E.-1857 is an alternative to the standard method of dumping the ash directly into the cell. Ash is placed into the ash disposal cell by mixing it with water. A "Portable Fly Ash Hopper" is utilized to mix the water and ash. The ash is brought in by dump trailers and either belly or end dumped on the grate located over the two (2) feet deep hopper. Simultaneously, water is pumped into the hopper through small pipes and spray nozzles at a rate of 4,400 gallons per minute. Each dump trailer carries 25 tons of ash. It is estimated that it takes approximately 4 minutes to unload the trailers. The daily water requirement is dependent upon the amount of ash to be placed each day. Assuming the ash is unloaded in the hopper eight (8) hours each day, the amount of water needed to wash the ash down to the ash disposal cells is calculated to be 2,112,000 gallons per day. Of course, this figure varies depending upon the time of year and the amount of ash available for placement. The site accepts ash on the average anywhere from four (4) to six (6) days a week. After the trailers unload the ash and it mixes with the water, it is diverted by a combination of simple ditches down to the portion of the cell that the operator wants it directed. The ash flows freely in the water and eventually settles out on the bottom of the cell. The ash is compacted with normal equipment movements. This primarily consists of a dozer to push the ash and a track-hoe to cut the diversion ditches within the cell.

As stated above, ADC #1 and ADC #2-#3 are currently separated by a large dike and ash is currently being directed into ADC #1. When ash was first placed in this cell, it was diverted directly below the hopper, but as time went on ash was diverted to the southeast end of the cell away from the hopper. Since then, "fly ash" ditches have been moved as needed in order to evenly distribute the ash throughout the cell. The height of the dike is increased as more ash is placed in ADC #1. This height will continue to increase until the final approved elevation of the cell is reached. Water in ADC #2-#3 is replenished through directed runoff of the surrounding area and direct precipitation events. However, recently due to lower water levels in ADC #2-#3 and the ash level in ADC #1 rising the water in this cell close to the top its spillway on the southeast end, a ditch has been cut between ADC #1 and ADC #2-#3 on the east end of the dike between the cells to allow water from the southeast end of ADC #1 to flow into ADC #2-#3. This allows water to be recycled through the hopper again. This ditch cut will at some point be replaced with a pipe between the two (2) cells. Eventually water will be utilized from another impoundment located on the property, but currently outside the bonded area.

Coal Combustion By-Product Standards

CCBs were exempted from all solid waste permitting requirements in 1991, pursuant to Title 63 of the Oklahoma Statutes as long as such was constructively reutilized, or disposed of in any active or inactive coal or noncoal mining operation subject to the provisions contained in Title 45 of the Oklahoma Statutes. Since that time, the Oklahoma Department of Mines has dealt with operators that exercised their option of utilizing coal combustion by-products in their mining operation without funds or rules regulating such actions. That partially changed in 2001 when the Oklahoma legislature appropriated funds to the ODM to regulate mine sites that were utilizing coal combustion by-products for reclamation purposes and thus establishing the CCB Program. However, even though inspections were conducted and site analysis was underway, there were still no specific rules addressing these sites in place. This all changed in 2003 with the adoption of the Coal Combustion By-Product Standards (OAC 460:30).

These new rules, in combination with the previously established Non-Coal Rules and Regulations (OAC 460:10), give ODM the necessary tools that are needed to ensure regulatory adherence of the legislature's actions.

All new permits issued are subject to the guidelines of OAC 460:30. However, the existing ten (10) permits had to request that they be placed under these rules or be grand fathered under their existing permit. Due to the fact that these rules were compiled based on looking at the practices of other States and Federal Agencies, scientific data and, most importantly, several discussions with current permittees, to date, most of the preexisting permit holders have requested and are currently operating under these new rules. Those that changed saw it as an advantage in many aspects of their operation to be regulated under these rules and chose to amend their respective permits to include such information. Even though these rules are by no means lengthy or exhaustive as compared to the voluminous amounts of environmental regulations adopted throughout the environmental regulation arena, for the sake of brevity, the following discussion highlights the different aspects of these rules. It should be noted that local conditions and site-specific circumstances are taken into consideration for each permit. L.E.-1857 now operates under these new rules.

OAC 460:30-3 (CCB Placement Applications) addresses a broad range of issues pertinent to the operation of a CCB placement site. First, basic liability insurance requirements are set forth followed by environmental resources information such as the consideration of cultural, community and historic resources that would be affected by the operation. Geologic information from drill logs, permeability tests and maps are a part of the permit. For example, OAC 460: 30-3-4(b) establishes that the permeability of the *in situ* material, or any liner material approved by ODM, on the sides and bottom of the proposed CCB placement site must not exceed 10 to the minus 5 centimeters per second. Soil descriptions, soil maps and soil permeability tests, general operational information, health and safety accommodations, and sources of CCB are all included as the basics in a CCB placement permit.

This Subchapter addresses initial hydrologic resources, such as surface and ground water data gathered prior to placement which are essential in the monitoring and assessment of the effects of the CCB placement upon the water in the area of the operation. A set of "pre-placement" parameters has been established to give ODM baseline data on the water quality in the area of each individual permit. These parameters measure both the physical and chemical composition of the water. Since both ash and CKD sites are permitted under these rules, it was necessary to differentiate between those types of operations. In regard to hydrologic information, the parameters are basically the same, but certain parameters that were more prevalent or of environmental concern for either ash or CKD were set forth in the section addressing the respective CCB. For example, TSS, Total Iron, Total Manganese, Fluoride, flow in streams (surface), flow rate of discharges of impoundments (surface), impoundment freeboard (surface) and impoundment depth (surface) are analyzed in water from ash sites but not from CKD sites while Chloride, Potassium, Sodium, Antimony, Beryllium, Mercury, Nickel, Silver and Thallium are analyzed in water from CKD sites but not ash sites. Parameters including pH, Depth to Water (ground), TDS, Specific conductance, Sulfates, Temperature, Arsenic, Barium, Cadmium, Chromium (total), Lead, and Selenium are analyzed in water from all CCB sites. Logs and cross sections of wells drilled for the collection of ground water information and inventory of the chemical composition and completion data available from other reporting agencies or other sources for all ground water wells within one-half mile of the permit boundary are all to be included in the permit. A minimum of one (1) up-gradient and two (2) down-gradient monitoring wells are required at each site with the specific number of wells being determined by ODM on a case-by-case basis.

After the permit application undergoes ODM review and the site is permitted, water is analyzed on either an annual, semi-annual, or quarterly basis depending on the activity level at the site. The same parameters that are analyzed in the initial application are analyzed in the periodic submissions of water monitoring reports. All sites conduct water monitoring on an annual basis, sites that are active, inactive, and in temporary cessation conduct water monitoring on a semi-annual basis and sites that have never accepted CCB only conduct water monitoring on an annual basis. ODM has reserved the right to require additional monitoring when justifiably necessary. Water monitoring is conducted until the release of all performance bonds.

Initial applications contain a complete analysis, including: (1) Toxicity Characteristic Leaching Procedure (T.C.L.P.) And (2) chemical analysis of pH of the CCB to be placed in the mining permit area. CCB analysis, for both ash and CKD, will also be conducted biennially and annually. A T.C.L.P. is conducted biennially. Additional analysis is necessary if the source of ash changes for permits that accept ash and if the manufacturing/production processes that effect CKD changes in permits that accept CKD. Parameters that are examined in T.C.L.P. analysis in ash permits are: Arsenic, Barium, Cadmium, Chromium (total), Lead, and Selenium. Parameters that are examined in T.C.L.P.

analysis in CKD permits are: Antimony, Arsenic, Barium, Beryllium, Cadmium, Chromium (total), Lead, Mercury, Nickel, Selenium, Silver, and Thallium. Chemical analysis of pH is conducted annually. Additional analysis is necessary if the source of ash changes for permits that accept ash and if the manufacturing/production processes that effect CKD change in permits that accept CKD.

All permits include a variety of maps that establish locations for permit boundaries, landowner boundaries, structures within the permit area and within 1000 feet of the permit area, surface and subsurface man made features within or passing through the permit area, public road easements and cemeteries within 100 feet of the proposed permit area, as well as the boundaries of any public park, wildlife refuge, or recreational area within or bordering the permit area. Environmental resource maps, such as geologic, hydrologic, and soils maps indicated above, surface and ground water monitoring points, active, inactive, reclaimed or abandoned surface and underground mines, surface water bodies within and adjacent to the permit mining area, existing areas of mining waste, and water, oil or gas wells within the mining area are added to the permit. Potentiometric, cross-section, flood plain, and any other maps found to be pertinent to the particular site are also included in the permit. Operational and reclamation plans require that maps be submitted in the application that include buildings, roads, facilities, and utility lines to be used. The detailed operation plan according to the sequence of CCB placement is required; as is each topsoil, cover material, water storage, treatment, discharge and diversion facility, air pollution control facility, sedimentation pond, and permanent impoundment to be used. A contoured map showing the final surface configuration of the disturbed area is also included in the application submission.

OAC 460:30-5 (Performance Standards) sets forth the minimum environmental protection performance standards to be adopted and implemented for mining permits that include CCB placement as part of its reclamation plan. Adequate signage and plugged drill holes are addressed as well as the salvage of topsoil and the protection of water quality in the area. Flood-prone areas are identified and if applicable, effluent discharge standards contained in the permittee's National Pollutant Discharge Elimination System (NPDES) permit are addressed. Diversions are constructed to minimize the contribution of suspended solids to surface water flow outside the mining permit area. As discussed above, the surface and ground water is monitored based on a schedule corresponding to the activity on site. Provisions are included for wetlands, impoundments, waste control, fugitive dust control, intermediate cover, revegetation, temporary cessation of operations, roads, and utility installations. These Standards require that the final cover be a minimum of one and one-half (1.5) feet thick plus have a minimum of six (6) inches of topsoil or other suitable material as approved by ODM and that the final slope of CCB placement areas not exceed twenty-five percent (4:1). Copies of OAC 460:30 (Coal Combustion By-Products Standards) can be obtained by contacting the Oklahoma Department of Mines at (405) 521-3859.

Permit Specifics

L.E.-1857, like most permits issued by ODM has a similar format. However, each permit will vary based on local, geologic, cultural, environmental and other site-specific factors. To gain a better understanding of the basics of an ODM CCB Placement Permit, L.E.-1857's permit will be broken down into its general permitting format.

Prior to the permit being issued, the application is reviewed by ODM administrative and technical staff to ensure that the required permitting standards are being met. This includes everything from bonding issues to specific environmental requirements as established in OAC 460:10 and OAC 460:30. Written Findings are issued to answer the basic permitting questions and deficiencies are noted. After ODM finds the application to be complete the applicant next must post a public notice of the application in a local newspaper and notify nearby landowners that the application is available for review. After the review period is complete, if any citizen has requested a public hearing on the application, such will be held. Such a hearing was held prior to the issuance of L.E.-1857 and citizen concerns were addressed. This public hearing also became a part of the permit. This entire process can take several months to complete. As a side note, public hearings are only held again if the applicant requests a revision to the permit after publication and notification. The permitting application process is not unlike most application processes in other States.

L.E.-1857's permit begins with a general information requirements such as a description of the proposed operation, legal and ownership items, description of the adjacent area, population, roads and properties, operation hours and equipment used, amount and types of wastes to be accepted, life expectancy of the site, and proof of publication. General operational standards are described including the how the CCB will be handled, the employees will be kept

safe, the environmental resources such as air and water will be protected and monitored, and the reclamation standards and site closure will be accomplished. Many of the items in the CCB Standards, if applicable to the site, are addressed in this section. General technical information including the geology, soils, ground and surface water hydrology of the area are discussed. L.E.-1857's permit focuses next on surface drainage, the *in situ* liner and the characteristics of the ash. Tables and figures describing the geological features of the area, vegetation and diversion requirements of the site, water quality monitoring, and CCB analysis programs are detailed. L.E.-1857 currently submits semi-annual water monitoring reports with an expanded list of parameters analyzed annually and annually submits ash analysis reports. Drill logs for ground water monitoring wells and permeability tests and inventory of all wells and usage within one-half mile of the permit area are recorded. L.E.-1857 has also sent out Notices of Intent to the Oklahoma Department of Environmental Quality (ODEQ) in regard to water and air quality issues and consulted other governmental agencies in regard to licenses, permits, and clearances on issues related to those agencies. ODM requires the applicant to obtain all necessary licenses/approvals prior to a mining permit being issued. As required by the CCB Standards, maps addressing all aspects of the permit are submitted with the application. Design specifications for impoundments and diversion channels are included in this permit as well as detailed calculations estimating the cost of reclamation.

Reclamation at L.E.-1857

The Reclamation Plan at L.E.-1857 states that of the 333.0 acres in the permit area, 315.0 acres will be reclaimed as a pasture and 18.0 acres will be reclaimed as a water reservoir. As expected, the area will be filled up to the 550 feet elevation, leveled, sloped and graded at a 4:1 slope to allow the adequate drainage of water. Ditches and terraces will also be constructed to properly direct the water. Haul roads will be left for access around the site and an on-site office/maintenance building will remain on the property for future use. Guymon Bermudagrass, Common Lespedeza, Tall Fescue, Perennial Rye Grass, Weeping Lovegrass, Redtop, Italian Ryegrass, Winter Rye, Sudan grass and Wheat will be utilized as permanent and Temporary Species. Water monitoring will continue until the site is released by ODM.

As stated above, all ODM permitted sites include an estimated cost of reclamation. The original bond posted for this site was for the placement of ash in ADC #1. It was estimated that 58 acres would be covered with overburden and a total of 73 acres would be topsoiled and revegetated. Given all the particulars associated with reclaiming this site, including but not limited to the estimated expense for cover material handling, the bond in the original application was estimated at \$224,057.00. The bond amounts for L.E.-1857, as bond estimates in general, are subject to change based on periodic bond reviews, permit revisions, and alterations to the reclamation plan for this site.

pH Trends at Selected Sites

ODM does not have jurisdiction over enforcing water quality standards, but in the instance that a water quality standard is exceeded over time, ODM will notify the appropriate agency of such occurrence. ODM requires CCB placement permit holders to analyze: (1) 18 parameters at ash site and 21 parameters at CKD site ground water monitoring points; and (2) 15 parameters at ash site and 21 parameters at CKD site surface water monitoring points. However, the pH at some selected permitted sites was reviewed for the purpose of this discussion to show that, to date, pH levels at CCB placement sites in Oklahoma on the average have been staying within their respective range of limits. The initial pH analyses for the water monitoring points at L.E.-1857 were taken in March 2000. In the three plus years that data has been gathered at this site, it is difficult to detect any long-term trends and draw any conclusions. However, on the average, the pH on the up-gradient ground water monitoring point has slightly decreased while the pH on the two (2) down-gradient ground water monitoring points have slightly increased. At the same time, the pH on the five (5) surface water monitoring points has shown a slight to moderate increase. The pH parameter, along with the other parameters analyzed at L.E.-1857 will continue to be monitored to ensure environmental compliance.

The Brazil Creek Shady Point #1 site has already been reclaimed and the bond released. The Brazil Creek Shady Point #2 site is currently permitted to receive ash and is considered for permitting purposes to be active. However, it has not received ash for a few years. Shady Point #2 lies directly east of Shady Point #1. Water monitoring data was collected at Shady Point #1 from March 1992 to January 2001 when it was closed. Records were kept on several parameters, but pH is noted here as varying little during this time period. In fact, the average pH in up-gradient GWMP1 only slightly increased during this time and the average pH in down-gradient GWMP3 only slightly decreased during this time. Water monitoring data was collected at Shady Point #2 in the 4th Quarter of

1993 and then again since the 1st Quarter 2001. The average pH in up-gradient GWMP9 has slightly increased during this time and the average pH in down-gradient GWMP10 has only slightly increased during this time.

L.E.-1685, M.M.H.F.'s first permitted site, last received ash May 2001. Water monitoring analysis was conducted at this site six (6) straight months from October 1995 to March 1996. It was then collected again beginning with the 4th Quarter 2000. The trend in the average pH for up-gradient GWMP3 is a moderate increase over this time. The trend in the average pH for down-gradient GWMP1 is a slight increase over this time.

It can be deduced from the 10 plus years of pH data gathered from the water monitoring points at L.E.-1685, the Brazil Creek Shady Point sites and L.E.-1857 that, to date, the effect of reclaiming these mine sites with ash has minimally affected the pH. Of course, there has been the occasional spike in the analysis, but for the most part the trends remain constant for both up-gradient and down-gradient water monitoring points. Further data will be gathered and analyzed as it becomes available and based on the trends to date, little change is expected.

Conclusions

Thanks to the Oklahoma legislature, ODM now has the tools it needs to monitor and regulate the placement of CCBs at CCB sites throughout Oklahoma. The adoption of the Coal Combustion By-Product Standards (OAC 460:30) in 2003 set forth the guidelines that permitted sites in Oklahoma are now utilizing. Even though most CCB sites are very similar in terms of basic permitting requirements, local conditions play an important role in the construction of a permit and the reclamation possibilities at each site. L.E.-1857 is no different than other permitted sites except that this site utilizes an "alternative" method of ash placement. The mixing of the ash with water as it is placed into the ash disposal cell allows the ash to be more evenly dispersed throughout the cell and cuts down on the equipment movement time. The only limiting factor is the availability of water.

L.E.-1857's permit is only a few years old and there is not a lot of data gathered yet. However, early indications, using pH as the indicator, are that the ash being placed at this site is having a minimal effect on the groundwater. This trend is also seen in the three (3) other permitted Oklahoma sites mentioned here. As more data is gathered it will more accurately reflect the actual effect ash is having on the water in the area of these sites.

From the discussion above, it is obvious that the new CCB Standards and the permit description of the L.E.-1857 permit are not all-inclusive, but in general CCB permits in Oklahoma follow an extensive outline to ensure compliance with all applicable rules and regulations protecting public health and the environment. This includes monthly environmental inspections conducted on all sites and the periodic submission of water and ash analysis. The Coal Combustion By-Product Standards are a great starting point for achieving these compliance goals, but these Standards and the regulation thereof, must be fluid and adaptable in order to meet the ever-changing environmental needs.

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HISTORICAL PERSPECTIVE OF COAL COMBUSTION WASTE MANAGEMENT AT BASIN ELECTRIC ANTELOPE VALLEY STATION, DAKOTA GASIFICATION COMPANY, AND COTEAU PROPERTIES MINE, BEULAH, NORTH DAKOTA

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Introduction

Conceived during the 1970s energy crisis, the Dakota Gasification Company's Great Plains Synfuels Plant (the first and largest commercial coal gasification plant in the United States), the Basin Electric Antelope Valley Station power plant, and the Coteau Properties (Coteau) Freedom Mine (the largest lignite mine in the United States) comprise an energy/industrial complex with a unique perspective on the management of coal combustion material. Located north of Beulah, North Dakota, both energy plants use coal from the Coteau Properties (Coteau) mine, Dakota Gasification Company (DGC) burning the coarser coal components and Antelope Valley Station (AVS) burning the finer coal components. Management, however, of the coal combustion material from the two plants differs in respect to both disposal and beneficial use.

Due to differing waste characteristics, the design and operation of the landfills differ. Both plants have landfills developed in former mine areas. The landfill requirements under the State's rules, however, allow flexibility in design and operation as necessary to ensure environmental protection. The lessons learned in waste management, over the last 20 years at the DGC coal gasification facility, are helpful for future consideration of this developing technology.

The differing waste characteristics also influence the beneficial use of the materials. The AVS dry ash/FGD waste is similar to many power plants in the U.S. The beneficial use of the AVS ash, both in the Coteau mine and at the current DGC landfill, provides cost-effective and environmentally sound solutions for the respective users. While all of the DGC gasifier ash is disposed in landfills, a significant portion of the fly ash/FGD product from the AVS plant is beneficially used by the Coteau Mine for soil stabilization and by DGC for waste solidification and leachate control. The beneficial reuse of coal combustion byproducts is gaining acceptance in North Dakota pursuant to guidelines developed by the North Dakota Department of Health (1). North Dakota's experience with beneficial use of coal combustion material has wider implications for many potential users and for the industry.

Background

The near surface Tertiary lignite deposits north of Beulah, North Dakota are interbedded with significant sequences of clay, silts and sands. The existing bedrock topography has been altered somewhat by glaciation. A veneer of Quaternary glacial deposits overlies soft bedrock materials and the area is somewhat dissected by deeply incised glacial meltwater channels underlain with fluvial deposits. Groundwater in these fluvial deposits can be locally important for farm and irrigation use. The two coal-fired facilities discussed in this paper are located over such a meltwater channel, the Antelope Valley, using coal mined from the adjacent uplands.

The Coteau Properties Mine excavates the uppermost lignite sequence in the uplands adjacent to the Antelope Valley. The limited groundwater at the base of the upland lignite seams is disrupted by mining activities. The groundwater levels slowly reestablish themselves after mining, resulting in a highly mineralized, alkaline saturated zone at the base of the mined area. Prior to plant construction, it was envisioned that primary waste disposal activities would be confined to the less environmentally sensitive mining area, affording greater environmental protection to the Antelope Valley's groundwater and surface water resources.

The North Dakota Solid Waste Management Rules for coal combustion waste disposal follow the general model of the Resource Conservation and Recovery Act (RCRA) Subtitle D criteria for municipal waste; however, the specific requirements are tailored for North Dakota’s geology, climate, and the coal combustion waste materials (1). The North Dakota Solid Waste Management Rules, Chapter 33-20 of the North Dakota Administrative Code (NDAC) promulgated pursuant to North Dakota Century Code, Chapter 23-29, includes appropriate standards for various types and sources of solid waste. Coal combustion waste is included in the definition of “special waste.” The criteria for special waste disposal facilities include:

- Location Standards
- Operation Requirements
- Design Requirements
- Groundwater Monitoring/Corrective Action Requirements
- Closure/Postclosure Requirements
- Financial Assurance Requirements

Antelope Valley Station (AVS) Basin Electric Power Cooperative

The Basin Electric Power Cooperative Antelope Valley Station (AVS), with two electric generating units, each rated at 450 megawatts, produces approximately 2000 tons per day of fly ash/FGD waste and bottom ash. While some of the fly ash/FGD material is beneficially used for stabilizing haul roads and equipment storage areas in the Coteau Mine and for waste solidification and a drying agent within the Dakota Gasification Landfill, most of the AVS waste is disposed in a landfill (permit SP-160) within a former mine area. The facility also has an earlier closed landfill (permit SP-025).

Groundwater reestablishing at the base of the mine spoils near the AVS SP-180 landfill typically has a total dissolved solids (TDS) concentration ranging from 4000 to 6400 mg/l (2). The SP-160 disposal facility does not have a formal liner; however, the area was developed with 100 feet of compacted clay-rich spoil between the waste and the base of the spoil. Careful control of surface water runoff (contact water) in the landfill helps minimize contact with waste. A plastic lined pond within the disposal cell collects contact water which is periodically sprayed on the disposal areas for dust control. Fly ash/FGD waste material is generally placed in layers and is covered with bottom ash and/or sprayed with water to control dust. As areas are brought to grade, the plant utilizes “sequential partial closure” to incrementally bring filled areas to grade and then close or reclaim them, reducing the open area and reclaiming the site. Closed areas are carefully sloped, with cover typically exceeding the minimum requirement (5 feet) of the North Dakota Solid Waste Management Rules.

All special waste disposal facilities have upgradient and downgradient groundwater monitoring wells. It is useful to compare a leachate analysis of Basin Electric’s FGD/fly ash with groundwater in the area of the landfills. The ASTM D-3987 extraction procedure used for the fly ash analysis cited below utilizes 20 parts of deionized water with one part of fly ash, a test for ash preferred by the North Dakota Department of Health over the EPA - TCLP test. The following tables compare the ash and spoils groundwater (2):

Basin AVS Fly Ash/FGD Analysis*	Spoils Groundwater, AVS 2001
• TDS 3590 mg/l	TDS 4,759 - 6,836 mg/l
• Sulfate 1960 mg/l	Sulfate 2,750 - 4,100 mg/l
• Sodium 229 mg/l	Sodium 474 - 993 mg/l
• Boron 1.86 mg/l	Boron .3 - .71 mg/l
• Selenium .039 mg/l	Selenium .002 - .1 mg/l
• Mercury <.0002 mg/l	Mercury Not analyzed
• Arsenic .002 mg/l	Arsenic .002 - .004 mg/l
• pH 11.9	
*Method ASTM D-3987 shake extraction B 20:1 water to waste 2/2000	

The groundwater at the base of the spoils is similar to the leach analysis of the FGD/fly ash with elevated dissolved

solids, high sulfates and sodium. Boron levels are higher in the ash analysis compared to groundwater at the base of the spoils, consistent with other coal ashes in the State. Selenium and arsenic in the spoils and in the ash are not at excessive levels. The pH of the liquid from the ash analysis was 11.9.

The earlier SP-025 landfill for AVS was located closer to the edge of the mine, closer to the base of the spoil, and closer to the Antelope Valley. The SP-025 landfill had a formal 4-foot liner constructed to specifications to protect groundwater resources. The later, larger SP-160 landfill is located further in the mine, away from the Antelope Valley and is separated from the base of the spoils by about 100 feet of compacted clay-rich material. Due to the combination of the better location, separation from the base of the spoils, the fine, dry waste material, and the operational measures (sequential closure, contact water management, etc.), this landfill was not required to have an engineered liner.

Dakota Gasification Company Great Plains Synfuels Plant

The Great Plains Synfuels Plant, now owned by Dakota Gasification Company (DGC) is based on a similar, but even larger facility near Sasolburg, South Africa using Lurgi gasifiers, German technology utilized by the Nazi's to address their fuel shortages. Development of the gasification plant near Beulah in the early 1980s was undertaken by Great Plains Gasification Associates, a consortium of energy and pipeline companies with loan guarantees by the U.S. Department of Energy (DOE). The collapse of energy prices in the 1980s and other financial issues resulted in the sale of the facility in 1988 to Dakota Gasification Company, a subsidiary of Basin Electric Power Cooperative.

The DGC coal gasification plant utilizes about 16,000 tons per day of coarse lignite coal (3) placed into pressurized gasification chambers into which oxygen and steam are injected at controlled rates. Incomplete oxidation derives a synthetic gas which goes through various cleanup and methanation processes. About 160 million standard cubic feet of syngas per day is compressed and sent via pipeline, connecting to the Northern Border Pipeline (3). The gasification process also generates various byproducts.

Byproducts are a significant source of revenue for the plant. They include anhydrous ammonia, ammonium sulfate fertilizer, cresylic acid, phenol, krypton/xenon inert gases, naphtha, liquid nitrogen, and carbon dioxide (CO₂) gas (3) (4). Waste gas, primarily CO₂, is marketed via pipeline to oil companies in Saskatchewan, Canada, to enhance oil recovery. Injection of CO₂ affords greater oil field efficiency and also helps trap or "sequester" some of the gases and sulfides in deep geologic formations, reducing CO₂ impacts to the environment. The enhanced oil recovery process is favorable and future sales may be possible in the oil fields of North Dakota, Montana, and Wyoming (4).

The manufacture of ammonium sulfate fertilizer is the result of a unique scrubber developed by DGC to reduce SO₂ emissions. This process, approved by the North Dakota Department of Health to resolve SO₂ emissions, is the result of over \$100 million spent to enhance air quality and provide a valuable fertilizer (DakSul45) which is marketed by the company (3) (4).

While gasifier ash is the main waste stream, coal gasification generates a wider variety of waste streams than a power plant, including coal tars, liquid waste streams, API separator sludge, and other wastes, much of which is now reinjected into the gasifiers and is reclaimed to make additional gas (5). The gasification process can also utilize other materials for fuels including shredded tires, refinery wastes, and other hydrocarbon-rich materials (4) (5).

DGC also has other waste management facilities to handle the broader variety of wastes. Three surface impoundments constructed with composite liners of 3 feet of clay overlain by a synthetic liner, a secondary synthetic liner with a leachate detection/collection system between the two liners handle waste liquids. This more sophisticated liner/leachate detection-collection system helps protect the Antelope Valley Aquifer.

For some waste water treatment and disposal, DGC uses some unique facilities. The plant was designed with a liquid waste incinerator to manage various liquids; however, this unit has been deactivated. Two 6,400-foot deep injection wells dispose waste liquids into the Minnelusa-Kibbey Formation. The plant also utilizes microorganisms in the large cooling towers for treating and breaking down organic constituents, reducing odors, controlling waste waters, and recycling cooling water.

DGC has three landfills located within portions of the mine area. The first permitted landfill, SP-100, is located north and east of the plant and is now closed. The second, now closed landfill regulated under Permit SP-101, is located immediately east of the plant. The latest, larger landfill (SP-109) is further east, well into the mine area, further away from the Antelope Valley and the associated aquifer, thus affording greater protection of that groundwater source (5).

In addition, during the early years of operation, the plant used a “temporary” storage area for gasifier ash and sludge management. Located on the edge of the mine, close to the Antelope Valley, this temporary operation presented some challenges; ash has been removed and the site is now undergoing corrective action. This initial storage area is now incorporated into the area of the second Permit SP-101 landfill. Gasifier ash has cementitious properties (7) but physically resembles a clinker or coarse aggregate, differing in physical appearance and properties from finer fly ash/FGD products typical of many coal-fired power plants. Gasifier ash exiting the bottom of the 14 gasifiers at DGC is quenched in water which also sluices or transports the ash to decanting bins. Once the ash is drained or decanted, quench water is directed to two basins where fine particles settle out prior to recirculation. This black “quench water” has an extremely high pH, typically about 13.8, and is super-saturated with dissolved solids. The quench water has elevated levels of some heavy metals. During early operation, the ash handling system and the ash waste stream at DGC presented a number of challenges which will be discussed later. Residual quench water contained in the fine ash sludge and gasifier ash strongly influences leachate quality in the facilities landfills.

A comparison of historic laboratory analysis of DGC’s gasifier ash, compared to recent analysis (2003) of leachate from the DGC SP-109 landfill, shows an interesting contrast (5):

DGC Gasifier Ash Analysis* -1990-98		DGC Leachate Analysis*	
Arsenic	.038 - .067 mg/l	Arsenic	34 B 68 mg/l
Selenium	<.003 - .05 mg/l	Selenium	<0.01 B 4.9 mg/l
Barium	1 - 1.6 mg/l	TDS	170,000 B 204,000 mg/l
Cadmium	<.003 - .013 mg/l	pH	12 B 13.5 (avg about 13.4)
Chromium	<.011 mg/l		
Lead	<.056 mg/l		
Mercury	.0003 - .0007 mg/l		
Silver	<.011 mg/l		
TCLP analysis (20:1 acidic water to waste) of the coarse, dewatered gasifier ash		*From DGC 2003 monthly landfill reports, NDDH Permit SP-109	

Arsenic in the ash water (leachate) is about three orders of magnitude higher than the Toxicity Characteristic Leaching Procedure (TCLP) ash analysis and selenium is up to two orders higher. While the applicability of the TCLP test for predicting leachate quality resulting from alkaline ash in a monofill is questioned by the NDDH and many scientists, the predominant problem of highly mineralized landfill leachate appears to be primarily due to the interaction of normal precipitation with the quench water contained in sludge periodically cleaned from the ash water settling basins and, to a lesser extent, the quenched ash. Co-disposed with ash, the sludge comprises only about 10 percent of the total waste stream; however, the coarser ash or “clinker” has no buffering or absorptive capacity and also contains residual quench water near field capacity. Early operations experienced significantly higher levels of quench water in the ash and in the landfills due to problems both in the ash handling system and waste management practices (5).

Early Gasification Waste Issues

At its inception in 1984, the Great Plains plant pioneered large-scale coal gasification in North America. Unforeseen problems in the ash management/quench system resulted in wet ash and generation of excess liquids and sludge, challenging plant operations and environmental staff as well as the State environmental regulatory programs. Modification in waste handling practices, redesign of some of the ash handling system, and the changes in landfill design and operation took some years to implement.

The uniqueness of the system combined with the following factors: (1) plugging of the ash decantation bins; (2) accumulation of excess fine material in the settling basins; (3) difficulties in dewatering both coarse ash and fine ash in the basins; and (4) the design of the ash handling area resulted in elevated quench water content in the waste streams and spillage in the plant area and haul roads. The lack of an appropriately designed landfill was also an issue.

In the early years, poorly decanted waste stored in the temporary storage area resulted in the accumulation of quench water in a holding pond close to the edge of the Antelope Valley. Spillage of ash quench water in the plant and haul roads was also an issue. NDDH staff worked closely with the facility to ensure prompt attention and modification.

To facilitate improved management of ash, sludge, and quench water, the Department made repeated inspections and revised permits (2-year permits) with conditions requiring: (1) routine training sessions for operations staff and (2) monthly reports to the Department on ash management issues. Modifications to the screens in the decantation bins improved dewatering of the ash. Modifications in sludge management to remove more water from the fine ash sludge in the settling basins helped reduce liquid levels in that waste stream. Areas of the ash handling area were paved and a concrete sump was built to facilitate cleaning. Strong emphasis on spillage control and cleanup also solved problems.

Impacts from the storage of ash liquids from 1984 to 1987 are still being remediated via a French drain placed between the Antelope Valley and the former storage area. This example of corrective action, as codified in the current North Dakota solid waste rules, shows how problems can be addressed after facility closure.

The first landfill (now closed), currently regulated by the NDDH under permit SP-100, received gasifier ash and sludge, off-spec sulfur from the Stretford Unit (earlier air pollution control equipment) as well as API separator sludge and other waste streams. Problems with management of sludge and excess liquids were still apparent. While impacts to the groundwater at the base of the spoils is not apparent, recent placement of recovery wells in the landfill sump for removal of quench water/leachate should help reduce the potential for leachate migration. This is another example of pro-active measures by the company. The second major landfill for Dakota Gasification Company, regulated by the Department under permit SP-101, is located immediately east of the plant in a portion of the mine relatively close to the Antelope Valley. Constructed with a 5-foot compacted clay liner, a graded base to promote leachate flow, and a synthetically lined sump, this second landfill adapted some elements of a leachate drainage system to help to manage liquids. Closed since 1994, this facility has been more aggressively dewatered than SP-100. In 2003, nine years after facility closure, about 224,000 gallons of leachate was removed from the closed landfill as part of the postclosure maintenance requirements. All removed liquids are recycled to the plant as makeup water in the ash quench system. Since closure, leachate levels have declined considerably.

Current Issues

The third landfill was placed into operation in 1994 under Permit SP-109 and is currently in use by DGC. This facility is in an even better location, further within the mine, away from the Antelope Valley, in clay-rich spoil materials. The lowest portion of the landfill is elevated above the base of the spoils by over 20 feet of compacted clay-rich material. In addition, a carefully engineered 4-foot compacted clay liner at the base and sides of the landfill helps minimize leachate migration. While some of the liner was damaged by frost effects and desiccation cracks, repair of these areas following NDDH Quality Assurance/Quality Control Guidelines (6) along with prompt protection of the liner from adverse frost and drying conditions helps restore the integrity of the liner system prior to usage.

The base liner of the SP-109 landfill is graded to central drainage “V”s which, along with piping and coarse granular drainage material, promotes concentration of leachate into main sumps. The coarse nature of the gasifier ash facilitates drainage of leachate along the base of the carefully sloped liner. Vertical pipes placed through the waste materials provide pump access for leachate removal (5).

Initial operation of the SP-109 landfill revealed a significant design flaw: the leachate collection piping was not directly connected to the leachate sump access pipe in first phase of the landfill. Shortly after the onset of operation, the access pipe plugged and liquids accumulated in the landfill. Repeated efforts to open the pipe were unsuccessful. The company eventually adapted by removing leachate through the vertical riser pipes, relying on some of the redundant design features incorporated under the approved plans (5). Modifications of the landfill design in future phases could be considered to improve leachate collection.

Operations of the SP-109 landfill now emphasize removal and return of leachate to the plant (for makeup water) to keep levels less than one foot above the landfill liner as required under State rules. Improved management of sludge within the SP-109 landfill, combined with beneficial use of AVS fly ash/FGD for waste solidification and containment are also positive steps for the facility. In addition, phased development of the landfill, with sequential partial closure, as required under the North Dakota Solid Waste Management Rules, emphasize keeping the active disposal area confined to as small an area as possible and bringing waste elevations up to grade. Capping or covering the waste to reduce surface water infiltration, promoting surface water runoff and revegetating filled areas is also required (1).

Storm water control at the landfill base relies on maintenance of dikes and plugs in the leachate piping to separate active fill areas from undeveloped phases, preventing clean storm water from mixing with leachate (5), an issue in early operations. Leachate is transferred by pipe or by truck to the plant for ash-quench makeup water. The system could be improved if lines to the plant were protected from freezing and/or other provisions were made to manage and use leachate in the winter. DGC is considering use of a tank in the plant to accumulate leachate water during the winter.

Located further in the mine, in clay-rich spoil material, more isolated from the Antelope Valley aquifer, with leachate control systems, the SP-109 landfill appears to be a sound facility. With approximately 20 years of space remaining, the facility will provide long-term capacity for gasifier waste disposal. DGC's commitment to continued improvements in operation enhances the facilities service.

Ammonium Sulfate Issues

While a detailed description of air quality issues is outside the scope of this paper, the development of the CO₂ marketing/sequestration process along with the development of the ammonia scrubber system helped solve air quality issues for DGC (3) (4); however, some challenges became apparent. Early emission of excess ammonium sulfate particulate was solved with the construction of an electrostatic precipitator (4). Ammonium sulfate product quality issues and product handling problems in the plant area resulted in disposal of excess product into the SP-109 gasifier ash landfill (5). Spillage of ammonium sulfate within the plant posed surface water and groundwater issues in the Antelope Valley and generated additional waste materials. Again, cooperating with NDDH staff during focused visits, DGC worked to solve the handling and product quality issues.

Interaction of ammonium sulfate with the caustic waste materials in the landfill resulted in strong ammonia odors.

DGC employed a process engineer and made modifications to the ammonium sulfate plant, resulting in significantly less waste. The process improvements afforded increased efficiency, improved product quality, increased profits, solved ammonia odors, and afforded greater protection of groundwater in the plant area: a Pollution Prevention (P2) success story! Indeed, in 2003, previously disposed ammonium sulfate was removed from the landfill for reprocessing in the upgraded fertilizer plant (5).

Beneficial Use Considerations Dakota Gasification Company

DGC has not found an economic use for their gasifier ash. DGC does utilize some fly ash/FGD byproduct from the adjacent Antelope Valley Power Station for waste solidification and stabilization purposes as well as for construction of waste containment structures within the current gasifier ash landfill. The beneficial properties of dry fly ash for waste solidification is commonly understood in the waste industry; however, research in 1985 by McCarthy et al. demonstrated the ability of AVS ash/FGD product to fixate arsenic, barium, boron, molybdenum, strontium, and potassium (7).

McCarthy et al. suggested that the waste streams from AVS should be co-mingled with the DGC waste stream in a common landfill to reduce mobility of most of the heavy metals (7). While this has not been fully adopted, increased use of AVS ash/FGD product in recent years has had beneficial effects and helps control leachate levels in the landfill. Given the elevated arsenic in the landfill leachate (34 - 68 mg/l) cited above, along with the need to minimize leachate levels, increased use of the fly ash/FGD product, possibly as a landfill cover would appear to be beneficial.

Gasification Challenges and Lessons Learned

Development of large scale coal gasification in North Dakota has required innovation and adaptation by plant designers, operations, environmental staff, and the regulatory agencies. Essential issues in need of future consideration of this technology should include facility siting, design (avoid use of water for quenching and handling ash if possible), waste and product handling, water balance issues, and training and communication between facility staff and regulators. If water is used for ash management, waste handling and disposal issues are more critical. Use of dry fly ash/FGD as a product to cover and/or solidify waste is suggested.

In contrast to a coal-fired power plant, coal gasification presents unique challenges with a wider variety of waste and more complex operations. Siting of coal gasification plants should try to avoid environmentally sensitive areas such as shallow aquifers and/or river plains. Providing redundant design elements to address spillage and accidents is also appropriate. Through prudent operation and oversight, DGC plant staff have diligently worked to maintain pond liners and minimize and cleanup the effects of spillage; however, such problems do arise even at a well-run facility. Luckily, suitable locations in the adjacent mine were available for primary solid waste facilities.

Critical water balance issues during early plant operation necessitated rapid construction of additional pond capacity to handle waste waters. Due to material handling problems, spillage in the ash handling area and accumulation of ash quench water in the ash storage and disposal areas were significant challenges. It is suggested that future plant design avoid, if possible, utilizing water for quenching and handling gasifier ash.

At this facility and at others, the Department learned that temporary storage of waste material should generally be avoided. It is best to have secure, well-designed disposal facilities constructed prior to the initiation of operation.

If a plant is designed with an ash quench system, careful design and operation of the ash dewatering system is vital to decant highly caustic quench water from ash and sludge. The ash handling area should be developed to contain spillage and afford cleanup. Staff training on safety and health issues regarding caustic materials as well as environmental issues can help avoid problems.

The Department found it most helpful to work closely with facility staff and maintain a routine presence. Thorough training and oversight of sensitive operations, along with timely response helps reduce impacts from spillage of wastes, liquids, and byproducts. In addition, during construction and maintenance, spillage of chemicals and cleaners by contractors or subcontractors on the plant should be avoided and, when it occurs, be promptly cleaned up.

Regular inspections by plant staff and reports (monthly and/or quarterly) to the Department helped ensure communication. In addition, early permits required regular training and meetings between operations and environmental staff.

In North Dakota, landfills located well within the mine, above the base of the spoils, work very well to afford environmental protection. Design of a leachate collection system, essential if wet wastes or higher precipitation is an issue, must carefully evaluate system plumbing and pumps. Systems for returning ash liquids and leachate back to the plant during all operations, including winter months, should be afforded. Leachate storage near the landfill and/or within the plant should be addressed. Having staff with significant experience with solid waste, environmental issues, as well as management of materials and waste water is important. Finally, utilization of fly ash or other sorbent materials is advised to help reduce leachate generation and to solidify waste materials in the landfill.

Coteau Properties – Freedom Mine Beneficial Use of Coal Combustion Material

The Coteau Properties Company (Coteau) utilizes fly ash/FGD product from the Basin Electric AVS power plant to solidify and stabilize soils on their haul roads, bridge abutments, and equipment repair pads. The North Dakota Department of Health works carefully with the mining companies, power companies, and potential users of coal combustion material to ensure potential beneficial uses are carefully considered to maximize the benefits while at the same time ensuring environmental protection. Working closely with the Department of Health's divisions of Waste Management and Water Quality, the North Dakota Public Service Commission's Mine Reclamation Division, and local health officials, Coteau Properties Company has carefully studied soil stabilization measures and has evaluated surface water runoff from ash treated areas.

Surface mining in North Dakota, on the scale of the Coteau Properties' Freedom Mine, is a challenge. Coteau typically mines about 16 million tons of coal per year which is transported in mining trucks with a total loaded weight of about 500 tons (8). Road construction over regraded spoils in challenging climatic and operating conditions is a serious issue. Use of ash for soil stabilization replaces use of gravel which would otherwise have to be hauled into the mine from up to 40 miles away (8). The mine carefully controls dust and manages surface water runoff. The staff indicates their soil stabilization measures saves millions of dollars in road construction each year, reduces maintenance costs, reduces equipment wear and tear, and saves fuel. In addition, use of ash/FGD for soil stabilization reduces the off-site impact from gravel mining and hauling (8).

Coteau Properties' haul roads typically are about 90 feet wide and are comprised of about 3 feet of carefully compacted spoil material mixed with 20 percent fly ash/FGD material. These 3-foot thick roads are constructed in 8-inch lifts (8). Fly ash from the AVS plant is typically brought in with belly dump trailers which is then spread and disced into the soil. Moisture levels are maintained at optimum moisture plus or minus 2 percent and each lift is carefully compacted utilizing a 75,000 pound sheep's-foot compactor to approximately 95 percent of modified proctor. Additional 8-inch lifts are applied until the total road thickness of 3 feet is achieved. Side slopes of the finished road are covered with one foot of clean, clay-rich spoil material and the road surface is covered with 6 inches of gravel for a wear surface (8).

Adapting the standard penetration testing method to its near-surface road construction, Coteau staff finds the fly ash/soil mixture is typically about 255 percent stronger than roads that are untreated (8). Utilizing the California Bearing Ratio (CBR) strength test, Coteau's fly ash/sandy lean clay mixture at a 7-day cure, 4-day soak achieves a CBR strength of 175 percent compared to an untreated (no ash) soil strength of 85 percent. With increasing moisture, the treated soil maintains significant strength compared to untreated compacted sandy lean clay. Unconfined compressive strength for fly ash treated roads at 7 days, 35 days, and 90 days is 264 pounds per square

inch, 326 pounds per square inch, and 402 pounds per square inch, respectively (8).

Use of AVS fly ash/FGD by the Coteau Properties Company has increased from 21,050 tons in 1999 to nearly 58,000 tons in 2002 (8). In addition to saving to the mining company money and reducing impact to the environment from gravel mining and hauling, the savings in landfill space is significant. The performance of fly ash treated roads is outstanding. With 25,000 to 30,000 loads of coal hauled per year in trucks weighing 500 tons on roads constructed in regraded spoils, the effects of such heavy equipment on these roads, combined with wintertime frost heave and spring melt out presents substantial challenges. Utilization of ash/FGD is invaluable to Coteau Properties Company and the company's environmental monitoring demonstrates the suitability of ash/FGD utilization.

Working with the Department and the North Dakota Public Service Commission, Coteau Properties Company conducted appropriate analysis of the mixed material and identified appropriate measures for environmental monitoring. Coteau Properties recent analysis of runoff from treated and untreated roads showed that most samples had extremely low concentrations of constituents and that average runoff for both fly ash/FGD treated and untreated road types meet all State water quality standards for Class 1 streams (9). Selenium concentrations were within the State water quality standards and the average concentrations of appropriate constituents in runoff was below the current maximum contaminant levels (MCLs); however, the average arsenic concentration from treated roads (.0158 mg/l) and untreated roads (.0149 mg/l) exceed the January 2006 revised MCL for arsenic of .010 mg/l (9). Given the closeness of arsenic values in treated and untreated roads, it is difficult to determine if there is a significant difference in the values: such levels of arsenic are fairly common in North Dakota soils and groundwater.

While appreciable lead is not typical in North Dakota ash or coal, Coteau Properties did find that the average lead concentration from treated roads (.017 mg/l) and untreated roads (.031 mg/l) exceed the regulatory action level of .015 mg/l for public drinking water systems (9). A lower lead level in the runoff from treated roads may perhaps be due to the positive influence of lime in the fly ash/FGD material.

Coteau Properties concludes that utilizing AVS fly ash/FGD material provides extremely durable low cost roads. Treated soil strength exceeds untreated soil by a factor of about 2.5. The company also likes the material since it acts as a drying agent, allowing construction in inclement weather and soil conditions. Ash stabilized roads show virtually no frost heave and stay dry and strong during extended periods of inclement weather. Utilization of ash extends the construction season and the company has not found significant environmental issues regarding runoff (9). By working with the North Dakota Department of Health and the North Dakota Public Service Commission, treated soil from haul roads is managed within the mine as spoil material which is reclaimed with a cover of 4 feet of clean spoil and suitable plant growth material (10).

Further information on North Dakota's approach to beneficial use of coal combustion material can be found in the publication: "Guideline 11, Ash Utilization for Soil Stabilization, Filler Materials and Other Engineering Uses" which is online at: <http://www.health.state.nd.us/ndhd/enviro/wm/pubs.htm#swpubs>. In addition, a description of North Dakota's approach to coal combustion waste regulation and beneficial use is described in the 2002 OSM Technical Interactive Forum proceedings in the North Dakota paper: "Coal Combustion Waste, North Dakota Regulatory Perspective" (1).

Conclusions

Regulatory development in North Dakota, following the model of the Federal Subtitle D rules for municipal solid waste provides a good framework for managing coal combustion waste materials. Coal users in North Dakota have worked with the North Dakota Department of Health and the North Dakota Public Service Commission to resolve many issues. Disposal in mine settings can provide optimal environmental protection. If problems do arise, the North Dakota Solid Waste Rules provide a framework to address issues such as corrective action. Adaptation of landfill design is necessary based on waste characteristics, liquid content, site characteristics, etc. The State's experience with the coal gasification facility could be useful for future consideration of this technology.

Based on the information to date, coal combustion byproduct usage in haul road construction and other soil stabilization purposes along with use for waste treatment/solidification appears: (1) sound, (2) saves resources, (3) reduces equipment wear and tear, (4) reduces off-site impacts, and (5) affords some improvements in environmental

performance. While there certainly are still challenges and improvements to be made, certain coal combustion byproducts show potential for greater development both within and outside mining areas.

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Session 2

STATE PROGRAM REGULATION OF CCBs AT MINE SITES: CASE STUDIES

Session Chairperson:

Dan Wheeler

Department of Commerce and Economic Opportunity

Office of Coal Development

Springfield, Illinois

Permitting Issues Associated with Wet FGD Placement at the Mettiki Mine in Maryland

James Ashby, Mettiki Coal, LLC, Oakland, Maryland

The Use of Dredged Material and Alkaline Activated Coal Ash in Abandoned Mine Reclamation: Final Report of the Bark Camp Demonstration Project

Andrew Voros, NY/NJ Clean Ocean and Shore Trust (OCEAN), New York, New York

Alternative Methods of Ash Placement at a Non Coal Mine Site in Oklahoma

Bret Sholar, Oklahoma Department of Mines, Oklahoma City, Oklahoma

Historical Perspective of Coal Combustion Waste Management at Basin Electric Antelope Valley Station, Dakota Gasification Company, and Coteau Properties Mine, Beulah, North Dakota

Steven Tillotson, North Dakota Department of Health, Bismark, North Dakota

CHISMAN CREEK SUPERFUND SITE: A RETROSPECTIVE REVIEW

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Abstract

The Chisman Creek Superfund Site is the nation's only coal combustion by-product monofill site on the National Priorities List (NPL). The site contains a unique combustion by-product, a mixture of fly ash and bottom ash derived from burning bituminous coal and delayed petroleum coke. Chisman Creek is located in York County, Virginia, near the historic village of Yorktown. By-products from the nearby Yorktown Power Station were hauled by truck by a local contractor and placed into three contractor-owned sand and gravel surface mines, know locally as borrow pits, from 1957-1974. The borrow pits were unlined, located in highly permeable sands, and poorly maintained. During its operating days, local citizens complained about dust, erosion, and runoff into Chisman Creek, a tidal tributary of the Chesapeake Bay. In 1980, contamination of a nearby residential well with trace metals was linked to the site and led to the initial investigation by local and state agencies. Following final designation as a Superfund site in 1985, Virginia Electric and Power Company, a unit of Dominion Resources, Inc. (Dominion), owner of the power station that generated the by-products, agreed to remediate the site under oversight of the U. S. Environmental Protection Agency. Dominion remediated the site in the late 1980s and, working jointly with the County of York, converted it into a public park. This paper recounts the history of the site, describes the unique nature of by-product, examines the site geology and contamination, and defines the significance of the site with regard to the reuse of coal combustion by-products.

Site Location and Description

The Chisman Creek Superfund Site is located in eastern Virginia approximately 4 miles southeast of the historic village of Yorktown. The site consists of four former ash disposal areas, three man-made ponds, freshwater intermittent stream that drains the site, and the upper tidal portion of the Chisman Creek estuary. Chisman Creek is a tributary of the Chesapeake Bay.

The ash disposal areas are former sand/aggregate quarries or borrow pits that were filled with coal combustion byproducts from Yorktown Power Station.¹ The disposal areas are known as Areas A, B, C, and D. Ash in Area D was later removed and relocated in Area C. The three man-made ponds are borrow pits that never received ash and are known as Ponds A, B, and C.

Topographically, the site is a broad, terraced plain incised by the freshwater tributary. The terraced plain is an ancient geologic shoreline of the Chesapeake Bay. Surface elevations range from approximately 45 feet above mean sea level immediately to the west of the site to sea level in the east. Ash placement activities in the past have elevated the surface of the disposal pits as much as 20 feet above the surrounding terrain.

The local surface soil is known as the Tabb Formation and is composed of medium sand with some gravel and fine silty clayey sand. The Tabb Formation at the site ranges in thickness from 0 to 20 feet and is of moderate to high permeability. The Yorktown Formation underlies the Tabb Formation and is composed of silty sand with calcareous fossil debris. The Yorktown Formation is of comparatively lower permeability. The interface separating the strata is a significant geochemical and physical groundwater boundary. Local groundwater flow in the Tabb Formation is generally toward the east and north, toward the tributary stream and estuarine creek. Shallow groundwater flow has

¹ Yorktown Power Station is a fossil fuel steam electric generating station located approximately one mile east of the village of Yorktown and approximated 2 miles north of the site. Virginia Electric and Power Company, a unit of Dominion Resources, Inc. (Dominion), owned and operated Yorktown Power Station and contracted with Brandt for ash disposal services.

been interrupted by the presence of the borrow pits and ponds.

Prior to remediation, the site resembled vacant lots and fields that were in transition from an old field and shrub habitat to a mixed pine and forest hardwood habitat. Figure 3 shows the site in the early 1980s. Vegetation was growing in a thin soil cover or directly in ash. In several areas, ash was exposed and eroded by wind and precipitation. Access to the site was unrestricted and local citizens used the site for riding motorbikes, hiking, gardening, and dumping of household and construction debris. The ponds supported bass and sunfish populations and were popular fishing locations. Pond C was used by York County officials as a source of minnows used for stocking other water bodies to aid in mosquito control.

Site History

From 1957 to 1974, R. L. Brandt and Sons, Inc. (Brandt) trucked ash from Yorktown Power Station and deposited the ash into the borrow pits. Brandt was a small “Mom and Pop” business under contract to Dominion for ash disposal services. The disposal pits were owned or otherwise controlled by Brandt. Ash was transported on public roads, often in open or partially covered trucks, and end-dumped and compacted into the pits. Numerous complaints by local officials and citizens during this period about Brandt’s transportation and disposal practices, especially dust and erosion problems, led to legal action to correct certain practices. Some improvement eventually occurred but overall care and custody of the ash in the site during its seventeen-year operating life was poor.

Approximately 500,000 tons of ash was placed at the site. Ash disposal ended in 1974 following conversion of Yorktown Power Station to oil and the termination of the contract between Brandt and Dominion. Brandt subsequently closed the site and placed a soil cap on the ash. The closure of the site was in response to enforcement action by the Virginia Water Control Board, now Virginia Department of Environmental Quality (VDEQ).

In 1980, water in a domestic well located in the Tabb Formation near Pit C developed a yellowish green color. The well was located approximately 100 feet from Pit C and ironically was at the residence of a former Brandt employee. Analysis of the well water revealed higher than expected levels of certain trace metals, such as nickel, vanadium, arsenic, and selenium. In response, local officials provided municipal water service to this residence and several nearby homes and notified state officials.

The Virginia Department of Health and VDEQ investigated local groundwater and searched for sources of the contamination in late 1980 and continuing into 1981. With support from Dominion, State officials tested domestic well water, groundwater from newly installed monitoring wells, and samples of ash. Of the 33 domestic wells tested, only two contained elevated levels of trace metals, principally vanadium. Elevated levels of vanadium were found in the monitoring wells and the ash samples. The VDEQ concluded that the ash in the disposal pits had caused shallow groundwater contamination within and adjacent to the site.²

In December 1980, the Comprehensive Environmental Response Compensation and Liability Act of 1980 (CERCLA) was enacted. A few months after passage, EPA sought candidate sites from each State for the initial National Priorities List (NPL) which is commonly known as the “Superfund List.” The VDEQ identified Chisman Creek as a possible site based on the 1980-81 site investigation and EPA listed the site as #114 on the initial list. Ironically, the VDEQ did not consider Chisman Creek to be one of the worst 114 sites in the nation or even one of the worst in the State and sought to nominate other sites considered to be more serious. However, EPA rejected these sites.

In 1983, EPA finalized the inclusion of the site on the NPL. Dominion objected to the NPL listing but was unsuccessful in convincing EPA to remove the site. Facing a statutory deadline, Dominion joined with other petitioners and filed in federal court to overturn the listing. The case was consolidated with other petitioners as

² The Virginia Institute of Marine Sciences (VIMS) undertook a second, more comprehensive study of the site in 1981-1983. This study was with partially funded by Dominion. VIMS confirmed the previous findings of the 1980-1981 study and further concluded that ash eroding from the site was not adversely impacting shellfish in Chisman Creek.

Eagle-Picher, Inc. et al. v U.S. Environmental Protection Agency.³ In 1985, the court ruled in favor of EPA and the listing was finalized.

EPA began the Remedial Investigation (RI) and Feasibility Study (FS) in January 1984. The RI/FS process experienced many of the now familiar problems that characterized the early years of the CERCLA program, such as inadequate planning, inexperienced project personnel, laboratory delays, and personnel turnover. The RI/FS was originally slated for 18 months but ultimately took 32 months.

After beginning the RI/FS, EPA divided the remediation investigation and hence, the site into two spheres of study or operable units. Operable Unit 1 consisted of the ash pits and surrounding groundwater. Operable Unit 2 consisted of three ponds, the freshwater creek, and the Chisman Creek estuary.

The Collective RI identified the presence of trace metal contaminants in several media at and near the site. In general, the extent of contamination is primarily limited to groundwater in and around Pit C, the oldest and deepest disposal area. The RI findings are summarized below:

- Contaminant pathways include transport via erosion, surface water, and groundwater with groundwater transport judged to be the more significant.
- No organic contaminants attributable to site ash were detected.
- All ash samples passed the EP and TCLP toxicity tests and therefore are not hazardous wastes.
- Nickel and vanadium were the most ubiquitous trace metal contaminants with elevated concentrations of arsenic, beryllium, chromium, copper, molybdenum, and selenium found in some parts of the study area.
- Trace metal contamination was limited to shallow groundwater in the Tabb Formation.
- Filtered groundwater samples outside the ash pits contained no contaminants at levels exceeding EPA guidelines except for nickel in two shallow monitoring wells adjacent to Pit C.
- None of the samples from residential wells contained contaminants above primary and secondary MCLs.
- Surface water was not toxic to fish but was mildly toxic to a freshwater invertebrate test organism. Sediment was not toxic to shellfish but was mildly toxic to a marine invertebrate test organism.
- Histological tests of fish and oysters revealed no impact from the site.
- The consumption of fish and oysters from the site did not present a human health hazard and the ingestion of water and sediments were within EPA human health risk guidelines.

The Feasibility Studies identified and evaluated numerous potential remedies for the site. Capital and short term implementation costs for Operable Unit 1 ranged from \$180,000 (No Action) to \$49,000,000 (Ash Excavation and Offsite Disposal). For Operable Unit 2 the costs ranged from \$170,000 (No Action) to \$3,000,000 (Dredge Creek Sediments).

EPA issued the Record of Decision for Operable Unit 1 in September 1986, which detailed the remedy selection. The cost of remedial action for Operable Unit 1 was estimated to be \$14,100,000. EPA issued the Record of Decision for Operable Unit 2 in March 1988 and estimated the cost of remedial action to be \$300,000.

The collective remedy selected by EPA for both operable units is summarized below:

- Provide municipal water service to 55 residences near the site.
- Establish land use controls to prohibit excavation, control site development, and restrict groundwater use in and adjacent to the site.
- Install graded soil cap and stormwater controls on Pits A and B to prevent direct human contact with the ash and prevent erosion.
- Install low permeability cap and stormwater controls on Pit C to prevent direct human contact with the ash, minimize erosion, and reduce infiltration.

³ Dominion argued that because coal ash, including ash from the site, which is more than 50 percent coal ash, was exempted by EPA from the Resource Conservation and Recovery Act (RCRA) and CERCLA incorporates RCRA wastes for regulation under CERCLA, EPA improperly listed the Chisman Creek Site. The court disagreed and ruled that EPA could list sites in the NPL that contained wastes, exempted or otherwise, that have RCRA regulated hazardous substances as constituents. Because virtually all material (e.g., construction debris, mining waste, etc) contains constituents that are hazardous substances under RCRA, the effect of the court's ruling was a broadening of EPA's response authority under CERCLA.

- Install subsurface perimeter drain around three sides of Pit C to collect contaminated groundwater and to lower the groundwater table within Pit C to below the level of the ash.⁴
- Install and operate a treatment facility to treat groundwater collected from Pit C until NPDES limits are attained.
- Implement post-closure monitoring of groundwater, surface water, and sediments.

From Fly Ash to Fly Balls

Throughout the RI/FS process and continuing into the remedial design phase, Dominion remained in close contact with officials and citizens of York County. Early EPA plans included land use controls and the installation of barbed wire fences around the site. Many citizens objected to the presence of this type of fence in their neighborhood and discussions ensued regarding potential future uses of the site. Dominion initially evaluated the site as a potential company office site, future golf course, commerce park, and transportation facility. In discussions with York County officials, the use of the site as a public recreation emerged as a prime need. Dominion proposed a conceptual plan to York County, which was reviewed and endorsed by a local citizen committee. The York County Board of Supervisors approved the plan⁵ and in December 1986, EPA certified the conceptual plan, paving the way for final design and remedial construction.

Dominion agreed to perform the remedial action and began design, land acquisition, and construction in late 1986. Major site construction was completed in December 1988 with minor work continuing into 1989. Operation of the treatment plant began in August 1989. Dominion completed the remedial action for approximately \$10,000,000, including payment of approximately \$750,000 in EPA “oversight” costs.

The treatment plant operated until 1995. Water quality has improved to the point that it is discharged to the local sewerage system. As yet, it is not suitable for direct discharge to Chisman Creek due to the continuing presence of nickel and vanadium. More than 70,000,000 gallons of groundwater has been drained from Pit C and treated either in the onsite treatment plant or the local sewerage system.

EPA has classified Chisman Creek Superfund Site as a *Construction Completed – Long Term Monitoring* site. Thousands of samples have been collected, analyzed, and reported. Surface water and sediment quality has improved and is no longer monitored. Groundwater quality has improved and sampling frequency is reduced to reflect this improvement. The horizontal drains have stabilized water levels in Pit C.

No major site maintenance problems have developed. York County repairs minor erosion and stormwater runoff problems as part of their normal maintenance program. Two areas of subsidence related to animal burrows and heavy rains were jointly repaired in 2003. York County continues to improve the park infrastructure and the recreational complex receives heavy use by local recreational teams and fans.

Site Geology and Nature of Contamination

Chisman Creek is a series of sand and gravel surfaces mines or borrow pits. Geologically, the pits are located on an ancient sand shoreline of the Chesapeake Bay. The pits are less than 45 feet above mean sea level. Groundwater elevations are shallow, that is, close to the surface of the land. When the sand was mined, surrounding groundwater filled the borrow pit excavations. When the borrow pit excavations were filled with ash, the standing water in the pit was displaced.

⁴ Dominion contracted GAI Consultants, Inc. of Monroeville, PA to perform the remedial design. GAI modeled the performance of the perimeter drain and found that it would not dewater the interior of Pit C due to upward groundwater recharge from the underlying Yorktown Formation. GAI evaluated several alternative drain designs and eventually developed an alternative drain that exceeded the effectiveness of the original drain. This alternative was a horizontal drain system consisting of 23 drains drilled laterally across the base of Pit C and common header. This alternative design reduced the remedy cost by approximately \$1,000,000. EPA approved the remedy change.

⁵ Dominion leases the land to York County in an amount that equals the property taxes. The 60 year lease specifies that Dominion retains all environmental liability and York County maintains the site consistent with its normal park maintenance program. Dominion constructed the ground features (playing fields, utilities, foundations, etc.) of the park as part of the site grading work and York County installed the park appurtenances (fences, restrooms, sidewalks, etc.). York County had an active role in the design of the park complex. In May 1990 a ceremony featuring officials from Dominion, York County, and EPA opened Chisman Creek Park and Wolfrap Park to the public.

The borrow pits were unlined. The surrounding sand soils were highly permeable. The ash placed in the borrow pits was generally compacted but not in a defined manner or according to any technical specification. It is generally thought that the placed ash was less permeable than the surrounding sand soils. Nevertheless, groundwater contacting the placed ash, either by infiltration or lateral or vertical flows, caused the formation of ash leachate.

Leachate quality in and adjacent to Pits A and B is better than Pit C. These pits are shallow and contaminant transport is due to infiltration and lateral groundwater flow. Lateral groundwater flow is slow and groundwater in the highly permeable sand soils outside of the pits travels around the compacted, less permeable ash in the pits. In contrast, Pit C exhibits the highest levels of trace metal contamination relative to Pits A and B due to two factors. First, it is the largest and deepest pit, with the bottom of the pit extending downward into the Yorktown Formation. The depth of the pit removes a natural aquatard and allows upward flow of groundwater from the Yorktown Formation. Pit C experiences infiltration, lateral groundwater flow and vertical groundwater flow, with the latter component responsible for the contaminant signature of Pit C leachate.

Contamination at Chisman Creek is primarily a groundwater contamination issue with a secondary issue related to the impact of eroded ash into surface waters. The latter concern has been addressed with capping the site and the elimination of ash erosion. Groundwater contamination centers on trace metals, particularly nickel and vanadium. Contaminant levels have lessened over the years, due in part to the lowering of groundwater at Pit C below a large portion of the site.

The unmanaged placement of wastes on vacant land and in unlined borrow pits was a common waste management activity in the past. Even more common was the placement of wastes without regulatory requirements or technical standards. The science of waste management and the environmental impacts of certain practices were poorly known.

Chisman Creek Ash

The Chisman Creek site is distinctive because it contains combustion byproducts or ash—fly ash and bottom ash—from a steam electric generation station that burned a unique fuel. The fuel was a variable blend of central Southwest Virginia/West Virginia bituminous coal and delayed petroleum coke derived from Venezuelan crude oil. The coke was supplied by a refinery located adjacent to Yorktown Power Station. From 1957 to 1974 the cumulative fuel mix ratio was 51% coal and 49% coke.

The primary trace metal contaminants of concern at the site are nickel and vanadium. Nickel and vanadium are characteristically present in relatively high levels in petroleum coke and in relatively low levels in bituminous coal. Consequently, the resulting byproducts of this fuel were enriched in corresponding proportions. The following table provides a comparison of nickel and vanadium in unburned coal and coke, coal and coke ash and Chisman Creek fly ash.

CONCENTRATIONS OF NICKEL AND VANADIUM IN COAL, COKE AND ASH ASSOCIATED WITH THE CHISMAN CREEK SUPERFUND SITE		
Type of Material	Nickel in PPM (Weight)	Vanadium in PPM (Weight)
Coal ¹	15	35
Petroleum Coke ²	250	2,487
Coal Ash ³	100	180
Coke Ash ³	873	24,745
Chisman Creek Fly Ash ⁴	181	1,830

¹ Average trace metal content of 53 Eastern U.S. coals. From Table 201 in Coal Ash Disposal Manual Second Edition. Electric Power Research Institute, Final Report, October 1981.

² Analysis of petroleum coke samples from Yorktown Power Station. From Dominion Records, 1957.

³ Analysis of coal and coke ash samples from Yorktown Power Station. From Dominion Records, 1957.

⁴ Average of three Chisman Creek ash samples. From Remedial Investigation Report, Vol. 1, Chisman Creek Superfund Site. EPA WA 83.3L37D. W63237.00, 19857.

The leachate produced at the Chisman Creek site is markedly different from the leachate at a coal ash placement facility. Coal ash leachate characteristically contains low levels of trace metals while the coal/coke ash leachate at the Chisman Creek site is enriched with higher levels of nickel and vanadium. The following table compares the actual leachate quality from the Pit C horizontal drain system with the leachate quality from the active ash disposal facility at Dominion's Yorktown Power Station.

COMPARISON OF RAW LEACHATE FROM CHISMAN CREEK SUPERFUND SITE WITH RAW LEACHATE FROM YORKTOWN POWER STATION COAL ASH		
Parameter	Coal Ash Leachate ¹	Coal/Coke Ash Leachate ²
Arsenic	<5 ug/l	<15 ug/l
Copper	<3 ug/l	52 ug/l
Lead	<10 ug/l	<36 ug/l
Nickel	<40 ug/l	243 ug/l
Selenium	<5 ug/l	<15 ug/l
Vanadium	<200 ug/l	2,318 ug/l
Zinc	<15 ug/l	<1,186 ug/l

¹ Average of analytical results of monthly Yorktown Power Station ash disposal facility leachate samples collected from January, 1989 - December 1989.

² Average of analytical results of Chisman Creek treatment plant influent samples collected from November, 1989 -August, 1990.

The comparison data clearly demonstrates that the source of the elevated levels of the two principal contaminants of concern at the Chisman Creek Superfund Site, nickel and vanadium, is the petroleum coke ash disposed at the site. This distinction is important to the coal ash reutilization industry.

Significance of the Chisman Creek Superfund Site

The Chisman Creek Superfund Site is important historically because it happened. It is important because it was one of the worst sites in the nation, i.e., a site on the National Priorities List. It is important because it is also the only coal combustion byproduct monofill site in the nation. It is important because of coal ash and more importantly, because of petroleum coke ash. It is important because the site is a one-of-a-kind occurrence, arising out of a “perfect storm” of unfortunate timing and events, and because the site underwent a very remarkable renaissance.

Chisman Creek is well-known for its historical importance in the Superfund program. It was an early success story for the regulatory community, particularly in the early years of the program when success stories were scarce. Chisman Creek may be the first reuse of a large Superfund site in the nation and is often applauded as an example of what can be accomplished in restoring contaminated land.

Dominion views Chisman Creek as an excellent example of its corporate stewardship, ingenuity, and responsiveness to the local community. Early in the process, Dominion accepted its responsibility and committed to restore the site to a useful purpose. Dominion’s vision and leadership converted a community liability to a regional asset and in doing so, realized the value of a true public/private partnership.

Similarly, Chisman Creek is important to the citizens of the region. While it was once an unwanted and unkempt tract of land, the community liability has vanished. It has been replaced by a recreational complex that adds to property values and the quality of life. Governmental officials from York County exhibited leadership and farsightedness by partnering with Dominion and in exchange, received valuable additions to its park system at a very low cost. Today, the surrounding land is enjoying a building boom.

The site is important to the recycling and reutilization industry. Chisman Creek was one of the first sites on the NPL and it is true national success story. The practices that led to the site contamination, i.e., disposing of ash in an unlined sand pit with no management controls or engineering specifications, are inconsistent with ash management practices of today. Despite these poor practices, however, the extent of contamination at the Chisman Creek site is very limited and readily controlled.

Chisman Creek is important to the coal ash industry because the contamination responsible for its listing and notoriety is attributable to the high levels of contaminants in petroleum coke ash, not coal ash. Chisman Creek should not be held up as an example of the environmental risks of coal ash utilization or used to oppose sound coal ash reutilization initiatives. Rather, Chisman Creek should be trumpeted as an excellent example of community involvement, governmental leadership, public/private partnering, and corporate vision and stewardship.

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GROUND WATER IMPACTS FROM COAL COMBUSTION ASH DISPOSAL SITES IN WISCONSIN

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Abstract

The State of Wisconsin's Department of Natural Resources (WDNR) regulates many large and mid-sized coal-powered electric generating plants and both active and inactive ash disposal sites. These sites range from engineered landfills to older, unlined disposal sites located in abandoned bedrock quarries and sand & gravel pits. WDNR's evaluation of the groundwater monitoring data from licensed disposal sites extending back to the late 1970's shows that the impacts to groundwater quality vary considerably. The primary factors influencing the level of impacts include the disposal method used, the hydrogeologic conditions of the site, the geochemical properties of the coal from which the ash was derived, and the design of the disposal facility. Boron, sulfate, arsenic and selenium commonly exceed Wisconsin's groundwater quality standards adjacent to unlined ash landfills. Some sites also have elevated levels of lead, manganese, mercury, and molybdenum. To illustrate the nature of groundwater contamination from coal combustion byproducts (CCB) disposal sites, two landfills cited by the U.S. EPA, the Cedar-Sauk Landfill and the Hwy. 59 Ash Landfill, were examined in detail. The study shows that CCB disposal under both saturated and partially saturated conditions promoted rapid leaching of constituents that affect water quality.

Introduction

The State of Wisconsin's Department of Natural Resources (WDNR) regulates both coal-powered electric generating plants and the corresponding ash disposal sites. These sites range from active, engineered landfills to closed, unlined disposal sites located in abandoned bedrock quarries, wetlands, and sand & gravel pits. The disposal of coal combustion by-products (CCB) in unlined sites was only an accepted practice in Wisconsin prior to the promulgation of administrative codes and the establishment of procedures for siting and constructing waste disposal facilities. Chapter NR180, Wisconsin Administrative Code for Solid Waste Management was promulgated in 1988. Solid waste disposal, environmental monitoring, and beneficial reuse are currently regulated under administrative code chapters referred to as the NR500 series (NR500-NR590).

Groundwater quality standards for regulating solid and hazardous waste facilities are specified under Chapter NR140, Wis. Adm. Code. Groundwater quality is measured against the enforcement standard (ES) and the preventive action limit (PAL) for substances that are of public health concern. The enforcement standards are roughly equivalent to the Federal maximum contaminant levels (MCLs). The preventive action limit is generally 10% of the ES for all substances that have carcinogenic, mutagenic, teratogenic properties or interactive effects and 20% of the ES for other compounds. Enforcement standards and preventive action limits for boron became effective on January 1, 1999. Samples from potable drinking water wells generally are not filtered, and represent total values for compounds and are regulated under Safe Drinking Water Standards in Chapter 809. The regulatory standards for some groundwater contaminants have changed over time while others have been only recently codified.

The primary purpose of this paper is to analyze groundwater quality data from a wide variety of Wisconsin CCB disposal sites and to compare and contrast these results with water quality data from the two detailed study sites: the WE Energies Highway 59 and Cedar Sauk landfills. The data was also examined to determine possible trends or similarities between different sites and among various disposal site categories.

Wisconsin Groundwater Quality Data

To evaluate the impact of CCB disposal sites on groundwater quality, it became necessary to identify both active and closed CCB monofills in Wisconsin that have active groundwater monitoring. All groundwater quality data for solid waste facilities that is reported to the WDNR is recorded in the Groundwater and Environmental Monitoring System (GEMS) database. The GEMS database is one of the largest in the nation, containing over 10,994,294 analytical results from 18,467 active and 4,259 inactive monitoring points.

From the 22 identified CCB monofill landfills in GEMS, 12 sites (6 active and 6 closed) were chosen based on size, known groundwater impacts, and the completeness of their data sets. A Monitoring Data Summary by Parameter report was reviewed for each of these 12 sites to determine the most impacted monitoring wells at each site. Four compounds were identified as the most commonly detected at levels that exceeded the State of Wisconsin's groundwater protection standards. These compounds include boron, sulfate, arsenic, and selenium. Elevated levels of manganese were detected at about half the sites and there were sporadic exceedances of lead, chromium, and mercury at several sites. Typically, 3 or 4 down gradient monitoring wells would account for the majority of the exceedances at each site. From there, a Summary Exceedance Report detailing all parameters and results was generated for the most impacted well at each site. The report listed the lowest and highest reported sample results and calculated the mean of all samples collected. A summary of these findings is listed in Table 2.

It must be acknowledged that this analysis is intended as a broad attempt to categorize groundwater quality data that is strongly influenced by a large number of variables, most of which are not easily quantified and not addressed in this study. The type and concentrations of contaminants available for release into a groundwater system are directly related to the coal source, the combustion method employed by the generating plant, and environmental conditions of the disposal site. This information is often difficult to obtain or unavailable, especially for older, long closed sites.

In addition, the groundwater monitoring data presented are the mean value of all available data for the most impacted well at the facility. Some of the wells represented are no longer in service. The location of the well can strongly influence the results and some of the monitoring points were actually screened through or just below the waste mass. Especially with regards to the older, closed sites, the mean can be skewed higher by older data that recorded values that have since declined, in some cases dramatically. Even given these constraints, some interesting results can be discerned from the groundwater quality data.

Measurable quantities of boron were recorded in at least one down gradient monitoring well at all the sites reviewed, including active sites that were constructed with clay liners. However, the boron concentrations tended to be rather low at active lined sites, with two sites recording mean values at concentrations below the Wisconsin groundwater standard (0.960 mg/l) for boron. That is in stark contrast to several closed sites that have mean concentrations of boron over 10 times the mean concentration of even the highest value at an active site.

The highest mean boron concentration among all sites was recorded at the Cedar Sauk Landfill (mean 101 mg/l). This value is well above the next highest mean concentrations (69 mg/l and 30.7 mg/l, respectively) for either active or closed sites. The GEMS database documents that the highest recorded boron concentration in groundwater monitoring wells (as high as 186 mg/l) were recorded at wells W-3 and W-3A at the Cedar Sauk Landfill. These elevated boron concentrations were most likely influenced by the boron concentrations in the original coal source and the transmissivity of the disposal environment. Data from EPRI database (January 2004) on the boron content of various coals from different regions of the country indicates that the range of boron content can vary considerably.

Table 1

Boron Concentrations in Coal, by USGS Region							
Summary Statistics (mg/kg dry weight)							
Coal Type	Median	Average	Std. Dev.	Maximum Detected	Minimum Detected	Number of Data Points	Number of Nondetect Values
Eastern Bituminous	25	41	43	201	0.26	60	4
Eastern Interior (Illinois Basin) Bituminous	100	113	57	214	11	47	1
Northern Great Plains Lignite	0.5	53	68	150	126	10	6
Northern Great Plains Subbituminous	29	62	99	353	22	22	3
Rocky Mountain Subbituminous	43	56	22	80	39	5	0

Source: EPRI PISCES Database (January, 2004)

Notes: The concentrations presented in the tables were calculated using one half of the detection limits for all nondetected values. Nondetected values (represented by one half the detection limit) greater than two times the highest detected concentration were removed from the data set before statistics were calculated.

NR = Not Reported.

All data that are designated as “poor” or “unacceptable” quality in the PISCES Database were excluded from all calculations. All “unknown” quality data were included in the calculations.

Chlorine and fluorine concentrations measured in the coal and fuel oil analyses are normally reported as chloride and fluoride. However, because the forms of these elements present in the fuels are not known, the elemental designations of fluorine and chlorine have been used in reporting the concentrations in the tables.

Coal analyses are for “as-fired” or “as-received” coals and are presented on a dry weight basis.

As with boron, sulfate was detected in measurable quantities at down gradient wells at all the sites studied. Unlike boron, however, elevated sulfate levels were common among most sites including the active, engineered landfills. The mean sulfate value was above the Wisconsin enforcement standard (ES) of 250 mg/l at 10 of the 12 sites studied. Only one site, the NSP-Woodfield Landfill, had a down gradient well that never recorded a value above the ES. Two active landfills had mean values at (or above) the mean values for most of the closed sites.

Most active sites had mean arsenic values just below the Wisconsin enforcement standard of 10 ug/l. The one active site that had a mean arsenic value above 10 ug/l was the Highway 32 Landfill (30 ug/l). However, the mean was calculated from only two data points, so they may not be representative of the actual arsenic concentrations near the landfill. The arsenic levels at other monitoring wells down gradient of the Highway 32 Landfill were significantly lower.

The results that most stand out are the extremely elevated mean arsenic values at three of the closed sites, Edgewater 1-4 (364 ug/l), Nelson Dewey (197 ug/l), and Alliant-Columbia (40 ug/l). The one thing that all these sites have in common is that CCB waste was sluiced to the site before they closed. None of the other parameters studied has such a close correlation with the waste disposal method. In addition, at least one study site, the Highway 59 Landfill, had very low values of arsenic (mean of 3.04 ug/l) in spite of having a documented significant release into a transmissive subsurface environment. All this suggests that the elevated arsenic levels are more related to the coal source or combustion process rather than hydrogeologic conditions.

Selenium rarely exceeds concentrations of 1 ug/l in natural surface or groundwater (Hem, 1992), so it is useful in determining potential releases from CCB disposal sites. Low levels of selenium were detected at all the study sites, with the lowest mean concentrations occurring in the active sites. However, only two sites, the Cedar Sauk Landfill

(77 ug/l) and the Nelson Dewey Landfill (78.8 ug/l), had mean selenium values that exceeded the Wisconsin enforcement standard of 50 ug/l. While they had similar mean values, the Cedar Sauk Landfill had some of the highest selenium values (up to 730 ug/l) of any landfill in the study. Other than their elevated selenium levels, the two sites have little in common. This suggests that the coal source may be the dominant factor in determining whether or not excessive selenium contamination will be a problem at any given site.

Site Specific Studies

To further illustrate the nature of groundwater contamination from CCB disposal sites, two landfills, the Cedar Sauk Ash Landfill and Highway 59 Ash Landfill, were examined in detail. WE Energies, formerly the Wisconsin Electric Power Company (WEPCO), operated both landfills. Both sites were located in old nonmetallic mine sites, neither was constructed with a liner, and both were closed by 1980. These sites were chosen based on their identification by the US Environmental Protection Agency (US EPA) as examples of sites where there was proven or potential damage involving the placement of coal combustion wastes in sand and gravel pits or mines (US EPA, 2000). A third Wisconsin site not included in this study, the Lemberger Landfill in the Town of Franklin, was also identified by the US EPA. The Lemberger Landfill accepted municipal, industrial waste, and power plant ash during its life. Due to the nature of the disposal methods, it was determined that groundwater impacts from contaminants in the municipal and industrial waste could not be separated from potential impacts from coal ash. Given these uncertainties, this site was determined to be unsuitable for any study attempting to quantify the impacts of CCB disposal on groundwater and was therefore not included in this report.

Cedar Sauk Ash Landfill

Background

The Cedar Sauk Ash Landfill (WDNR License # 603) located in the Town of Cedarburg, Ozaukee County, was operated as a disposal site from 1969 through 1979. The fly and bottom ash was generated by the Port Washington Power Plant (WEPCO, 1985). During this time, approximately 650,000 cubic yards of ash was disposed of on 25 acres of a 42-acre site. Prior to its development as a landfill, the parcel was used as a sand and gravel pit. The landfill was sited before Wisconsin's solid waste disposal laws were enacted so the site was not constructed with either a liner or a leachate collection system.

In response to approval conditions from the WDNR, WEPCO installed and began monitoring a series of monitoring wells in late 1978. The initial results indicated elevated levels of boron, sulfate and selenium in wells down gradient of the landfill. Follow-up inspections of the closed landfill in 1980 and 1981 revealed vegetative stress characteristic of boron toxicity in the wetland vegetation down gradient of the landfill. Analysis of plant tissue collected from the visibly impacted plants showed boron concentrations from between 300 and 1,600 mg/l. Healthy plant tissue sampled in the area had boron levels between 7 and 61 mg/l (WEPCO, 1982). WEPCO concluded that precipitation percolating through the waste mass and a water table near the base of the landfill were likely the cause of these impacts and they proposed some remedial actions.

In 1983, WEPCO proposed a groundwater mitigation plan to the WDNR consisting of 3 groundwater extraction wells to be placed down gradient of the fill area. The water was to be pumped into Mole's Creek after flowing over a rock-filled trench. The WDNR approved the action and the system began operating on September 7, 1984.

During the first 5 years of its operation, the extraction well system seemed to operating well and the contaminant levels in the down gradient monitoring wells dropped. However, by 1990 the levels of all contaminants began to rise again in spite of the extraction system (WEPCO, 1995). Based on groundwater modeling of the system, WEPCO concluded that the capture zone of the three extraction wells was inadequate to capture all groundwater contamination migrating from the landfill. To address this problem, WEPCO installed an engineered cap over the landfill in the summer of 1997 consisting of recompacted clay, 30-mil PVC geomembrane and a rooting layer with two feet of general fill and six inches of topsoil. The cap work was completed on October 3, 1997 and the groundwater extraction system was permanently abandoned.

Geology and Hydrology

The Cedar Sauk Landfill is located in an area of hummocky end moraine and glacial outwash deposits from the last of the glacial advances of the Lake Michigan Lobe some 13,000 to 14,000 years before present. The underlying unconsolidated glacial sediments generally consist of poorly to moderately well sorted sand and gravel deposited on and beneath glacial ice by melt water streams near the ice margin (Mickelson & Syverson, 1997). These deposits and underlying tills are assigned to the Oak Creek Formation. Underlying the Wisconsinian-aged glacial deposits is a relatively thick deposit of Silurian dolostone of the Niagaran Series. This massive dolomitic limestone has a highly irregular erosional surface and contains numerous fractures and solution features. Cross-sections based on borings near and beneath the landfill area show that the landfill was sited on a bedrock knob of Niagaran dolostone. The sand and gravel mining operation removed the unconsolidated material over the bedrock allowing waste to be placed directly on the fractured rock. North and east of the bedrock high, towards Mole's Creek, the sand and gravel outwash deposits thicken from absent to 25 feet thick beneath the landfill to over 50 feet thick near the creek.

There are two major aquifers present at the site: the upper sand and gravel outwash deposits and the lower, Niagaran dolostone (WEPCO, 1995). The sand and gravel aquifer is generally unconfined and forms the water table. To the east of the landfill, near Mole's Creek, the sand and gravel unit is underlain by silty clay to clayey silt glacial till unit. Beneath the landfill, this unit is either absent due to mining or directly underlain by bedrock.

Groundwater flow in the Niagaran dolostone is primarily through an extensive network of fractures and solution features (including paleokarst) in the bedrock. This aquifer is considered a regional aquifer and is used by private water supply wells in the town. However, beneath the landfill and elsewhere along the bedrock knob, it is in contact with the upper sand and gravel aquifer and the water table can occur within both units. Groundwater within both the sand and gravel aquifer and the Niagaran bedrock flows from the west to the east and southeast. Mole's Creek is considered a groundwater discharge point for shallow groundwater that flows from the landfill area. Groundwater table elevations within the shallow, unconfined sand and gravel aquifer range from 808 feet above mean sea level (MSL) at the southwest corner of the fill area to 779 along Mole's Creek. There are no strong vertical gradients at any of the nested monitoring wells, although wells west and within the fill area tend to have slight downward gradients and wells to the east tend to have slightly upward vertical gradients.

Groundwater Quality

The Cedar Sauk Landfill has had an extensive groundwater monitoring system in place since 1978. As might be expected, most of the contamination is concentrated in the sand and gravel aquifer that discharges into Mole's Creek. The most common elevated parameters include boron, sulfate, and selenium.

The highest levels of both dissolved boron and sulfate were recorded at monitoring wells W-3 and W-3A. These nested wells were located within the waste mass and both wells were screened in ash. The data from these two wells is of limited usefulness because they were only sampled from August of 1979 until June of 1983, while the site was closed but inadequately capped. During this limited period, boron levels ranged from 11 to 186 mg/l in well W-3A and sulfate ranged from 520 to 1923 mg/l in well W-3. Both these levels are well above the Wisconsin enforcement standard (ES) of 0.96 mg/l for boron and 250 mg/l for sulfate. While consistently elevated, the levels could vary considerably even from one month to the next. There were even some distinct seasonal trends over the four years of sampling. This pattern suggests that mobilization of both sulfate and boron were related to periodic inputs of precipitation infiltrating through the ash (WEPCO, 1995).

A detailed review of all groundwater quality data from the site indicated that monitoring well W-4 was the best candidate to represent groundwater impacts from the landfill. W-4 is an active well that has groundwater quality data stretching from August of 1979 to the present. It is located on the eastern edge of the fill area and is screened in the shallow sand and gravel unit. The data indicates that it has been significantly impacted by disposal practices. In fact, the highest levels of selenium, up to 730 ug/l, recorded at the site were from well W-4. The groundwater quality trends from this well are also very similar to general water quality trends found in other down gradient wells.

The pH values from well W-4 were very consistent and only varied between pH 6.7 to 7.6, with a mean value of pH 7.07, essentially neutral. This is in contrast to the leachate results from wells W-3 and W-3A, where the results were slightly alkaline with pH values between 6.8 and 8.7 with a mean of pH 7.8. These results mirror the results of the larger study presented in this paper, again suggesting that the buffering capacity of even sandy soils at the site were

adequate to neutralize the low-level alkalinity in the leachate.

Boron concentrations were consistently elevated at well W-4 with levels between 6 and 140 mg/l, well above the ES of 0.96 mg/l. What is most remarkable about these concentrations is the way they responded to site activity. The boron levels steadily climbed as the site was being filled up to its final grades, going as high as 112 mg/l. However, when the site was finally closed and the relief well system was activated in 1984, the boron levels dropped dramatically from 52 mg/l to 6.8 mg/l in the space of 3 months. The levels fluctuated, but stayed relatively low until they began creeping back up starting in 1993. The exact cause of this upward trend in spite of the continued operation of the relief well extraction system was evaluated (WEPCO, 1995), but an exact cause was not determined. It was suggested that the well system was no longer effectively capturing most of the contamination coming from the landfill.

The boron concentrations reached their peak in August of 1997, when the relief well system was shut down and the new geomembrane cap was being installed. However, after 1997, the boron levels dropped rapidly and dramatically, from a high of 140 mg/l to a low of 14 mg/l in 2003. This drop was mirrored by the sulfate levels, which decreased from 1070 mg/l in 1997 to 262 mg/l in 2003. Clearly, reducing infiltration through the waste mass by installing a geomembrane cover has effectively reduced contaminant concentrations down gradient of the landfill.

The boron and sulfate at well W-4 reacted in similar ways to the remedial measures at the landfill while the selenium levels reacted a bit differently. The selenium levels in W-4 were generally elevated, with levels as high as 980 ug/l, before the site was closed. The selenium concentrations dropped sharply after the site was closed and the relief well system became operational in 1984. Within 3 months (September, 1984 to December, 1984, the same period mentioned with the boron and sulfate results), the selenium levels dropped from 240 ug/l to 12 ug/l and they never recovered. These results suggest that selenium does not stay in solution as readily as either sulfate or boron.

Highway 59 Ash Landfill

Background

The Highway 59 Ash Landfill (License number 918) is located southeast of intersection of State trunk highways 59 (Arcadian Avenue) and 164 in the Town of Waukesha, in Waukesha County. The landfill was sited in a former sand and gravel quarry, which had ceased operation in 1940. The quarry was operated into a hillside, and had a pond that was approximately ten feet deep and covered 1.5 acres. Coal ash disposal occurred at the site from 1969 to 1978. The facility was originally licensed in 1970, and covered approximately 30 acres. The CCB wastes were derived from eastern bituminous coal fly ash and bottom ash from the Valley Power Plant. Approximately 500,000 cubic yards of solid waste (Class F ash) were deposited in this landfill. The ash has pozzolanic properties but is not self-cementitious (Ladwig, 1989). Waste thickness ranged from less than 10 ft in the southern one-half of the site to more than 40 ft in the northern one-half. Approximately 8,000 cubic yards of ash was deposited below the water table in the pond on the northern portion of the site, and 30,000 yards in the southern portion of the site behind an earthen dike constructed on clay till.

Numerous active private water supply wells are located around the landfill with the exception of the southern side where railroad & utility corridors and low marshy terrain limited construction. Most private wells in the area serve family residences, but there are a few two-family buildings and businesses with wells. Immediately west of the ash landfill is an old abandoned unlicensed (Pre-regulation) municipal waste dump operated by the City of Waukesha in the 1950's and 1960's. An old abandoned foundry sand waste disposal site lies directly west of the City's waste site. Monitoring data from these two sites were used to refine the groundwater flow patterns in the area and evaluate the migration of the CCB contaminant plume.

Geology and Hydrology

The geological setting of this site is generally similar to the Cedar Sauk site. There are unconsolidated glacial deposits below the site that range from 90-150 feet in thickness. These deposits were subdivided into four units: an upper sand and gravel unit (suitable as an aquifer), a sandy glacial-till (diamicton) unit, a silty clay unit, and a lower sand and gravel unit (also suitable as an aquifer). The lower sand & gravel unit is in direct contact with the weathered bedrock north of the site. The underlying bedrock is also Silurian Dolomite, which serves as the primary aquifer for the City of Waukesha and many Town of Waukesha. The CCB ash landfill is located up gradient of both

of the two other disposal sites mentioned above. The thorough evaluation of the limits of fill and the careful placement of monitoring wells between the ash landfill and the other nearby sites made the task of evaluating the water quality impacts complex but possible.

The groundwater monitoring around the landfill was initiated 1970's. WEPCO initiated a hydrological investigation in 1988, which included the installation of 13 additional monitoring wells and piezometers and the sampling of four private water supply wells adjacent to the site. A report to the Department in 1989 indicated elevated concentrations of sulfate and boron in private water supply wells to the west of the site. There were no NR140 water quality standards for boron at the time of the investigation in 1989. Subsequent groundwater monitoring consisted of sampling 20 monitoring wells and private water supply wells on a regular basis.

Groundwater Evaluation

The time vs. concentration (Tvc) plots for various monitoring wells and private drinking water supply wells show a variety of interesting trends. There is a decrease in sulfate levels for shallow wells near the landfill, but boron levels are increasing. This appears to document the continued migration of the high mobility of boron ions. The data for this site showed a close correlation between the occurrence of boron and high levels of sulfate. Molybdenum occurred in the CCB leachate in the low ug/l range. Although it does not have a NR140 standard, it served as a marker for leachate contamination from this site. This distinction was essential for discriminating between leachate contamination from CCB materials, and background levels of sulfate and/or the potential commingling of leachate plumes from the other two waste sites.

The highest levels of both boron and sulfate in monitoring wells were recorded in monitoring well nest W-8, which was located in native materials down gradient and in close proximity to the ash disposed of in the former quarry pond. The sulfate levels are usually three orders of magnitude higher than the boron, but some plots show sulfate concentrations decreasing while the boron levels have climbed (In Figures 1 & 2: Well W-8A in black, W-8B in blue, and W-8C in red).

The monitoring wells and shallow piezometers are screened in the sand and gravel units. The sand & gravel units located to the southeast, southwest, west, and northwest of the ash site show contamination above enforcement standards for sulfate and boron. Nine piezometers were completed in the bedrock in order to evaluate deeper groundwater impacts. The migration of CCB contaminates into the deep aquifer may have been hastened by a private water supply well (WJAC) near the western edge of the ash fill. The business operating the well may have been pumping it on a regular basis to wash the numerous cars on their sales lot. This well was the first private well to be impacted near the landfill. The concentration of sulfates made the water undrinkable and bottled water was supplied up until the time the well was abandoned and the facility placed on city water.

Elevated concentrations of sulfate and boron have been detected in deeper bedrock piezometers primarily northwest of the site. This information confirms that there is northwestern component to the contamination plume, which was not identified in the early stages of the off-site investigation. This component of flow may have been influenced by the installation of a higher capacity water supply well for a large retail store north-northwest of the landfill. This bedrock well has since been abandoned and the building connected to municipal water.

Remediation

WEPCO retained Natural Resources Technology, Inc. and Science & Technology Management, Inc. to evaluate various remedial action alternatives. An environmental contamination assessment (ECA) report was submitted in 1995. Between 1995 and 1998, WEPCO submitted additional reports assessing the degree and extent of the contamination from the landfill. In 1999, WEPCO submitted a remediation plan that included removal of the saturated ash from the northern portion of the site and replacement of ash above the seasonal high groundwater elevation (excess ash was returned in their Pleasant Prairie Power Plant). It has been noted that several samples of leachate from the on-site basin constructed for dewatering the ash showed molybdenum values two orders of magnitude higher than other samples.

The majority of landfill area was recapped with a synthetic geomembrane. Although ash fill covers about 26 acres, only 19 acres were recapped due to presence of buildings and other structures; some of the remaining ash landfill was covered by asphalt and is being used for parking. The cover design consists of 3-inches of existing cover fill, a

4-inch sand bedding layer, a 30-mil PVC geomembrane, a geocomposite drainage layer (consisting of geotextile and geonet), 2-feet of general fill, and 6-inches of topsoil.

As part of the remediation plan, WE Energies paid for the abandonment of contaminated private water supply wells and the connection to municipal water from the City of Waukesha. The restoration of clean water to the affected citizens was only possible through the cooperation of the local Town and City of Waukesha officials. WE Energies supplied bottled water to residents with affected wells after contamination was detected. Long-term environmental sampling of the monitoring network was initiated, which consists of wells and piezometers as well as numerous private water supply wells. The WDNR conditionally approved the remedial option selected by WEPCO. The Remedial Design Report was submitted on May 1999.

WE Energies made a case for not excavating the 30,000 cubic yards of saturated ash along the southern portion of this landfill. WDNR recommended the removal of these materials since they seemed to represent an additional source of leachate generation. The WDNR included additional environmental sampling of private water supply wells as a condition of approving the remedial action plan. The monitoring frequency for down gradient water wells is currently being increased in response to movement of the plume of CCB contaminants.

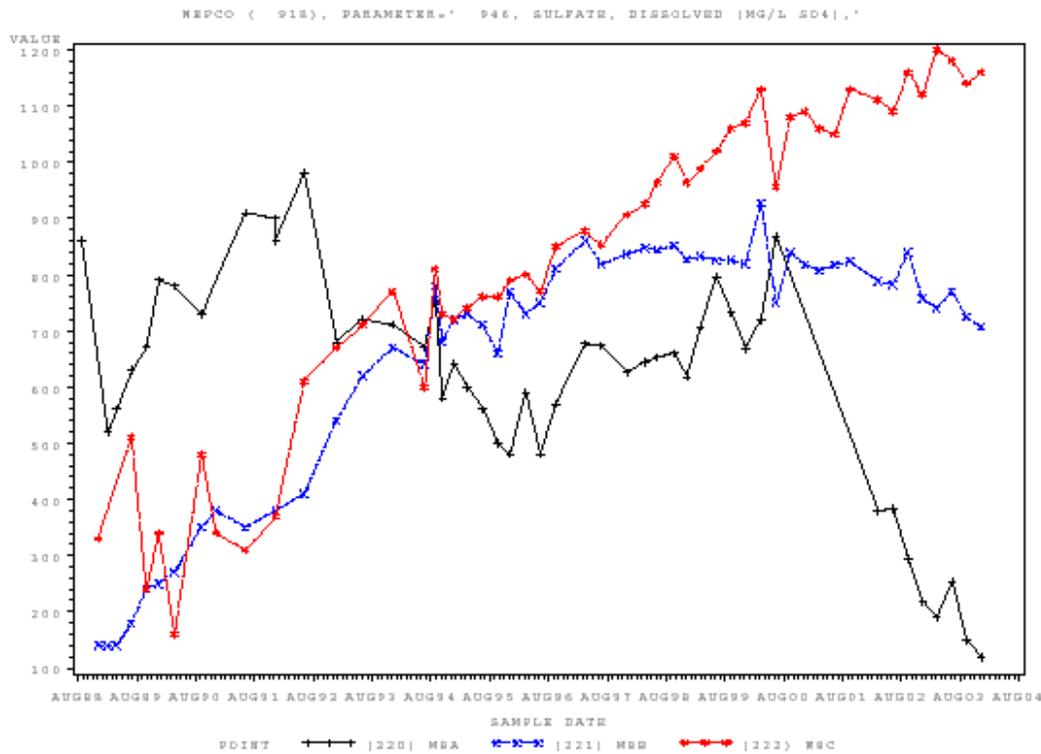
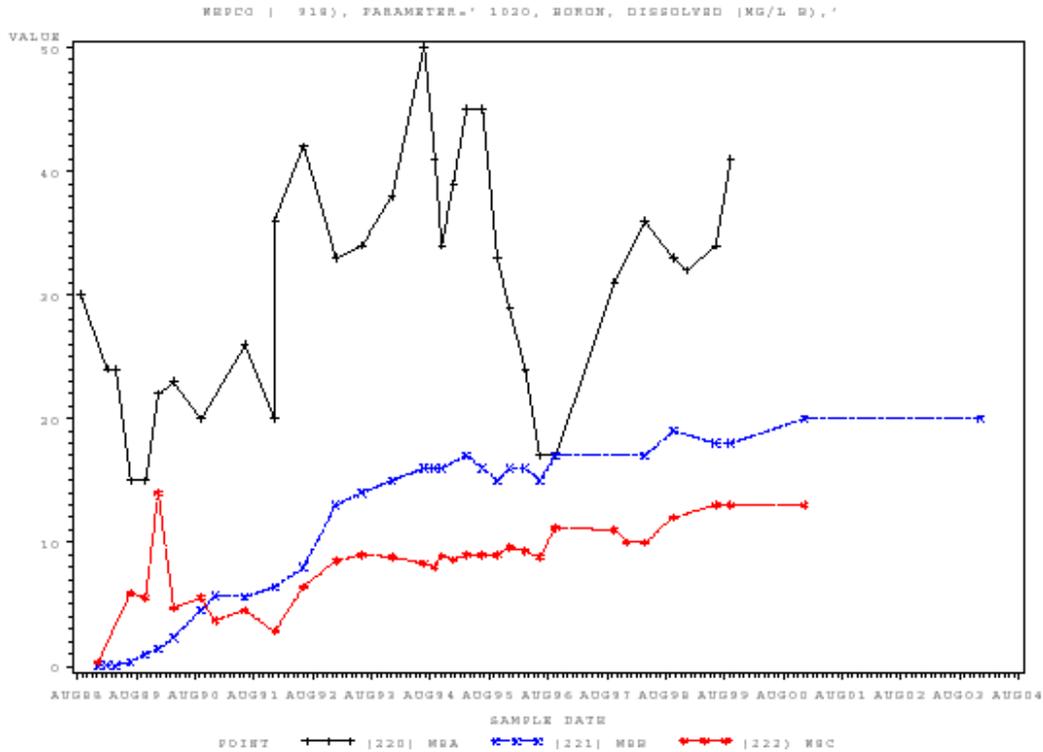


Figure 1

Figure 2

Conclusions

As mentioned earlier, this study was a simplified attempt to extract some meaningful conclusions out of a very large



amount of groundwater monitoring data. The results can be considered fairly conservative because the means were calculated based on the most contaminated well at the site using all historical data without regard to the well's placement (including those located in the waste mass itself) or status. While much of the analysis was inconclusive, several general trends can be discerned.

First of all, it appears that the engineered, recompacted clay liners at the active sites evaluated are effective at limiting groundwater contaminate releases. They are by no means perfect and there were measurable groundwater impacts in at least one monitoring well down gradient of each landfill. However, the contaminant levels were, in most cases, an order of magnitude lower when compared with the results from the closed, unlined landfills. This is especially true with regards to arsenic and boron concentrations.

A review of groundwater quality data at both active and closed disposal sites in Wisconsin indicates that boron, sulfate, arsenic and selenium are generally the best indicators of water quality impacts from CCB disposal. Manganese was also indicative of groundwater impacts, but there weren't enough sample results to adequately characterize the connection. Of all the parameters studied, boron seemed the best indicator of groundwater impacts. Arsenic was only useful at sites that sluiced their waste or used certain coals that were high in arsenic. Arsenic was also only found in appreciable amounts at wells that were screened within the waste or fairly close to the waste fill limits and may be documenting an ion halo effect. Further out, arsenic likely undergoes a geochemical interaction with the groundwater, complexes with iron hydroxides or organics and is therefore no longer chemically available to go into solution (Hem, 1992).

The data certainly suggests a strong connection between the practice of sluicing CCB waste and strongly elevated levels of certain compounds, including heavy metals and arsenic, in the leachate and groundwater. This problem is confined to older, closed sites, as sluicing CCB waste has not been practiced in Wisconsin since the mid-1980's.

In fact, the volumes of CCB waste disposed in Wisconsin has dropped dramatically as the power utilities have

switched to dry handling and have aggressively pursued beneficial reuse options for their CCB wastes. In fact, as of 2003, WE Energies now reuses more CCB waste than they produce. This is due to a WE Energies project at their Pleasant Prairie Landfill where they are mining an old site, reburning the ash, and beneficially reusing the subsequent ash. Most ash is beneficially reused in concrete as an additive or as fill in construction projects.

As for the site specific studies, it should be noted that the results at Cedar Sauk and Highway 59 Ash Landfills were the result of a very specific set of circumstances that should not be extrapolated to any specific site or use except in a very general sense. Each disposal site was unique with respect to the geologic environment and the type and nature of the CCB waste. All of these factors combined to determine the degree and extent of contamination at each site, including the type of contaminant found.

Another factor that should be considered, especially at the Cedar Sauk Landfill, was that the most contaminated monitoring wells were located and screened in waste. In effect, they were measuring leachate quality, not groundwater. This is hardly unique to Cedar Sauk, as the statewide study revealed several other old sites where the highest contamination was from wells that were constructed within the waste mass as generally recommended by the US EPA at the time of the installation of the wells. It is also significant that some of the highest contamination levels were detected before the sites were closed and capped. These early results certainly can skew the mean values higher.

The documented impacts to groundwater quality at both sites were also exacerbated by the geology of the disposal sites, which included fractured bedrock and highly transmissive outwash sands and gravels with little attenuation capacity. Inadequate capping of the waste masses until recently also allowed precipitation to infiltrate through the waste mass and pick up contaminants that was then transmitted to the groundwater.

However, it is important to note that, under current rules, the siting and facility design of either of these landfills would not be acceptable if they were still operating. The State of Wisconsin has very vigorous design and monitoring standards for the very reason that many areas of the State contain high-quality shallow groundwater that is used and valued by its citizens.

It should also be noted that the remedial measures installed by WE Energies at both these sites have been effective at dramatically reducing groundwater impacts. The Cedar Sauk Landfill remediation project shows that proper capping of a CCB landfill is highly effective at reducing negative impacts to groundwater quality. Water quality at the Highway 59 site has improved, but several plumes are still migrating through both the sand & gravel and bedrock aquifers. The size and extent of the contamination makes off-site groundwater extraction difficult when balanced against the decreasingly available water resources for the City of Waukesha.

Acknowledgments

The authors wish to acknowledge WE Energies and EPRI for data on boron values for coal ash analyses.

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Table 2

Parameter	Boron (mg/l) ²	Sulfate (mg/l)	Arsenic (ug/l) ³	Selenium (ug/l)	Other Compounds	Status
Facility name						
Alliant - Columbia Landfill Lic# 2325	<u>Range</u> 0.18-4.3 <u>Mean</u> 3.2	<u>Range</u> 210-870 <u>Mean</u> 457	<u>Range</u> 5-170 <u>Mean</u> 40	<u>Range</u> 0.5-7 <u>Mean</u> 1.86	<u>Lead</u> - 10-140ug/l <u>Manganese</u> - 320-830 ug/l	Closed Sluice Site
WP&L - Rock River Landfill Lic# 0728	<u>Range</u> 1.3-3.7 <u>Mean</u> 2.2	<u>Range</u> 53-900 <u>Mean</u> 322	<u>Range</u> 7-11 <u>Mean</u> 9	<u>Range</u> 1-5 <u>Mean</u> 3		Active
WP&L - Nelson Dewey Landfill Lic# 2525	<u>Range</u> 2.3-28.4 <u>Mean</u> 9.23	<u>Range</u> 75-2100 <u>Mean</u> 1144	<u>Range</u> 5.5-800 <u>Mean</u> 197	<u>Range</u> 10-320 <u>Mean</u> 78.8	<u>Manganese</u> - 720- 1200ug/l <u>Chromium</u> - 45-68 ug/l	Closed Sluice Site
WEPCO - Pleasant Prairie Landfill Lic# 2786	<u>Range</u> 1.7-3.6 <u>Mean</u> 2.8	<u>Range</u> 1120- 1340 <u>Mean</u> 1215	No Data	<u>Range</u> 3.1-11 <u>Mean</u> 5.65	<u>Mercury</u> - 0.3-1.6 ug/l	Closed Being Mined
WEPCO - Hwy. 32 Landfill Lic# 2801	<u>Range</u> 0.5-3.7 <u>Mean</u> 0.85	<u>Range</u> 87-1100 <u>Mean</u> 447	<u>Range</u> 25-35 <u>Mean</u> 30	<u>Range</u> 0.9-4 <u>Mean</u> 2.4		Active
WP&L B Edgewater 1-4 Landfill Lic# 2524	<u>Range</u> 29-120 <u>Mean</u> 69	<u>Range</u> 640-2600 <u>Mean</u> 1661	<u>Range</u> 150-540 <u>Mean</u> 364	<u>Range</u> 1-8.4 <u>Mean</u> 4.7		Closed Sluice Site
NSP B Woodfield Landfill Lic# 3233	<u>Range</u> 0.17-0.24 <u>Mean</u> 0.24	<u>Range</u> 40-96 <u>Mean</u> 63	<u>Range</u> 7-11 <u>Mean</u> 9.5	<u>Range</u> 0.5-8 <u>Mean</u> 3	<u>Lead</u> - 28-38 ug/l <u>Manganese</u> - 380-510 ug/l	Active
Dairyland B Belvidere Landfill Lic# 2927	<u>Range</u> 1.4-9.1 <u>Mean</u> 5.25	<u>Range</u> 750-2000 <u>Mean</u> 1403	No Data	<u>Range</u> 2.9-30 <u>Mean</u> 18		Active
WPSC - Weston #3 Landfill Lic# 2879	<u>Range</u> 0.22-2.68 <u>Mean</u> 1.02	<u>Range</u> 51-791 <u>Mean</u> 210	<u>Range</u> 6.7-16.7 <u>Mean</u> 10.7	<u>Range</u> -91.4 <u>Mean</u> 23	<u>Lead</u> - 0-9.7 ug/l <u>Manganese</u> - 460-1620 ug/l	Closed
WE Energies - Cedar Sauk Landfill Lic# 0603	<u>Range</u> 11-186 <u>Mean</u> 101	<u>Range</u> 520-1923 <u>Mean</u> 1397	No Data	<u>Range</u> 1-730 <u>Mean</u> 77		Closed
WE Energies - Caledonia Landfill Lic# 3232	<u>Range</u> 0.2-9.3 <u>Mean</u> 4.37	<u>Range</u> 260-430 <u>Mean</u> 311.6	<u>Range</u> 1-17 <u>Mean</u> 5.26	<u>Range</u> 0.6-5 <u>Mean</u> 1.6	<u>Manganese</u> - 10-110 ug/l	Active

WE Energies - Highway 59 Landfill Lic# 0918	<u>Range</u> 15-50 <u>Mean</u> 30.7	<u>Range</u> 836-1170 <u>Mean</u> 1034	<u>Range</u> 2-5 <u>Mean</u> 3.04	<u>Range</u> 0.002-20 <u>Mean</u> 7.84	<u>Manganese</u> - 490-670 ug/l	Closed
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- 1 - Values represent the range and mean of all detects listed in the GEMS database for the most impacted groundwater monitoring well for each specific parameter at the facility.
- 2 - Milligrams per Liter (mg/L) or parts per million (ppm).
- 3 - Micrograms per Liter (ug/L) or parts per billion (ppb).

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LEACHING OF ARSENIC FROM A FLY ASH DUMP IN BEVERLY, MASSACHUSETTS

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Abstract

The Vitale Dump in Beverly, Massachusetts is a former sand and gravel pit where sea quenched coal fly ash, construction debris, and miscellaneous wastes were dumped for many years beginning in the 1950s. The site was a primary ash disposal area for New England Power Company who operated a power generation station in Salem, Massachusetts. Ash deposits buried at the site range in thickness from 14 to 36 feet. Ground water has been measured from 10 to 20 feet below grade leaving large portions of the dump below the water table. Arsenic concentrations in ground water at the up-gradient wells on the property were measured at 6 ug/L while down-gradient wells contained arsenic at 386 ug/L.

Airport Brook flows through the site discharges to Wenham Lake about 1,500 feet east. Wenham Lake is the public water supply for the 80,000 residents of Salem and Beverly, Massachusetts. Fly ash has been eroding from the site into Airport Brook. Testing in Wenham Lake has detected several feet of fly ash in a portion of the Lake. This has promoted a significant debate concerning actual and perceived public health risks. Current remediation plans include excavation of sediment along Airport Brook and Wenham Lake and placing the dredged materials on the Vitale Site. A solid waste cap will then be placed at the Vitale Dump site. Soccer fields have been proposed on the site after closure. At this time, the arsenic concentrations in ground water do not appear to require any active treatment to comply with the applicable State and Federal regulations.

Introduction and Site History

The Vitale Dump was owned by George H. Vitale and George W. Vitale and was initially developed during the 1940s for sand and gravel excavation which continued until the 1950s. The City of Beverly took ownership of the property in 1987.

The study area (Site) consists of the Vitale Dump, approximately 45 acres along Airport Brook and 25 acres within the northwest portion of Wenham Lake. Surface water drainage is northeasterly via Airport Brook into Wenham Lake which is the primary drinking water source for the 80,000 residents of Beverly and Salem Massachusetts. The Vitale site is divided into two distinct areas, the upland portion, which includes the fly ash dump (15 acres) and lowland portion which includes Wenham Lake and Airport Brook.

Based on a review of historical aerial photographs for the area from 1938, the Site was undeveloped and consisted of agricultural land. Following sand and gravel excavation, the Vitales used the fly ash, which was not considered a waste, to fill the sand and gravel pit. New England Power Company (NEP) had an agreement with the Vitales to dispose of saltwater-quenched fly ash from NEP's Salem Harbor Power Station beginning in the 1950s.

There have been a variety of consultants who have investigated the Site over the years, including Haley and Aldrich (1985), Gulf of Maine (1986), Normandeau (1988), Weston and Sampson (1989), Ransom (1995), Haley and Aldrich (2003) and Blaslund, Bouck & Lee (2003) and Hart Crowser (2003). Ransom reviewed aerial photographs dated 1938, 1957, 1969 and 1978 correlating with the known site history. The 1938 aerial photograph showed that the garage building had not yet been constructed, but a dirt road, coinciding approximately with the present entrance to the Site, extended north across the Site and divided into two paths extending across the northwest and northeast portions of the Site. In the 1969 aerial photograph, fly ash, lacking any vegetative cover, was apparent across most of the eastern half of the Site extending from the north of the garage to the northeast Site boundary and to the south abutting Airport Brook. Due to the lack of vegetative cover, erosion gullies are apparent along the edges (flank) of the fill in the area abutting the wetlands along the eastern border of the site and also along Airport Brook. The

conditions at this time as shown in the aerial photograph correlate with local complaints of erosion during the early 1970s. The garage had been constructed at the time of the 1969 photo (Ransom, 1995). The 1978 study shows the fly ash area to be partially vegetated.¹

Today, most of the Site is filled with 15 to 30 feet of fly ash and municipal fill and is covered with scrub vegetation and invasive trees. Wetland areas adjacent to Airport Brook are impacted with fly ash sediment from erosion of the Site. Fly ash has been found throughout the Airport Brook and fly ash delta has formed over the last thirty years at the discharge point to Wenham Lake. In addition to fly ash, roofing material, asbestos siding, household appliances, car bodies and empty steel drums were also reported to have been disposed of at the Site during the Vitale's and City's ownership.

History of Non-Compliance

Until the property was taken by the City of Beverly in 1987, the Vitale Brothers had a history of non-compliance with State and local regulation. In 1969, the Massachusetts Department of Environmental Quality Engineering (DEQE), now the Massachusetts Department of Environmental Protection (MADEP), reported that fly ash was being disposed of at the Site without a permit from the local Board of Health. In 1971, a restraining order was issued to the Vitales by the Massachusetts Department of Wetlands to stop any further work that would impact the wetlands on the Site. In 1973, a neighbor on Trask Street, 500 feet east of the Site, complained of flooding of his yard due to damming caused by the erosion of the fly ash. Similar notices of violations were cited during that time period. In 1975, the Massachusetts Department of Public Health issued an order to the Vitales to prevent siltation and pollution of the stream, cut back the slope to the wetland and grade, cap and vegetate the filled areas of the Site. The Vitales ignored this and subsequent State orders, resulting in litigation. In 1987, the City assumed ownership of the property from the Vitales due to their failure to pay real estate taxes. Beverly Fire Department records indicate that the garage building located on site burned in October of 1992.²

There has been considerable debate about the presence of fly ash at the Site. In 310 CMR 40.0006, the "presence of contaminants which are attributable to coal ash or wood ash associated with fill material are considered to be background." Pursuant to its regulations, MADEP continues to consider that status of coal ash to be exempt from closure requirements under the Massachusetts Contingency Plan and does not treat ash as a solid waste requiring a hybrid closure. This paper discusses current activities at one high profile ash disposal site in northeastern Massachusetts. It details how New England Power in cooperation with local and State governments as well as local environmental activists worked to a common goal to stabilize and close out the Vitale Dump Site and eliminate any potential public health and ecological risks associated with it.

Surface Water Hydrology

The Airport Brook watershed starts in the wetlands located adjacent the Beverly Airport and the west slope of Lords Hill. Surface water flow is southeast to the Site where it follows past the southern toe of the fly ash embankment and on through a culvert near the entrance to the Site. As the stream flows to the east and northeast it passes through greater than 20 feet of exposed fly ash on both banks at its deepest point. From this deep channel, the brook turns northerly following a man made channel toward Route 97. The Airport Brook then goes northeasterly through a culvert under Route 97 at Elevation. 37 feet before continuing through the nearby Dearborn property riparian zone into the wetland zone at approximately Elevation 35 feet and on to the northwest cove of Wenham Lake. High water in Wenham Lake is typically maintained below Elevation 34.4 feet.

Based on stream discharge and piezometer monitoring results from Haley and Aldrich (H&A) (2004), Airport Brook is a losing stream, at least throughout most of the periods between May and September of 2001 and 2002. As a result, this allows the surface water to infiltrate downward into the groundwater flow system. Any metals or organic chemicals present in Airport Brook would be available to enter groundwater.

Stream discharge measurements by H&A also confirm that Airport Brook is a losing stream through much of the

¹ Ransom Environmental, Phase II Comprehensive Site Assessment, April 2001

² Ransom Environmental, Phase II Comprehensive Site Assessment, April 2001

year, as there is little to no measurable stream flow during extended periods without rainfall. Therefore, most of Airport Brook functions as a seasonal stream where, during some periods of the year, the groundwater elevation is above the surface water elevation (i.e. a gaining stream) and most of the year the groundwater level is below the surface water elevation (i.e. a losing stream).³

Hydrogeologic Setting

The Vitale Site study area is located within a depressed region of low hills and coastal drainage basins referred to as the Seaboard Lowland Section of New England Physiographic Province. The underlying bedrock is highly faulted igneous intrusions called the Cape Ann Plutonic Series. Surficial deposits (soil) consist primarily of glacial and alluvial sediments that overlie bedrock. The major landforms in the vicinity include drumlins to the northwest and extensive sand and gravel kame and esker deposits at Wenham Lake.

Bedrock Geology

The Cape Ann Plutonic Series is unmetamorphosed granitic and dioritic rocks dominated by several large scale fault zones, trending to the northeast. The closest fault zones are the Ipswich Bay Fault and the Mystic Fault. Within the study area, two primary rock types occur, the Wenham Monzonite a light colored, medium grained granitic rock consisting primarily of the mineral feldspar that outcrops in several locations west of Route 97 and the Salem Gabbro-Diorite a dark colored rock of heterogeneous composition and texture locally identified as a medium to fine grained hornblende diorite.

Surficial Geology

Overlying bedrock is glacial material comprised of till, outwash, and glaciolacustrine deposits. In portions of the Site, they are locally overlain by more recent lacustrine, alluvial, and organic deposits along Airport Brook and low wetland areas. Fly ash and a variety of miscellaneous debris are present throughout the area and particularly in the low lying areas.

Glacial till deposited in elongated hills known as drumlins were formed at Lords Hill, north of the Vitale Property, and west of Wenham Lake about 25,000 years ago. These thick till deposits consist of a very dense mixture of clays and silts to sand and gravel with variable boulder content.

Years ago, the area below the Vitale Dump was formed by glaciers. Glacial meltwater streams deposited thick layers of sand and gravel onto the edge of drumlins within the study area. At various times while the ice retreated, one meltwater course flowed along the valley into Wenham Lake. Wenham Lake is made up of a group of glacial kettle holes created by the channeling of outwash sediments through stagnant glacial ice and the incorporation of ice block into meltwater deposits during the retreat of the glaciers. Sands and gravels underlie the lake and form numerous ice contact features including eskers, kame mounds, and kame terraces within the lake and the surrounding area. In this area, these deposits generally occur below Elevation 80 feet and overlie denser and less permeable glacial till deposits. The northwest cove of Wenham Lake is a glacial kettle bounded on the northeast by the remains of an esker.⁴

On the Vitale Property, outwash deposits reached a thickness of approximately 15 feet to 25 feet and were extensively quarried before and during the 1950s before being replaced with fly ash. During the waning stages of glaciation, melting of glacial ice and the initially poorly developed drainage patterns led to extensive but temporary impoundments of meltwater and the creation of small glacial lakes.⁵

³ Haley & Aldrich, Draft Data Report Vitale Fly Ash Investigation, February 2004

⁴ Haley & Aldrich, Draft Data Report Vitale Fly Ash Investigation, February 2004

⁵ Haley & Aldrich, Draft Data Report Vitale Fly Ash Investigation, February 2004

Discussion of Sediment and Water Quality

The Vitale Site is the major source of fly ash to the surrounding environment. Fly ash has migrated with surface water along Airport Brook from the Vitale property source area. Some of the fly ash has been deposited in Airport Brook and the remaining fly ash has been deposited in the cove and adjacent area in Wenham Lake approximately 1500 feet east.

Arsenic is one of the more toxic and most studied metals found in fly ash. Other metals detected (primarily in ash sediments) include barium, beryllium, chromium, nickel, strontium, vanadium and zinc. Arsenic has unique environmental mobility characteristics. Arsenic's can occur in the environment in four oxidation states (-3, 0, +3 [As (III)], and +5 [As (V)]). The two most common forms are As (III) and As (V). Of these forms, As (III) is the more mobile and toxic. This section summarizes the results of numerous sampling efforts conducted in the surface water of Wenham lake, the groundwater in overburden, deep overburden and bedrock throughout the study area, in sediment in the Wenham Lake Cove, and in the porewater of the sediment.

Surface Water

Surface water testing by Ransom Environmental generally contained non-detectable concentrations of arsenic in the brook near the Vitale Dump, except at SW-5, where arsenic was found at 7 micrograms per liter (ug/L). Results for the 2003 surface water samples within Wenham Lake indicated that the total arsenic concentrations ranged from 0.19 ug/L to 0.67 ug/L. That range is near the low end of the reported naturally occurring range (generally less than 1 ug/L) for arsenic in freshwater. One important consideration at this Site is that both the Federal and State drinking water standards for arsenic are currently 50 ug/L and will be lowered to 10 ug/L within the next several years. None of the detected concentrations of arsenic or any other constituent associated with fly ash have been detected in surface waters at concentrations in excess of the applicable current or proposed drinking water standards. The intakes for the Salem Beverly Water Supply System are located approximately 4,000 feet from the fly ash deposits in Wenham Lake Cove.

Ground Water Sampling Analysis

Ground water testing has occurred at the Vitale Dump for over 15 years. Table 1 contains the results of wells within the footprint of the ash deposit that have been tested first in July 1988.

Table 1. List of Wells With Elevated Dissolved Arsenic (ug/L) Within the Vitale Dump

Date	MWV-1S	MWV-2S	MWV-2D
7/12/88	BDL (50)	58	62
7/18/96	73	74	386
4/16/99	124	143	358
8/19/99	132	82	331
7/24/02	150	NT	NT
1/29/03	90	NT	NT
Date	MW-4S	MWV-3S	MWV-3D
12/88	191	52	56
7/18/96	97	58	43
4/16/99	BDL	BDL (50)	BDL (50)
8/19/99	116	NT	82 (1)
7/24/02	NT	NT	45
1/29/03	NT	NT	34

Note: (1) Testing date was 10/22/99; (50) indicates method detection limit in ug/L

Table 2 contains dissolved arsenic samples from locations hydraulically down gradient of the fly ash deposit.

Table 2. Arsenic Dissolved in Wells Down gradient of the Vitale Dump(ug/L)

Well	July/August 2002	January/February 2003
HA – 100D-OW	2	NT
HA – 101S-OW	BRL(1)	NT
HA – 101I-OW	13	17
HA – 101D-OW	BRL (1)	BRL (1.4)U
HA – 102S-OW	5	4J
HA – 102I-OW	BRL (1)	9
HA – 102D-OW	2	BRL (.0014)U
HA – 103D-OW	3	NT
HA – 104I-OW	.003	BRL (1.4)U
HA – 104D-OW	BRL (1)	BRL (1.4)U
HA – 105-OW	BRL (1)	1J
HA – 105I-OW	BRL(1)	NT

Terms: **ug/l** - micrograms per liter; **BRL** – Below Reporting Limit noted in parenthesis; **J** - values indicated an estimated concentration as the compound was detected below the BRL; **NT** – not tested

The methodology for well testing has evolved over the past 15 years and the more recent samples were collected using low-flow purging techniques where practical. Where recharge to well was slow or wells went dry, peristaltic pumps and bailers were used. These differences in collection techniques over the years likely had some small but non-quantifiable impact on the measurement of dissolved arsenic in ground water. Studies have shown that low flow sampling cause very little turbulence within the borehole and as a result natural aquifer materials (including fly ash particles) are not transported into the sample matrix.

The highest concentrations of dissolved arsenic were found in MWV-2D (Table 1) at 386 ug/L in July 1996 and 331 ug/L in August 1999. At location MWV-1S concentrations appear to reach their maximum value of 150 ug/L in July 2002 while declining to 90 ug/L in the January 2003. At location MWV-3D concentrations were measured at 45 ug/L in July 2002 while decreasing to 34 ug/L in the January 2003. MWV-1S is screened in coarse sand, inorganic silt and some cobbles. Both MWV-2D and MWV-3D are partially screened within the fly ash.

As shown in Table 2, samples collected downgradient of the Vitale Dump have consistently shown very low to non-detectable concentrations of arsenic. Wells screened both in the deep and shallow aquifers generally show low to non-detectable concentrations of dissolved arsenic, which suggests the As(III) present at the fly ash deposits has speciated to As (V), the more immobile form.

The ground water data suggests that As (III) desorbs/dissociates from fly ash into the porewater of the fly ash deposits. Percolating rainfall delivers the porewater to groundwater. The As (III) can then speciate to As (V) in ground water depending on redox/pH and sorb to soil particles. Groundwater located downgradient of the fly ash deposit has higher dissolved oxygen and redox potentials and, as a result, has lower concentrations of arsenic.

As (V) is also scavenged by sediment particles (particularly Fe and Mn minerals) and deposited within the aquifer or on surface sediment in Airport Brook, which ultimately discharges to Wenham Lake. However, this second process, although it does occur, is not the dominant process by which ash was delivered to Wenham Lake Cove. Due to the lack of vegetative cover, fly ash dumped at Vitale found its way by overland flow into Airport Brook and Wenham Lake. As stated earlier, erosion gullies were apparent along the edges (flank) of the dump and abutting the wetlands along the eastern border of the Site.

Sediment Sampling Analysis

Sediment is a complex mixture of solids, gases, and liquids. Within Wenham Lake cove, sediment solids include natural mineralogical material, natural organic materials, and fly ash particles. Sediment gases are generally created from decomposing organic material and share the spaces between sediment solids (i.e., pore spaces) with sediment liquids. Sediment liquids are comprised of porewater, which is the water that is located in the pore spaces between the sediment solids. The cove sediment porewater is derived from groundwater discharging from the upland

subsurface. This groundwater seeps into and through alluvial deposits that underlie the cove shallow sediment. Eventually, the pore water will flow to surface water.

Fly ash has been deposited in Wenham Lake Cove when the fly ash was eroded from the Vitale Dump, suspended in the surface water column and subsequently transported via Airport Brook. As the flow velocity drops at the terminus of Airport Brook in the cove area, attributable to significant widening and deepening of the Lake, the flow velocity drops and the fly ash is deposited.⁶

Fly ash sediment cores were collected, in the winter of 2001, in Wenham Lake Cove by the Wenham Lake Watershed Association (WLWA). Results indicated that fly ash transport and deposition was a dominant mechanism for ash deposition in this area. This transport of fly ash is the result of the unstable condition of the Vitale Dump property for many years. Outside the channel alignment, a higher proportion of natural sediment, compared to fly ash, was observed in shallow sediment sampling.

The average arsenic concentrations for 41 shallow sediment samples collected by H&A within the fly ash deposit area was approximately 47 milligrams per kilogram (mg/kg), compared to a naturally occurring sediment arsenic concentration range of about 20 to 25 mg/kg for ponds and lakes in eastern Massachusetts (Menzie-Cura, 2003). The highest sediment arsenic concentrations were found near the eastern edge of the fly ash deposit. Arsenic concentrations in bulk sediment samples do not correlate well with the amount of fly ash in sediment sample.

The sedimentation environment and processes that occur within the Lake are a key to understanding the transport and fate of the fly ash deposit. The sedimentation in the cove area of Wenham Lake and the extent of the fly ash deposit are well characterized because the fly ash has been transported and deposited over a 50-year period. The fly ash can also be distinguished visually from the natural sediment material.⁷ (BBL, 2003)

Sediment Pore water

Unlike the bulk sediment laboratory chemical analysis results, there was an apparent correlation between the occurrence of fly ash and increased arsenic concentrations in sediment porewater (as compared to background concentrations in arsenic). BBL and Hart Crowser collected a variety of porewater samples for metals analysis. Increased sediment porewater concentrations for the various forms of arsenic were detected in four of the five shallow fly ash/sediment samples collected from within the fly ash area. A sample collected outside the fly ash deposit also did not show increased porewater concentrations of arsenic

The porewater related As (III) is likely to originate predominantly in the Vitale Dump area, based on the relatively elevated arsenic concentrations measured in sediment porewater. During seasonal periods where the deeper water at the sediment surface water interface is anaerobic, the As (III) is likely dissolved in the water column. As the DO concentrations increase with seasonal changes, the As (III) likely speciates to As (V). The surface water sampling results indicate that Airport Brook and Wenham Lake appear to follow this typical behavior. As (V) is likely to bind strongly to sediment solids and does not dissolve as readily in the water column as As (III) particularly along the stream where oxygen is present. As (III) desorbs/dissociates from fly ash into sediment porewater. Advecting groundwater transports As (III) to surface water and depending on redox/pH, As (III) speciates to As (V). As (V) can be scavenged by suspended sediment particles (particularly Fe and Mn minerals) and deposited in surface sediment.

Microbial activity may also play an important and unpredictable role in arsenic mobility and fate. Microbial reductive dissolution of iron hydroxides and associated release of retained arsenic has been proposed as a probable mechanism for the release of arsenic within sediments. (National Research Council, 2001) Additionally, some aquatic organisms metabolize arsenic, forming organoarsenic compounds.⁸

Summary of Risk Characterization

⁶ Hart Crowser/Blasland, Bouck and Lee, Wenham Lake Draft Feasibility Study, Executive Summary, November 2003

⁷ Hart Crowser/Blasland, Bouck and Lee, Wenham Lake Draft Feasibility Study, Executive Summary, November 2003

⁸ Hart Crowser/Blasland, Bouck and Lee, Wenham Lake Draft Feasibility Study, Executive Summary, November 2003

Outside of the source area of the original fly ash deposit on the Vitale Site, there are no exceedences of Federal or State drinking water standards (50 ug/L for As, will be 10 ug/L in 2006). The Site is currently being evaluated for trespassers/recreational users. The likely conclusion will be No Significant Risk (NSR). A geotextile and soil cap has been proposed for the Vitale Dump property. The City of Beverly is also interested in placing athletic fields on top of the graded cover. There does not appear to be any widespread opposition to the final end use although several individuals have voiced their personal concerns for public health if children are allowed on the closed site.

Current surface water data indicates that detected concentrations of arsenic are all below ecological benchmark values. Menzie-Cura previously completed an Ecological Risk Assessment (ERC) and Hart Crowser completed a supplemental review focusing on fly ash deposit. The weight of evidence presently indicates “no significant risk” for the ecological receptors and pathways.

Clean up of Wenham Lake-A Public Water Supply

Wenham Lake is used as a public drinking water supply that provides drinking water to the approximately 80,000 residents of the City of Beverly and Salem. Originally, the proposal to address the fly ash in Wenham Lake was to hydraulically dredge the fly ash that could not be excavated by conventional means. After numerous discussions with various stakeholders, including the City of Beverly, City of Salem, MADEP and the Salem and Beverly Water Supply Board (SBWSB), NEP decided to reevaluate whether dredging was feasible or necessary.

A Feasibility Study (FS) report was prepared by Blaslund, Bouck & Lee (BBL) & Hart Crowser (HC) on behalf of NEP. The FS looked at numerous technologies that were then grouped into viable alternatives for removing the fly ash that had eroded from the Vitale property and along Airport Brook and was deposited into Wenham Lake. Based on previous investigations conducted at the Site, there are approximately 7,800 cy of fly ash present in Wenham Lake that has eroded from the former Vitale property. This fly ash covers an area of approximately 11 acres and ranges in thickness from a few particles of fly ash intermixed with natural sediment to up to 3 feet of fly ash.

The first alternative proposed is Monitored and Natural Attenuation (M&NR) for Wenham Lake. This alternative would monitor water quality and establish “Action Levels” to respond to results and, if necessary, re-open FS. In addition, this alternative would monitor sedimentation to evaluate natural restoration processes and enhance understanding of the Wenham Lake system. The positive aspects of this alternative are that water quality remains within acceptable levels and natural sedimentation buries fly ash, reducing contact with surface water and decreasing migration of porewater to surface water. The negative aspects are the perception that nothing is being done, fly ash remains in the Lake, and long-term commitment to monitoring (50 years used for cost comparison). The Net Present Value of this alternative was estimated to be \$4,200,000.

The second alternative proposed is Capping. This alternative would carefully place up to 40,000 cy of sand/apatite mineral capping material over the geotextile area of 12-1/2 acres. The positive aspects of Capping are isolating fly ash below the cap, reducing direct contact with surface water, and decreasing migration of porewater As to surface water. The negative aspects are not testing in water supplies, loss of storage capacity and aesthetics, implementation issues, and long term post-construction monitoring. The Net Present Value of this alternative was estimated to be \$9,500,000.

The third alternative is Removal. This alternative would remove some or all of the fly ash/sediments from the Lake bed. The positive aspect of this alternative would be the removal of fly ash from the drinking water reservoir. The negative aspect is the significant potential for the “cure being worse than the ailment”. Large volumes of water will require treatment or disposal (and replacement) and disturbing the Lake bottom could cause uncontrollable releases of arsenic to surface water. The Net Present Value of this alternative was estimated to be \$3,500,000 to \$5,700,000.

The fourth alternative would include a combination of Limited Mechanical Removal (in specific dry areas of the Lake basin during extreme annual low water (generally in October) and Monitoring and Natural Restoration (M&NR) within the areas where sediment removal was not possible.

The FS concluded that the recommended alternative for addressing the fly ash is a combination of Limited Mechanical Removal (in specific dry areas of the Lake basin) and Monitoring and Natural Restoration (M&NR). To date, none of the compounds that are associated with fly ash have been detected in surface water in the Lake at

concentrations greater than one tenth (1/10) of the current Federal drinking water standards. The FS concluded that the recommended alternative for addressing the fly ash is a combination of Limited Mechanical Removal (in specific dry areas of the Lake basin) and M&NR. The elevation of the Wenham Lake reservoir has large stage fluctuations. The fly ash in Wenham Lake that is exposed and dry during low water elevations (typically in the late fall) will be removed with conventional excavation means and placed on the Vitale site prior to final stabilization. Because this material will no longer be under water, impacts to the water supply are eliminated.

Clean up Plan for the Vitale Fly Ash Site

Prior to the removal of the exposed fly ash from Wenham lake, the following activities will be completed to address the remainder of the fly ash that has eroded from the Vitale property:

- The former Vitale property and the slopes of the fill area will be properly stabilized, which will eliminate future erosion of fly ash off of the property.
- Ash will be removed from the wetlands and Airport Brook, and these resource areas will be restored.
- The ash that is exposed during the lowest feasible water level in Wenham Lake will be removed to the extent possible with conventional excavation means (in the “dry”).
- The entire surface of the consolidated ash pile at the Vitale property will be covered with geo-textile, 12-inches of sand, and six-inches of loam, and then graded and seeded to control erosion.
- A monitoring program will be implemented to ensure that the limited extent of impacted groundwater does not change as a result of the fly ash consolidation, slope stabilization, and wetland restoration.

The City of Beverly is interested in placing athletic fields on top of the graded cover and any future concern regarding exposure to the fly ash will be eliminated by the cover system.

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ENVIRONMENTAL CONCERNS AND IMPACTS OF POWER PLANT WASTE PLACEMENT IN MINES

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Abstract

A review of the placement of power plant wastes in coal mines in Pennsylvania, West Virginia, Indiana, New Mexico, and other states reveals that ground waters and surface waters are being degraded by this activity. Data from designated ash monitoring points show rises in the concentrations of total dissolved solids, sulfate, manganese, iron, boron, and a variety of trace heavy metals in these waters that significantly exceed baseline concentrations. The degradation is observed for mine water degraded by acid mine drainage and water not degraded by mine acidity. The data raise fundamental questions about the adequacy of safeguards in permits authorizing ash placement in coal mines and the assertions that alkaline coal ashes are inherently reliable and safe materials for preventing acid drainage or remediating abandoned mined lands.

Introduction

Under the rubric of *beneficial use*, millions of tons of coal combustion ashes and other power plant wastes are being placed in mine sites across the country. Most of this ash is being placed in active, SMCRA-permitted mining operations. Some is being placed at abandoned sites in attempts to remediate them. The position of the coal companies, the generators, and the regulators of these various placement projects is uniform; ash placement projects at mines are beneficial and never cause environmental damage. There is no type, quantity, placement method, or placement setting of these materials that has ever had a negative effect in a mine setting.

All of the scientific evidence to date shows that placement of these materials at SMCRA mine sites has either been environmentally beneficial or had no negative effect. (Vories, OSM, 2002)

This is remarkable, and wonderful, if it can be demonstrated. If, in the mining setting, the impacts from ash placement projects are universally beneficial or neutral, there is something about the mine setting that is unique. Because, these same materials disposed in any other setting have had negative effects. The mining setting alone is reputed to have a perfectly clean slate.

A review of data from mine placement sites does not support such a position. Negative effects from ash placement operations exist and they are common at sites where monitoring is performed at places and times that are capable of detecting impacts. The data show that there is degradation of water resources at a variety of sites where ash placement is an integral part of the mine reclamation or site remediation. These are the physical measurements that document that what happens.

What does reasonably constitute a negative impact? With respect to impacts to ground water and surface water, it is argued that the following scenarios would constitute negative environmental impacts. It is a negative impact to a water resource if 1) contamination occurs at a level high enough to impair or preclude a potential use; 2) existing contamination increases or new contaminants are introduced; 3) attenuation or remediation of an existing contaminated water resource is delayed, deferred, extended, or ended; 4) ash placement is used to justify an otherwise unacceptable project, and the project contaminates the water resource; or 5) a new pattern of contamination replaces an existing pattern of contamination.

The following cases are examples of sites that show one or more of these negative environmental impacts. The limitations of the interactive forum preclude an in-depth discussion of even one of these sites. Rather, a survey of data is provided to indicate the negative impacts that are occurring at each that gives rise to environmental concern. More data and a more extensive discussion of the data are available upon request. Each of these sites is known to

the respective state’s mining agencies and none, so far as is known, is considered by the agency to be an environmental concern that results from its ash placement program.

Negative impacts, and the types and designs of ash placement projects that resulted in those impacts, need to be identified and cataloged for at least three reasons. First, if they are recognized, they can be avoided in the future. Second, until the negative impacts to water resources are recognized, one cannot make rational policy decisions ranking the relative gains and risks; *e.g.*, removing physical hazards at a cost of contaminating water resources. Third, the “no damage” mantra of the mine placement program has been seized upon by generators of other waste streams as a model, methodology, and justification for comparable programs for those waste streams. In many cases, these expanded programs have even fewer controls and restrictions than the program for ash placement in mines.

Case Studies

Example 1. Universal Ash Site, Indiana (structural fill)

The Universal Ash Site is located northwest of Terre Haute, Indiana. It is a surface mine final-cut pit that has been reclaimed primarily with a structural fill of mostly conventional power plant wastes from a single facility and some coal gasification wastes. Until the pit-floor pond was completely displaced by the fill, its water level controlled by episodically pumping it into the immediately adjacent North Coal Creek, approximately 2 miles above its confluence with the Wabash River. Across North Coal Creek is an end-cut pond that has contact only with mine spoil. That pond serves as an observational control for the pond water quality. Ground water quality within the fill is measured by a well within the fill. Control ground water quality is available from a monitoring well within mine spoil and from a well in up gradient unmined strata adjacent to the structural fill. The data presented below are from IN Department of Natural Resources, Division of Reclamation.

Table 1 shows a comparison of the recent water qualities from the structural fill, the spoil, the up gradient unmined area, the spoil pond, and the fill pond. The data shown are the median concentrations from the most recent nine quarters of monitoring data, 2001 – 2003. **Table 1** is a snap-shot of evolving water qualities, particularly in the structural fill. For example, had the comparison been made from data representing the mid-1990s, the chloride and sodium levels would have been dozens-fold higher.

Table 1. Water Quality Comparisons, UAS						
		Fill Water	Spoil Water	Unmined Water	Spoil Pond	Fill Pond
TOC	mg/L	2.2	1.2	< 1	5.1	2.6
pH	s.u.	9.4	6.8	7.2	7.9	8.0
Alkalinity	mg/L	210	830	370	140	100
TDS	mg/L	2800	2400	413	310	850
Sulfate	mg/L	1700	1160	46	70	420
Chloride	mg/L	9.9	42	14	27	29
Calcium	mg/L	680	260	85	48	120
Magnesium	mg/L	9	150	39	22	31
Sodium	mg/L	37	308	19	12	41
Potassium	mg/L	20	7.6	1.3	3	6.8
Aluminum	ug/L	440	< 100	< 100	< 100	< 100
Arsenic	ug/L	197	< 5	< 5	1.1	40

Table 1. Water Quality Comparisons, UAS						
Barium	ug/L	30	< 20	240	30	30
Boron	mg/L	49.7	1.8	0.2	0.1	9.9
Manganese	ug/L	19	730	70	25	20
Molybdenum	ug/L	760	< 20	< 20	< 20	190

Figures 1a-f show a comparison of the time-series of concentrations of arsenic and boron from the fill well, the fill pond, and the spoil pond. Arsenic in the ground water in the fill is 20 times the MCL versus non-detect in both the spoil and the spoil pond. Boron in the fill water is about 20 times the concentration observed in the spoil well and 40 times the health-based level of 0.9 mg/L. The spoil pond has little boron; the concentration is comparable to the background ground water and below the health-based limit. The fill pond had concentrations well above the boron limit by 1991 that continued to climb until the pond was nearly filled.

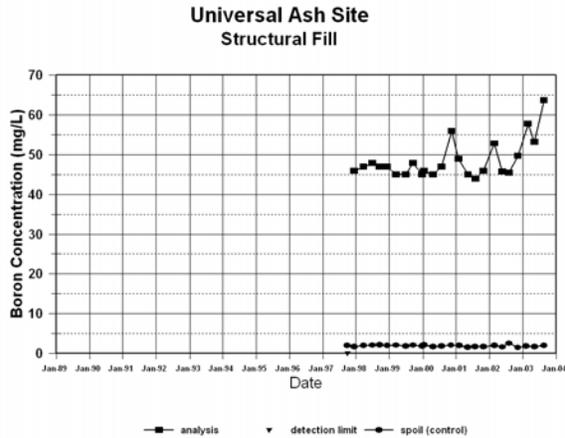


Figure 1a. Boron in fill

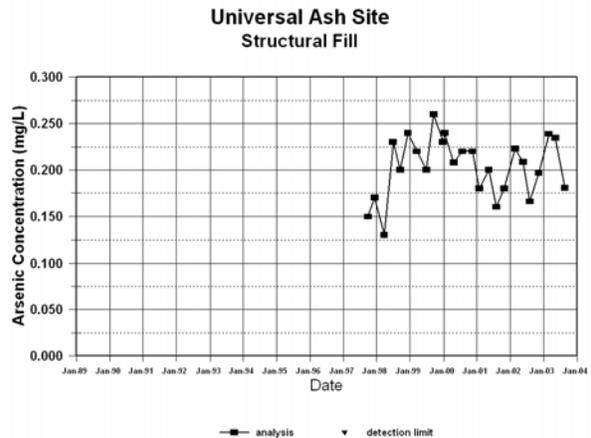


Figure 1b. Arsenic in fill

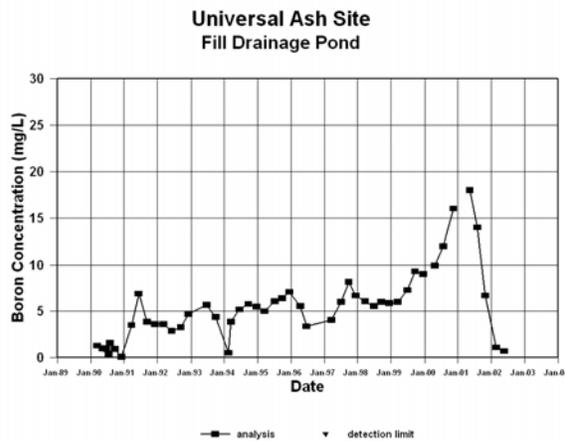


Figure 1c. Boron in fill pond

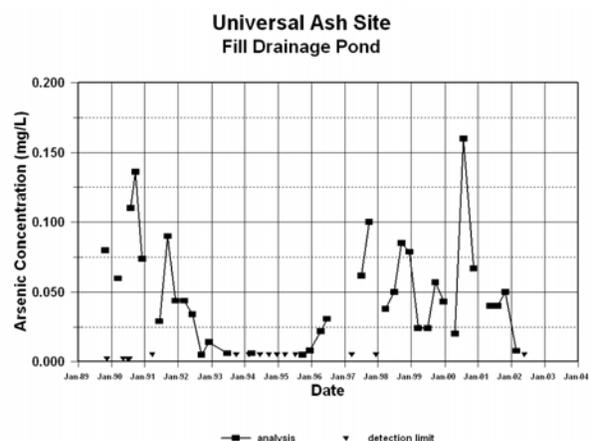


Figure 1d. Arsenic in fill pond

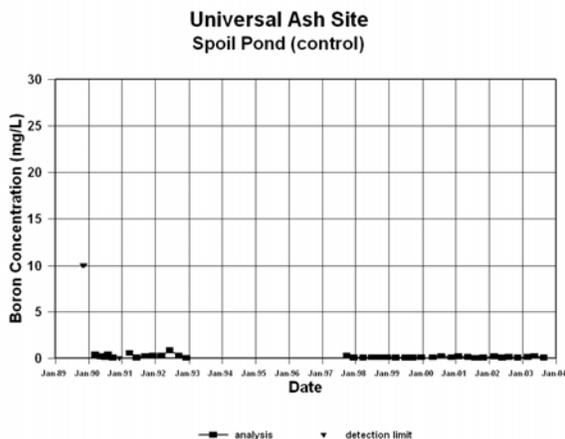


Figure 1e. Boron in spoil pond

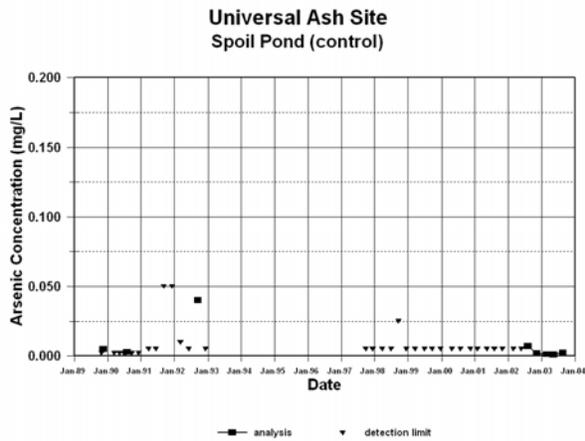


Figure 1f. Arsenic in spoil pond

Figures 2a-f show the time-series quality of the seepage from the northeast corner of the fill that drains immediately into North Coal Creek. Prior to about 2002, this seepage represented fill pond water. Since then, it represents the discharge of ground water from the fill.

Example 2. Prides Creek Mine, Indiana (structural fill)

The Prides Creek Mine is located just south of Petersburg, Indiana. Since 1995, reclamation under its SMCRA permit allows conventional power plant wastes from multiple facilities to be placed as structural fill in 10-ft layers in the spoil and in monofills. The data presented below are from IN Department of Natural Resources, Division of Reclamation.

The ash monitoring system for ground water consists of 7 monitoring wells; five wells around the perimeter of the mine, one well in strata below the mine, and one well in the spoil. At present, there is no monitoring point for ground water within the structural fills, although the permit currently calls for a monitoring well in the first monofill completed. The spoil well is the only well located where it may see impacts to date from the structural fill.

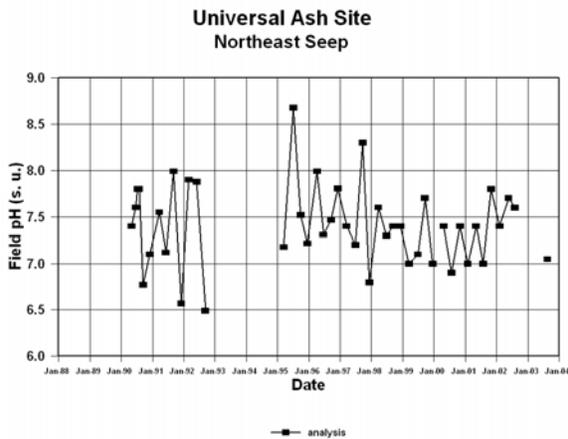


Figure 2a. pH of fill seep

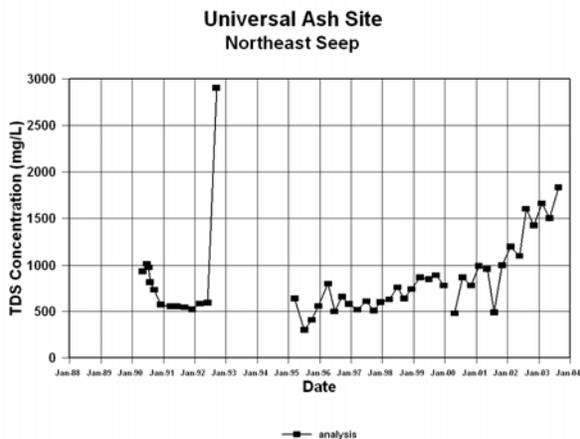


Figure 2b. TDS of fill seep

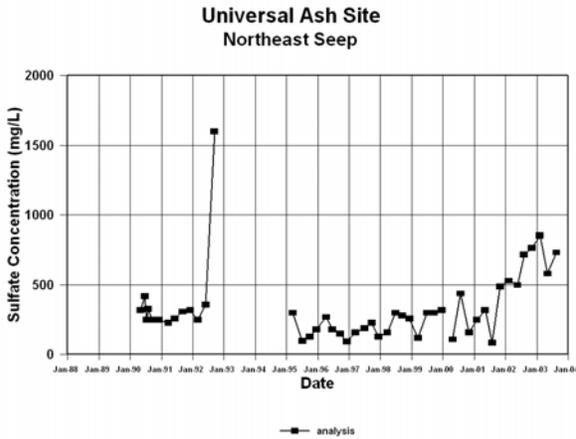


Figure 2c. Sulfate of fill seep

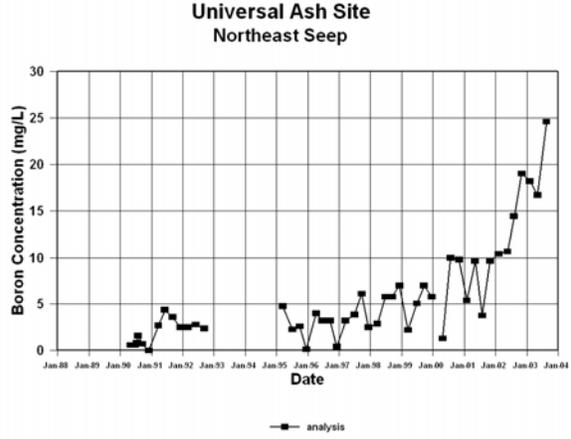


Figure 2d. Boron of fill seep

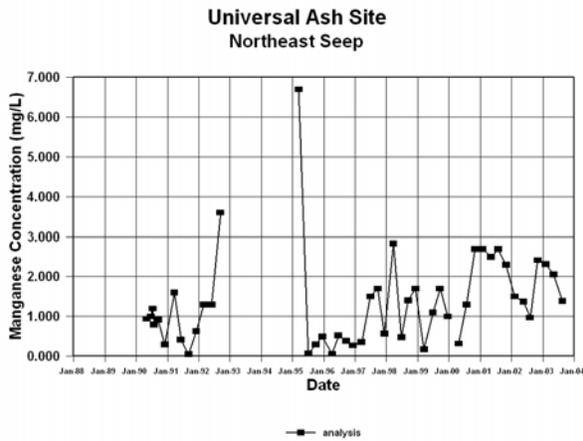


Figure 2e. Manganese of fill seep

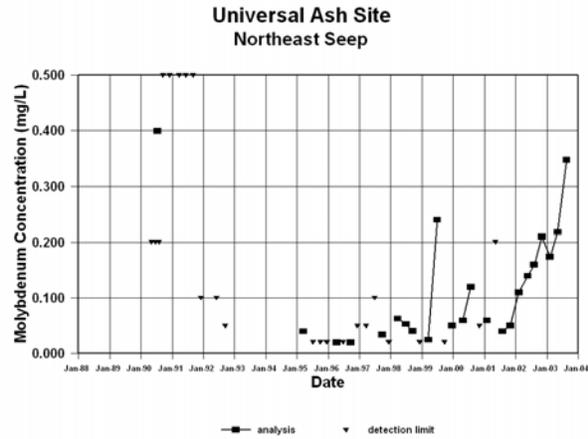


Figure 2f. Molybdenum of fill seep

Figures 3a-f show the time-series of concentrations of major constituents in ground water at the spoil well. If the spoil well at Prides Creek behaves in normal fashion, one would expect to see major contaminants like sulfate and TDS decline in response to the rising water. Initially, that is what is observed. However, that trend is interrupted and the water quality changes and deteriorates. What had been a calcium sulfate and carbonate water is replaced by a sodium sulfate and hydroxide water. The pH of the ground water climbs to above 9.5 s.u. and boron, a common marker of combustion waste leachates, more than doubles.

The water quality observed at the spoil well continues to evolve after the initial change.

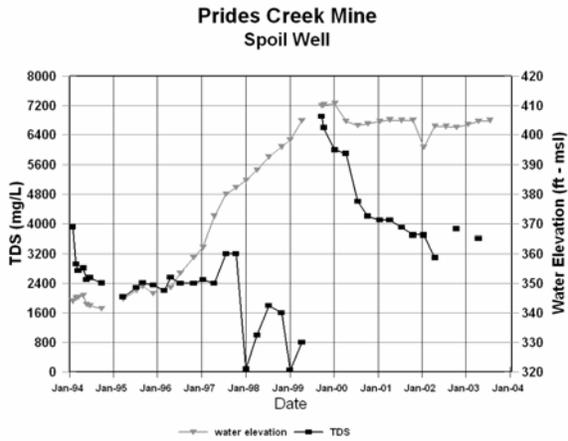


Figure 3a. TDS in spoil water

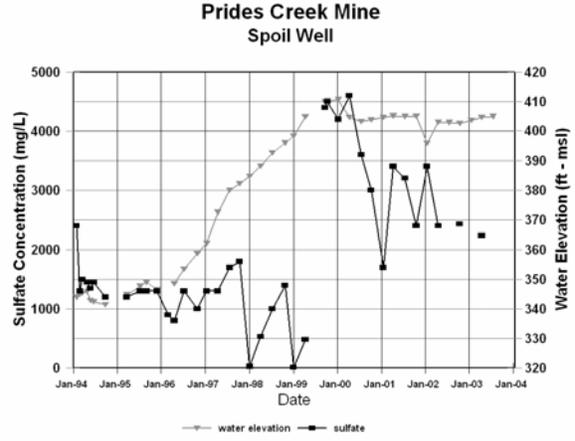


Figure 3b. Sulfate in spoil water

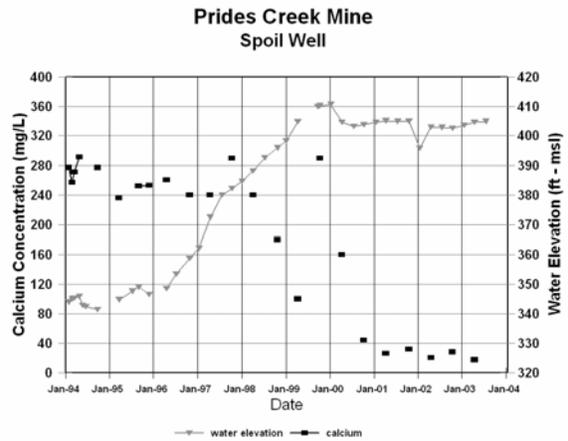


Figure 3c. Calcium in spoil water

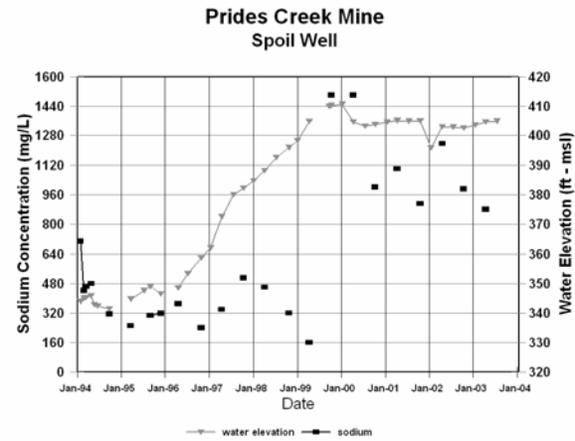


Figure 3d. Sodium in spoil water

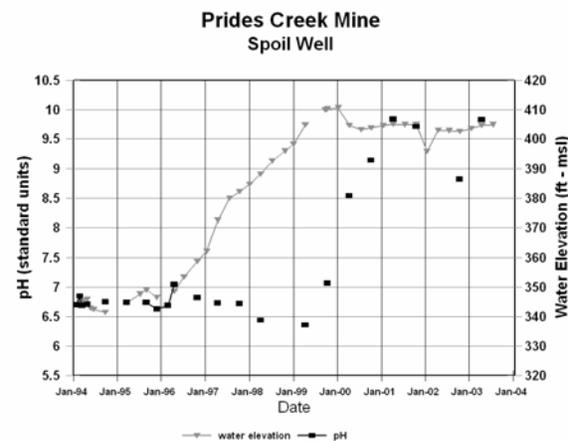


Figure 3e. pH in spoil water

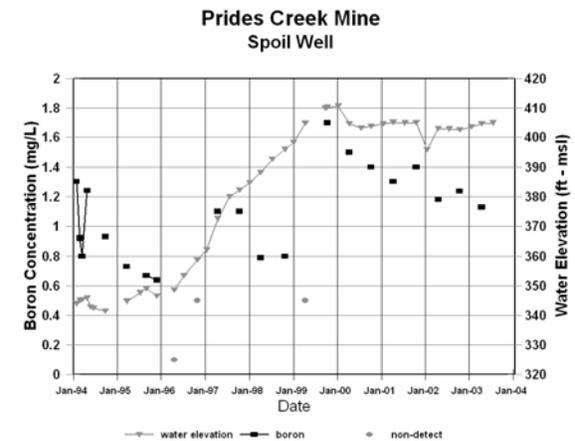


Figure 3f. Boron in spoil water

Figures 4a-e show the time-series concentrations of some minor constituents. Zinc initially climbs, but then drops

significantly in the most recent few analyses. This drop in zinc is concurrent with the sudden appearance of sulfide in the ground water. Arsenic, selenium, and molybdenum also appear for the first time at significant concentrations when the sulfide appears. Two important concepts are demonstrated in these data. First, metals can be mobile in high pH water and can first appear after a rise in pH. Second, depending upon the evolution of the fill with time, changes in flow direction of leachate plumes, and/or changing sources of fill, water quality can change dramatically through time.

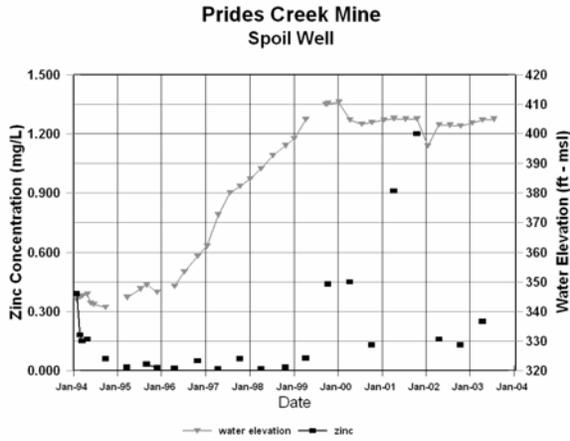


Figure 4a. Zinc in spoil water

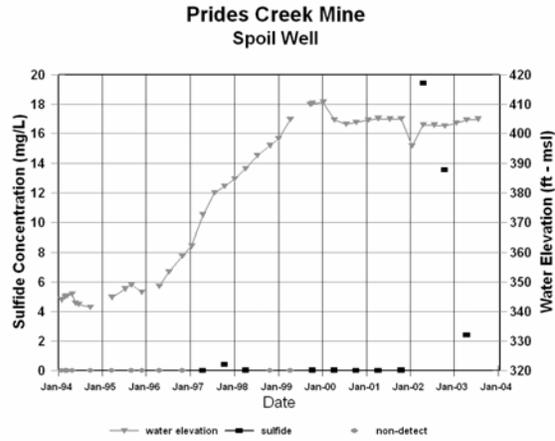


Figure 4b. Sulfide in spoil water

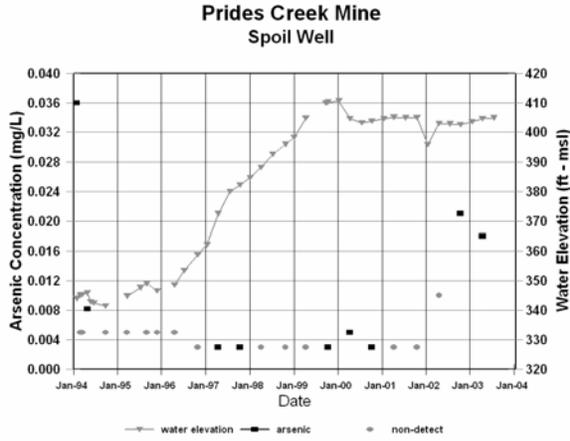


Figure 4c. Arsenic in spoil water

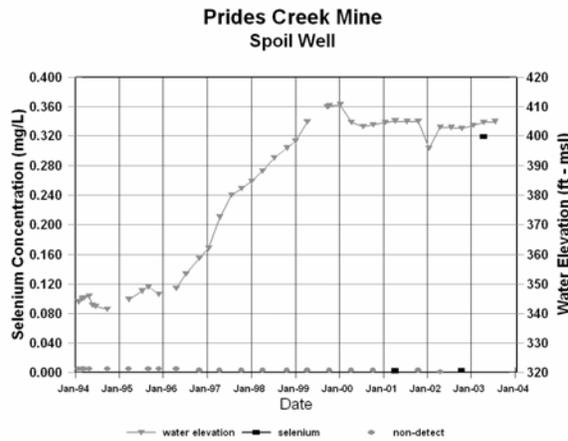


Figure 4d. Selenium in spoil water

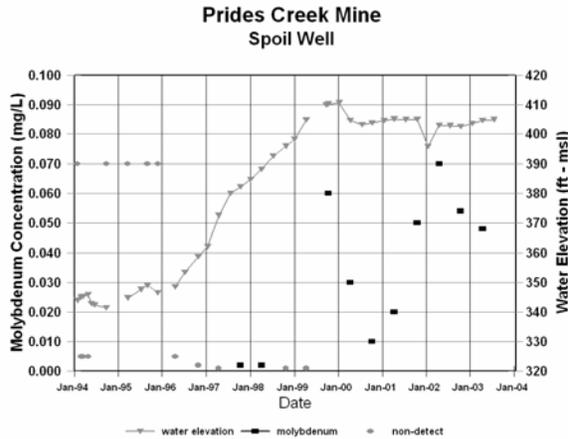


Figure 4e. Molybdenum in spoil water

Example 3. Kedron Mine Pilot Project, West Virginia (permeability barriers, containment, alkaline addition)

The mine plan proposed three standard methods of placement. The mine floor was to be covered with a compacted layer of ash that would set up to form a permeability barrier and alkaline barrier between infiltrating ground water and an acid-producing mine floor. A permeability barrier back wall to the excavation would be constructed of compacted ash to reduce water flow onto the mine, partially isolating the spoil and any acid forming rock there. The ash would be mixed with the spoil in the stack to provide alkaline addition in the event of acid generation.

A field pilot project was performed to obtain field-scale observations of materials placed using mining equipment. This offers one of the few opportunities to observe conditions within the placed materials for FBC ashes for three typical placement practices. The data presented are those reported to the WV Department of Environmental Protection for the pilot project. They are discussed in more detail in a draft report produced for Anker Energy and the West Virginia Highlands Conservancy (Norris, 2002).

Table 2 compares the results of the observations from months 8 through 13 after mining and placement. Trends in the data are generally absent, but where present it is not possible to determine whether they are characteristic of site evolution or seasonal variation. The data columns represent range and median values from the background, spoil stack, and the floor layer of the test project.

		Background			Spoil stack			Floor layer		
Param.	Unit	Low	High	Med.	Low	High	Med.	Low	High	Med.
pH	s.u.	5.2	6.7	5.9	6.2	7.4	6.4	10.6	12.3	12.2
Acidity	mg/L	< 5	12	6	<	576	< 5	< 5	20	< 5
Alka.	mg/L	4	140	7	20	148	73	68	740	460
TDS	mg/L	28	172	62	1116	3488	2876	5796	6440	6152
SO4=	mg/L	20	100	59	60	620	405	400	625	483
Cl-	mg/L	< 3	210	< 3	24	200	42	39	1500	1100
Boron	mg/L	<0.02	0.204	0.071	<0.02	0.235	0.072	<0.02	0.035	<0.02
Fe	mg/L	3.1	22.4	5.0	4.9	25.2	12.4	0.2	1.2	0.3

Mn	mg/L	<0.03	0.18	0.04	3.1	18.8	8.1	<0.03	0.06	<0.03
Al	mg/L	0.45	4.96	1.41	1.46	7.36	3.97	0.17	0.56	0.37
As	mg/L	<22.9	<22.9	<22.9	<22.9	42	<22.9	<22.9	<22.9	<22.9
Pb	mg/L	<22.5	22.7	<22.5	90	<22.5	<22.5	<22.5	<22.5	<22.5
Se	mg/L	<24.6	75	36.8	<24.6	437	81	<24.6	463	189

In addition to the monitoring of ground water, seeps and stream water quality were monitored. The monitoring of seeps includes some baseline data collected prior to the pilot project. The major seep from the pilot area re-established itself four months after mining. The pre-mining acidic drainage (pH of 3.5 +/-) appeared initially after 3 months and was followed in months 5 and 6 by an initial flush with pH values between 11.5 and 12.5 s.u. Thereafter, the pH has remained between 7.8 and 6.4 s.u. Thus, the discharge of toxic pH was over prior to the data collected from the ground water in the monitoring wells. The flush of caustic water from the test area suggests why these materials have not been effective at preventing acid mine drainage when they are used as alkaline addition. The alkalinity appears highly soluble and washes from the FBC ash on contact with water and is not retained as mineral alkalinity that will neutralize acid as it forms.

The highest levels of major constituents (sulfate, TDS, iron, aluminum, and manganese) other than pH occurred with the pH flush or immediately thereafter. All of the major constituents except iron were significantly higher after the flush than prior to the mining, from twice to more than 20-times the pre-mining concentrations. Minor constituents were only measured once after the flush period, so little can be concluded. Arsenic, selenium and lead were not detected after the flush, but at reporting limits below those of health or environmental concern.

Example 4. McDermott Mine, Pennsylvania (permeability barriers, containment, alkaline addition)

The McDermott operation is an example of a surface mine that would not be allowed without the incorporation of a plan for alkaline addition; the overburden does not provide enough alkalinity to neutralize the potential acid production (Kania, in PADEP, undated, Chapter 18, Site 2). Mining is allowed only if an alkaline addition, traditionally limestone, is brought in to offset the natural potential for acid generation. Against an operator preference for limestone, PADEP insisted on using FBC ash as that alkaline addition.

The McDermott operation created negative impacts in two ways. First, a spring used as a domestic water supply was contaminated when the flow of mine-impacted water was redirected by the placement of the FBC ash as the permeability barrier (Mukherjee, 2002). Second, the FBC ash as an alkaline addition was ineffectual in preventing acid mine drainage (Brady, 2002). As stated in that memorandum, "The degradation in water quality at MD5, and in particular, MD3 is due to mining on the McDermott site by Laurel Land and Development."

The failure of the McDermott operation to prevent acid drainage is attributed by PADEP to ineffective handling of the FBC ash (Brady, 2002, citing Schueck, et al., 2001) or insufficient addition. The detailed observations from the Kedron test project and other testing of FBC by Patriot Mining suggest that the inability to preventing acid drainage is related instead to the flushing of alkalinity from the FBC ash, leaving little left to neutralize acid generation when it occurred. Further, depending upon the oxidation state of metals and sulfur, even alkaline and neutral power plant wastes can generate acid drainage (Radian, 1989). Once flushed of readily soluble alkalinity, the same may hold true for FBC ashes, contributing to acid generation rather than preventing it.

In any case, the representations by some that no portion of the contaminants of acid generation originate from the placed FBC materials is without merit. Research by Skousen and Bhumbra (1998) demonstrates that once alkalinity is exhausted, the metals in the ash mobilize as well as any metals temporarily bound by the initial alkalinity.

No one disputes that the McDermott operation caused negative impacts. Proponents of placement in mines dismiss McDermott and others like it. This is inappropriate for a number of reasons. The negative impacts of the McDermott operation would not have occurred had there been no ash placement. Either the mine would not have

been approved, or the operator would have used traditional alkaline addition, likely effectively. McDermott is not alone. The placement plan that was used at McDermott is PADEP boilerplate, identical to the plan used at other sites, at least one of which has caused similar impacts (Norris, 2003).

Example 5. Albright Mine, West Virginia (permeability barriers, containment, alkaline addition, AMD treatment)

The Albright operation plan proposed to place FBC ash as an alkaline addition, mixing it into the overburden. The data presented below are from WV Department of Environmental Protection. Up to four million tons were proposed for placement. In addition to the mining operations, there was an area of adjacent activity by the state abandoned mined lands (AML) program. The state was extinguishing and reclaiming an abandoned, burning refuse pile, the result of an unsuccessful “controlled burnout” by U. S. Bureau of Mines. The operator proposed in the permit to make additional FBC ash available to the AML operations and, after AML reclamation was finished, to incorporate the reclaimed area into the mining permit as an “ash disposal area.”

A widely-used recreational river, the Cheat River, bounds the east side of the site. Prior to mining, there was visual acid drainage entering the river from the burning AML refuse pile. Seeps from the refuse had low pH and were laded with iron, aluminum and magnesium. Ground water between the refuse and the site was similarly degraded with acid mine drainage. Water quality in the river below the site was measurably degraded relative to above the site with respect to acid drainage parameters. It is not clear how significant the contribution from the refuse pile was, however. Loading calculations from baseline data account for less than 1% of the degradation measured in the river. The river was also degraded by two streams between the sample points, each of which was also impacted by pre-law acid mine drainage. The stream entering from the opposite side of the river alone showed loading equivalent to between 200 and 300 percent of the load increases in the river. As visually dramatic as the visible seepage and the burning refuse may have been to users of the river, the data do not suggest the site contributed significantly to the water quality problems relative to other sources.

Direct comparisons for most contaminants of interest are not available for pre- and post-placement scenarios. The wells that existed pre-placement were all removed as part of operations and there is little or no post-placement data available from them. The well that was installed as part of operations has no pre-placement data to compare with post-placement data. The post-placement surface water discharges are from outlets at sediment ponds that have no pre-operational data, and the seeps that were measured before operations began were either mined through or not measured after operations. Anecdotally, the Albright operations are reported to have greatly improved the conditions at the site, but there are no data in the reviewed files that allow a quantification of that improvement or a means of determining whether the improvements are the result of FBC ash placement or traditional reclamation techniques.

Ground water quality degradation can be documented in the new well that was installed down dip of the mining and placement operations.

Figures 5a-f are a composite of the time-series of concentrations for some parameters. The pH shows little evidence that the alkalinity of the FBC ash was retained as neutralization capacity for the ash-spoil mix; over the period of monitoring record, the pH at the monitoring well has decreased from neutral to slightly acidic. Manganese and sulfate peaked and have irregularly declined beginning in 1998, perhaps suggesting lower rates of pyrite oxidation. Concurrent with the sulfate decline, a variety of major and minor metals begin to appear at significant concentrations, documenting degradation of the ground water.

It is noted that the concentrations for arsenic and selenium episodically exceed those that are damaging to human or aquatic health. These follow similar peaking for lead and cadmium earlier in time. As part of ash characterization prior to placement, the operator went beyond state requirements and performed column leachate tests of the spoil and of the ash, followed by extraction tests of the column materials. In the extraction tests that followed column leaching, arsenic was 4 times as soluble from the ash as from the spoil. Selenium and molybdenum were more than three times as soluble from the ash as from the spoil. Thus, while the metals that have risen in the ground water certainly originate from both ash and spoil, the laboratory data indicate that the ash is likely the dominant source.

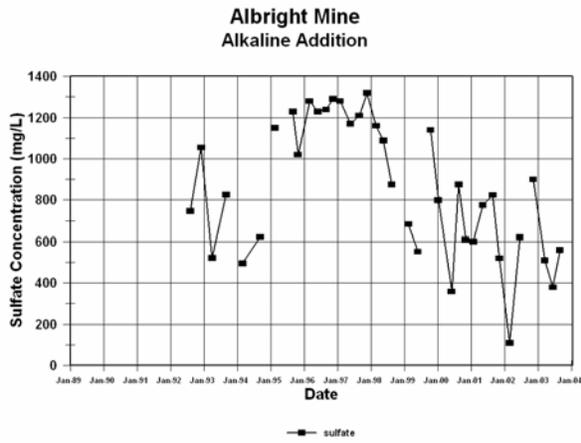


Figure 5a. Sulfate down gradient

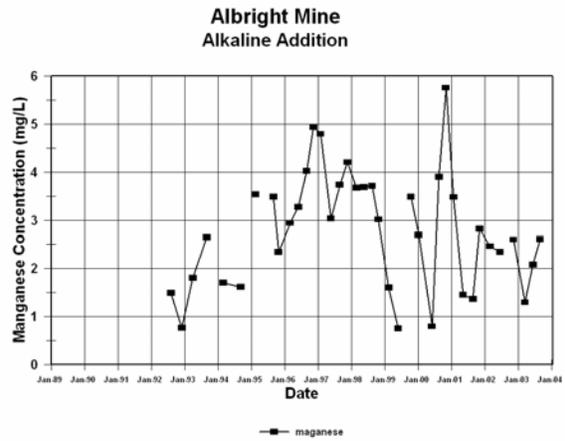


Figure 5b. Manganese down gradient

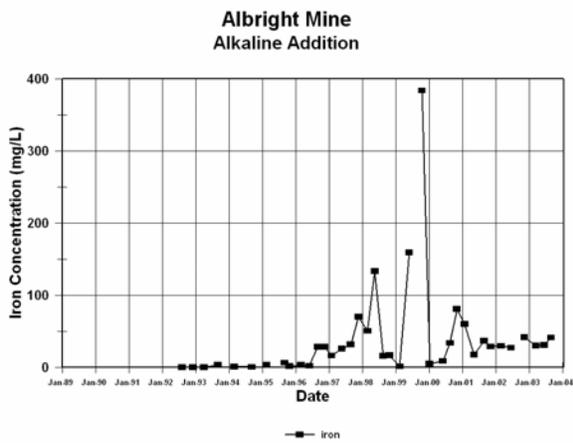


Figure 5c. Iron down gradient

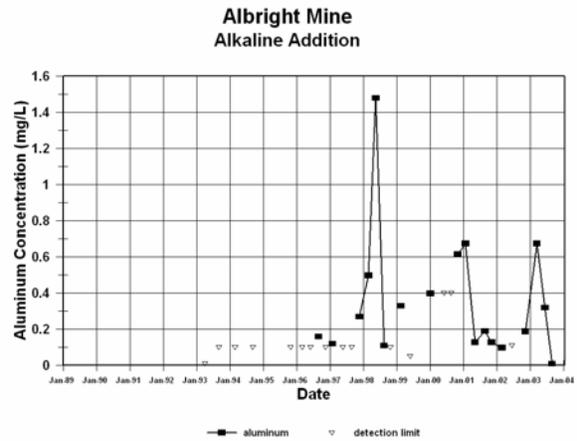


Figure 5d. Aluminum down gradient

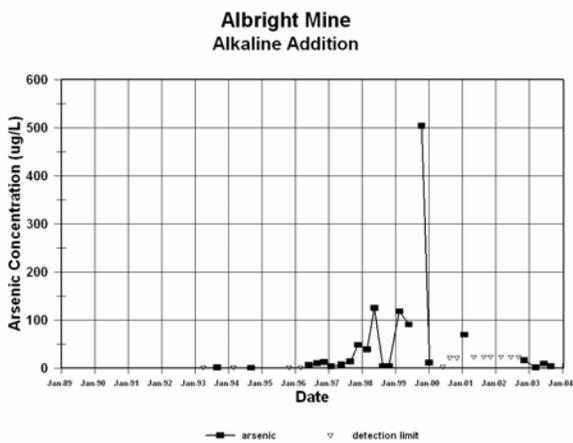


Figure 5e. Arsenic down gradient

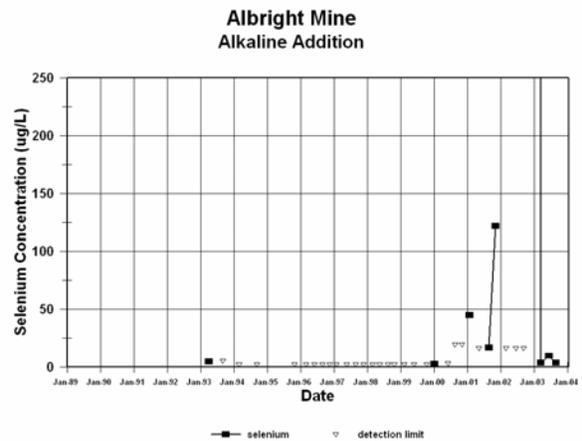


Figure 5f. Selenium down gradient

The NPDES permit for the Albright mine doesn't limit discharges of the trace metals and the SMCRA permit doesn't limit trace metals in the ground water. State regulators are aware of the data. It would appear that this degradation is acceptable within the state program.

Example 6. San Juan Mine, New Mexico (structural fill)

Power plant wastes from a mine-mouth conventional power plant are currently placed on the adjacent mine as structural fill. The mine was originally a surface mine but has moved underground. The mine and the generating station are separated by an arroyo. That arroyo, prior to the mine-generation-placement operations was ephemeral, but now is perennial at and below the generating station. The data presented below are from NM Department of Energy, Minerals and Natural Resources.

The separating arroyo is the original channel of the Westwater Arroyo and now conveys the surface flow of both the Westwater and the upper Shumway drainages. The Shumway Arroyo exists both above and below the mine permit area, but within the permit area has been mined through and/or filled with mine spoil and power plant wastes. The diverted Shumway surface flows and Westwater surface and alluvial flows rejoin the Shumway channel below the mined area at the historic confluence, and the lower Shumway conveys the combined flows.

The lower Shumway is perennial and for generations provided water for domestic and livestock uses for the community of Waterflow, NM. But for its degraded quality, it would still do so. The receiving river is the San Juan River. Under international treaty, the San Juan River has a zero degradation standard for dissolved solids.

Long term monitoring exists in the Shumway arroyo alluvium adjacent to the mine above its diversion to the Westwater arroyo and in the Shumway arroyo alluvium below the mine and above the confluence with the Westwater arroyo. Monitoring in the Westwater arroyo alluvium occurs adjacent to the generating station and in the Westwater arroyo alluvium above the confluence with the lower Shumway arroyo.

Severe degradation of the combined-arroyo surface and ground waters began when the mine-generation-placement operations began. This contamination was attributed solely to deficiencies of the generating station's waste management systems. Modifications to the management systems of the generating station reduced discharges of contaminated water into the combined arroyo drainage and placement of the wastes continued on the mined areas. Water quality within the combined arroyo drainage, although still heavily degraded, has steadily improved since those changes. The water quality in the natural arroyo alluvium draining the mine fill areas continues to degrade and is now at levels never previously seen. The monitoring of the alluvium above the diversion shows concentrations that are reasonably consistent with what is observed in areas without potential impacts, until mining and fill placement moved past the area. Subsequently water quality there has severely degraded also.

Figures 6a-f depict these patterns for some of the major constituents. The data showing degradation of the waters that flow around and from the mine were obtained from state regulatory files. The mining company denies that any problem exists. Under state regulation, no corrective response has been required, presumably because the state is content to allow it to continue to worsen. Under the existing regulatory structure, the federal regulators have deferred to the state programs. Civil action precipitated modifications to the generating station that are producing improvements to part of the system. Without citizen enforcement, there is no indication the existing regulatory system will generate any relief from worsening conditions from the placement areas for the downstream citizens or for the San Juan River.

Example 7. Ernest Mine, Pennsylvania (structural fill, alkaline addition)

The Ernest Mine is a remining operation of an existing refuse disposal area with back haul and placement of the combustion materials as structural fill. To a lesser degree, the placed material provides alkaline addition to refuse

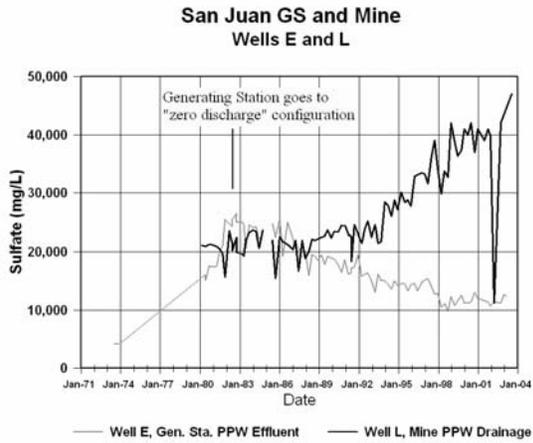


Figure 6a. Sulfate down gradient of mine

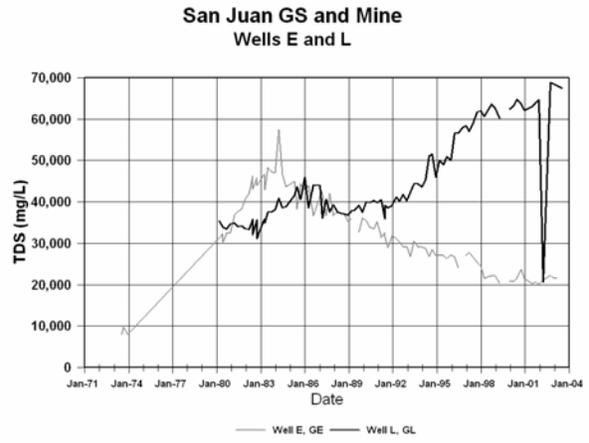


Figure 6b. TDS down gradient of mine

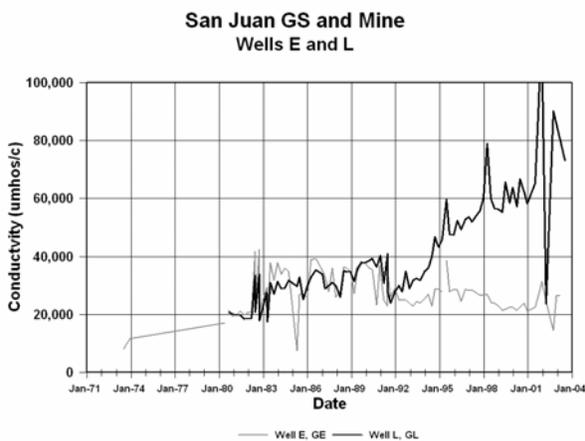


Figure 6c. Cond. down gradient of mine

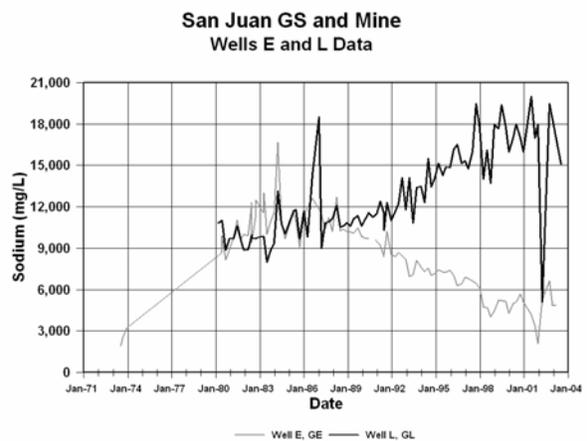


Figure 6d. Sodium down gradient of mine

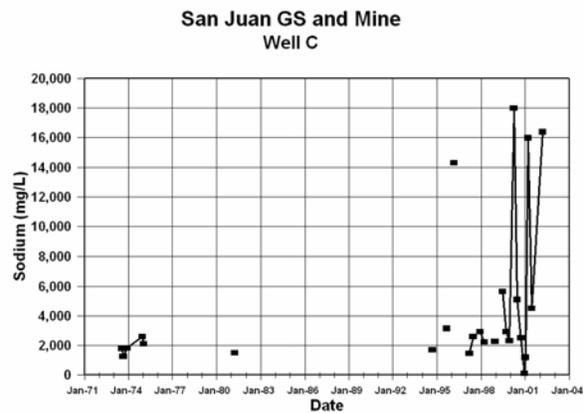


Figure 6e. Sodium side gradient of mine

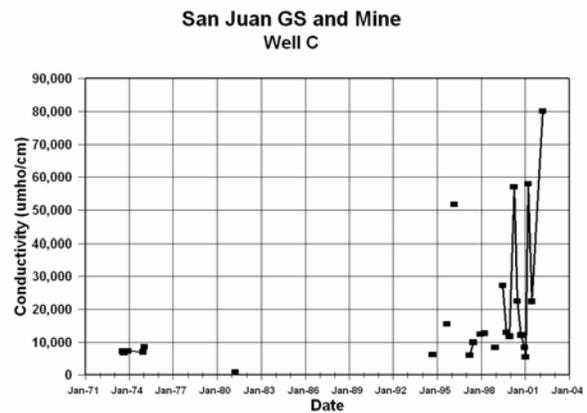


Figure 6f. Cond. side gradient of mine

that is unsuitable for burning. As characterized in baseline data, acid drainage from the site prior to operations averaged about 12 gpm. Upstream of the site, the receiving stream is used as a public water supply. Significant impacts to the receiving stream that are attributable to the pre-operational discharge are not apparent in the baseline data. The lack of such impacts is a reflection of the low rate of discharge from the site relative to the size of the receiving stream.

Independent third-party review of monitoring data from the Ernest mine identified degradation in the ground water quality at the designated ash monitoring points. Based in part on those observations, PADEP initiated its own study of the site, and confirmed the apparent degradation (Conrad, et al., 2002).

The PADEP memorandum went on to conclude that, although the specific cause of the degradation at the designated ash monitoring locations could not be determined, the source of the contaminants was not the ash that had been placed. Four principle lines of reasoning were advanced to support this conclusion. Impacts to surface water bodies which received run-off from placement areas were observed (non-permit, special monitoring) and those contamination patterns did not duplicate those at the ash monitoring locations. Quality control problems that have apparently persisted for years cast doubts on the reliability of the data from the ash monitoring locations. Based upon what is known about the mining history and geology of the site, and the location of fill placement to date, any leachates that are being produced from the fill are likely migrating away from the designated ash monitoring locations through deep mines, not toward them. If there is some shallow migration toward the designated ash monitoring locations, the locations are too remote (half-mile away) from the fill have yet seen any impacts.

The final two conclusions are particularly remarkable. The state apparently dismisses the degradation at the ash monitoring locations, and criticizing those who have commented on it, based upon a monitoring system that the state represents as incapable of identifying impacts from the fill that may be occurring. Further, it appears that the state has taken no actions under the permit to correct the long-standing quality control issues it believes exist, to require monitoring wells that are capable of tracking impacts from the fill in ground water, establish the cause of the observed degradation, or implement remedial measures to reverse the continuing degradation that is observed. In the final analysis, this operation is an example of an ash placement project that was justified on the premise that it would remediate contamination from an old site and, instead, the project is resulting in conditions that are worsening, not improving the state's water resources. And, according to PADEP, any additional impacts attributable to the fill itself are not monitored.

Example 8. Bark Camp, north side fill area. (Structural fill, alkaline amendment)

The Bark Camp experimental site is an area where a series of technologies and materials have been used to evaluate their ability to address various kinds and settings of AML problems. The greatest portion of the area has been used to study the effects of using sediments dredged from East Coast harbors mixed and/or stabilized with various mixes of combustion products as a structural fill to reclaim features like abandoned high walls. Bark Camp Run was also the site of several experimental artificial wetlands projects. The data presented below are from the PA Department of Environmental Protection.

In one area, on the north/east side of Bark Camp run upstream of the deep mine portals, the structural fill was coal combustion ash without other combustion products or dredged material. (Subsequent to the placement of the ash, there was also the placement of paper sludge, an organic waste.) The water discharging from this fill area has been monitored at SW-8 since November, 1997. The discharge is contaminated with a number of constituents, including boron and arsenic. Both contaminants are declining with time, but at the existing decline rates, it will be many years before the boron or arsenic will meet water quality standards.

Figures 7a-c show both of these contaminants and the pH of the discharge water.

It is noted that the high contamination levels of arsenic occur in a discharge water that is alkaline, not acidic. A commonly expressed misconception is that metals contamination is associated with acid drainage and that raising the pH is equivalent to eliminating dangers associated with metals mobility. The monitoring at point SW-8 demonstrates that such a simplistic approach can be badly in error. In this water, arsenic is clearly mobile in moderately high pH water. At another Bark Camp monitoring point, SW-32, drainage from a different fill material (not coal combustion) has a caustic pH of more than 11 s.u. and shows toxic levels of not only arsenic, but copper,

lead, manganese, molybdenum, nickel, and selenium. It is clear that lowering metals mobility is far more complex than simply raising the pH.

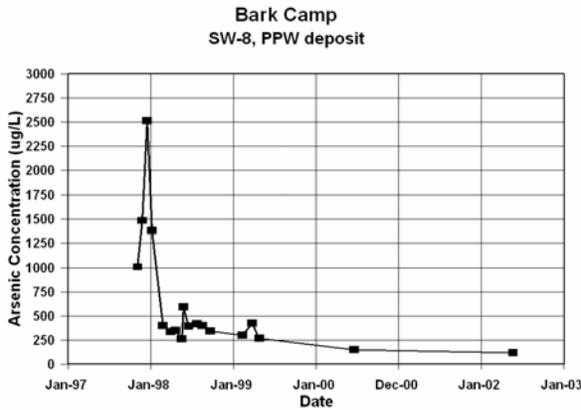


Figure 7a. Arsenic at SW-8, Bark Camp

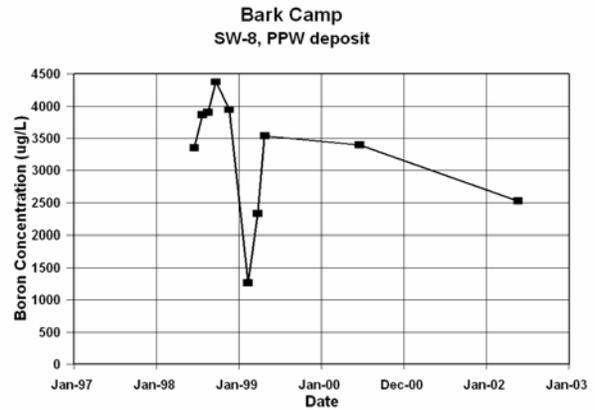


Figure 7b. Boron at SW-8, Bark Camp

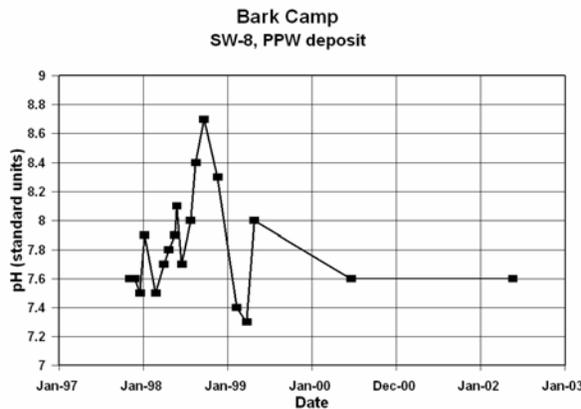


Figure 7c. pH at SW-8, Bark Camp

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IMPACTS OF AQUATIC DISPOSAL OF COAL COMBUSTION BY-PRODUCTS ON VERTEBRATES ON THE SAVANNAH RIVER SITE

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Abstract

Over the last 12 years, studies conducted at the D-area power facility have provided significant insight into the hazards posed by disposal of coal combustion byproducts to resident vertebrate wildlife. The 70 MW power plant, located on the Savannah River Site near Aiken, SC, discharges its wastes (fly and bottom ash) into a series of surface impoundments (i.e., settling basins) after it is mixed with water from the nearby Savannah River. Surface waters from the basins are eventually discharged into Beaver Dam Creek, a tributary of the river. The aquatic habitat associated with the disposal system attracts diverse wildlife that use the site for foraging, overwintering, and reproductive purposes. Amphibian biodiversity in D-area is particularly high; a total of 22 species have been documented at the site. Most of these species exhibit complex lifecycles, where adults migrate from adjacent terrestrial habitats to deposit their eggs in the settling basins. Amphibian embryos and larvae are subsequently exposed to high concentrations of trace elements (e.g., Se, As, Cd) during critical windows of development. Because developing amphibians are extremely sensitive to environmental changes, a variety of physiological, behavioral, and morphological abnormalities have emerged. In some cases, observed effects are profound enough to reduce survival, ultimately decreasing the number of individuals capable of metamorphosing (and thus having any future reproductive potential). Here, we briefly review our previous studies on amphibians in D-area, offer our perspective on population-level threats posed by these disposal procedures, and provide our vision of where future studies should be directed.

Background

The largest portion of electric utility capability in the U.S. remains fueled by coal (USDOE, 2000), consuming over 1 billion short tons in 1997 (USDOE, 1999). Global coal consumption is projected to increase by 27 % during the period from 1990 to 2010 (USDOE, 1995). This reliance on coal as a fuel source results in production of large amounts of coal combustion wastes (CCW), about 60 % being fly ash, a fine particulate material collected by emission control devices that is enriched in numerous trace elements, including As, Cd, Cr, Hg, and Se (Carlson and Adriano, 1993). Fly ash production has increased in the U.S. from approximately 26 million tons in 1970, to nearly 63 million tons in 1998 (EPRI, 1997; USEPA, 1997; ACCA, 1998). Approximately one third of the ash annually produced is water-slurried and pumped into open, aquatic settling basins, prior to discharge of surface waters into local water bodies (EPRI, 1997; USEPA, 1997).

A result of CCW disposal into aquatic systems is chronic exposure of resident wildlife to mixtures of trace elements. Chronic exposures result in uptake and accumulation of potentially toxic levels of contaminants (Cherry and Guthrie, 1977; Evans and Giesy, 1978; Guthrie and Cherry, 1979; Sorensen et al., 1984; Lemly, 1985, 1993; Brieger et al., 1992; Carlson and Adriano, 1993; Rowe, 1998; Rowe et al., 1996, 1998a; Hopkins et al., 1998, 1999b). In many instances, body burdens of trace elements greatly exceed established criteria for animal health (Lemly, 1993; Hopkins et al., 1999b), resulting in sublethal or lethal effects. Studies have identified sublethal and lethal effects on amphibians, reptiles, fish, and crustaceans in several CCW contaminated systems around the U.S. (Cumbie and Van Horn, 1978; Lemly 1985, 1993; Gillespie and Baumann, 1986; King, 1988; Rowe et al., 1996, 1998a and b; Hopkins et al., 1997, 1998, 1999a and b, 2000a and b; Raimondo et al., 1998; Rowe, 1998).

Study Site

Over the last 12 years, our research has focused on the D-Area Electric and Steam Generating Facility at the Savannah River Site, South Carolina. The Savannah River Site is a 310-mi² reservation on the Upper Coastal Plain of South Carolina. The U.S. government established the Savannah River Site (SRS) in 1950 for the production and processing of nuclear materials for national defense requirements. The Department of Energy manages SRS as a controlled area with limited public access. The Savannah River forms the site's southwestern boundary for 27 miles on the South Carolina-Georgia border. Site facilities occupy only about 10 percent of the SRS. The remainder of the area includes a variety of terrestrial and aquatic habitats typical of the region, including bottomland hardwood forests, cypress-tupelo swamps, oak-hickory forests, sandhills-longleaf pine uplands, pine plantations, blackwater streams, reservoirs, and Carolina bays and other seasonal wetlands. Within the site boundaries, thirty DOE Research Set-Aside Areas, totaling about 14,000 acres, have been set aside for ecological research. The entire site is designated as a National Environmental Research Park (NERP) and is used by ecology, forestry, and archaeology groups to study the impacts of human activities on the environment.

Disposal of coal combustion wastes at the 70 MW D-Area site is accomplished in the same manner as is used by many other power stations in the U.S. (EPRI, 1997), relying on passive removal of particulate material from the water-slurried waste stream as it passes sequentially through two settling basins and a drainage swamp. The plant burns bituminous coal from Middlesboro, TN. All bottom and fly ash is deposited in the basins and no other wastes are co-managed at the site. Water retention time in the entire basin system is around 60 days and water discharge from the site is ~5 million gallons/day. Sediments, water, and biota in the disposal system have elevated concentrations of trace elements and heavy metals derived from the ash (Cherry and Guthrie, 1977; Evans and Giesy, 1978; Guthrie and Cherry, 1979; Alberts et al., 1985; Sandhu et al., 1993; McCloskey and Newman, 1995; Rowe, 1998; Rowe et al., 1996, 1998a; Hopkins et al., 1998, 1999b, 2000a).

Amphibians as Models

Amphibians have become increasingly important as indicators of environmental quality because of life history traits which may make them particularly susceptible to environmental contaminants (see Dunson et al., 1992). Most amphibians have complex life cycles in which early (e.g., embryo and larvae) and later life stages (e.g., juveniles and adults) are separated by a morphological, physiological, and/or behavioral metamorphosis event. During the reproductive season, adults of many species migrate to ponds where they breed. Their embryos and larvae subsequently develop in the aquatic environment where they are sensitive to the quality of the aquatic breeding site. Thus, breeding sites of poor quality can produce fewer terrestrial recruits than habitats of higher quality, ultimately having implications for local population dynamics.

The D-area disposal system attracts numerous wildlife species (including amphibians, reptiles, fish, birds, and mammals), but amphibian biodiversity is particularly high. Twenty-two species of amphibians occur in D-area, most of which use the site for reproductive purposes. Research on amphibians in D-area indicates that exposure to the wastes results in accumulation of trace elements and a variety of adverse effects. In general, the observed effects can be divided into three categories: Direct Effects, Indirect Effects, and Maternal Effects. A very brief synopsis of each of these effects is presented below.

Direct Effects

The most obvious and commonly examined effects of contaminants on organisms relate to direct toxicity. In D-area, field surveys and field experiments have revealed a variety of adverse effects in amphibians that emerge from direct exposure to conditions in the site. Most work has focused on embryonic and larval life stages that develop in the contaminated settling basins.

Similar to previous work on fish, our research on amphibians has revealed developmental abnormalities associated with exposure to CCW. The two most conspicuous abnormalities relate to the oral and spinal regions. Oral abnormalities include degeneration of tooth rows and distortion of the labial papillae (Rowe et al., 1996). Up to 96 % of individuals in the site exhibit some degree of abnormal oral morphology. These oral structures are crucial for effective grazing of periphyton; when placed on a diet restricted to periphyton, amphibians with oral abnormalities exhibit negative growth rates as a result of decreased grazing efficiency (Rowe et al., 1996). Spinal abnormalities are primarily characterized as lateral axial curvatures (i.e., scoliosis) and can occur at frequencies as high as 37% in

field-collected animals (Hopkins et al., 2000). These curvatures alter the normal propulsive movements of the tail resulting in decreased swim velocity in affected individuals. Such reductions may further exacerbate the slow swimming speed and responsiveness of amphibian larvae from the site, which is believed to affect their ability to escape predation (Raimondo et al., 1998).

Amphibian larvae collected and transplanted to D-area also exhibit increased metabolic rates, a sublethal effect that often accompanies contaminant exposure. Standard metabolic rates (corrected for body mass) of affected individuals may exceed metabolic rates of individuals from reference sites by as much as 30% (Rowe et al., 1998). Such increases in energy expenditure are predicted to result in reductions in energy allocation to processes important to fitness including growth, reproduction, and/or storage. Indeed, a recent laboratory experiment demonstrated that even when food resources were controlled, amphibian larvae exposed to CCW exhibited reductions in growth (Snodgrass et al., 2004). CCW-induced reduction in growth may have been the result of the above described energy allocation tradeoffs.

Indirect Effects

Indirect effects of contaminants are often much less obvious than direct effects, but may be equally or more important. Indirect effects typically arise when a contaminant alters some important feature of an organism's operative environment. For example, some herbicides may not be directly toxic to vertebrates but by altering vegetative cover the use of herbicides may expose organisms to higher risk of predation.

Recent work suggests that CCW disposal in aquatic systems decreases productivity of these systems resulting in reduction in food resources for higher trophic level organisms. Phytoplankton, periphyton, zooplankton, and benthic invertebrate abundance are reduced in D-area and in experimental mesocosms containing sediments collected from D-area (Rowe et al., 2001; Hopkins et al., 2004; Roe and Hopkins, unpublished). Such reductions are associated with alterations in growth and metamorphic traits in vertebrates and may, in part, explain observed reductions in amphibian recruitment from the D-area site (Rowe et al., 2001). Thus, when amphibians are attracted to sites such as D-area for breeding, their offspring are not only exposed to contaminants but also suboptimal resource conditions. Current research is evaluating the relative importance of direct and indirect effects in both herbivorous anuran larvae and carnivorous salamander larvae (Roe and Hopkins, in progress).

Maternal Effects

Developmental pathways in embryonic amphibians can be altered when embryos are exposed to contaminants via two primary mechanisms, uptake from their surroundings (water) and transfer from mother to offspring. Virtually everything we currently know about the effects of environmental contaminants on amphibian development is derived from studies where embryos undergo aqueous exposure to varying concentrations of a contaminant. In fact, aqueous exposure is the focus of >90% of all toxicity tests published on amphibians and has formed the foundation for the most widely accepted standardized amphibian toxicity test (Frog Embryo Teratogenesis Assay- Xenopus; ASTM, 1998). In contrast, no studies have been performed on maternal transfer of contaminants in amphibians (Linder and Grillitsch, 2000), despite its devastating effects on development in every other major vertebrate lineage (i.e., mammals, birds, reptiles, and fish; e.g., Heinz, 1996; Lemly, 1996). However, several studies have demonstrated that amphibians partition significant quantities of contaminants in female reproductive tissues, suggesting the potential for maternal transfer to offspring (Linder et al., 1998; Punzo, 1993).

In general, oviparous (egg-laying) wildlife are very sensitive to the effects of maternal transfer of contaminants. Maternal transfer is initiated early in oogenesis when females begin synthesizing large quantities of lipoproteins which are critical to oocyte development. In oviparous fishes, which have an an-amniotic egg similar to the egg of amphibians, these lipoproteins are crucial for supplying nutrients, essential trace metals, and hormones to the developing embryo (Specker and Sullivan, 1994). Unfortunately, these same lipoproteins may serve as transport molecules for environmental contaminants from mother to offspring. In some cases, the quantities of contaminants transferred can be adequate to disrupt early development (Lemly, 1996). Because the early stages of embryonic development direct the future developmental trajectory of the individual, maternal transfer represents an important vulnerability for developing offspring and ultimately for the mother's fitness.

Maternal transfer of metals and metalloids has been studied in all oviparous vertebrates (i.e., fish, birds, and reptiles)

except amphibians. In fish, birds, and reptiles, much research has focused on the effects of Hg and Se, two elements that are commonly released from coal burning power plants and maternally transferred to eggs in a dose-dependent manner (Bryan et al., 2003; Staub et al., 2004; Wolfe et al., 1998; Hopkins et al., in press). Although the precise mechanism of Se and Hg transfer has not been adequately addressed, both elements are likely transported to the egg via maternal lipoproteins (Kroll and Doroshov, 1991; Wolfe et al., 1998). Once in the egg, Hg and Se disrupt embryonic development resulting in increased frequency of developmental abnormalities and reduced embryonic survival (Lemly, 1996; Wolfe et al., 1998).

We recently conducted two pilot studies that indirectly suggested amphibians, like other oviparous vertebrates, maternally transfer contaminants to their eggs. We collected adult narrow mouth toads (*Gastrophryne carolinensis*) during the breeding season from a reference site and from the D-area facility. Males and females from each site were paired and allowed to breed under controlled uncontaminated conditions. After collecting eggs from the pairs, we then conducted two pilot experiments to assess embryonic survival. In the first experiment, we transplanted eggs back into artificial wetlands containing sediments from the parents' home site. We found that embryonic survival of ASH eggs was significantly reduced compared to REF eggs. To ascertain whether this response might be due to the developmental environment of embryos, we then transplanted REF eggs to both REF and ASH wetlands; hatching success was >90% in both sites. These results suggested that the reduced survival of embryos from ASH females (in study #1) was not due to the developmental environment, but was more likely due to some aspect of egg quality. Based on these preliminary results, we hypothesized that egg quality of ASH females is compromised by maternal transfer of contaminants, ultimately disrupting normal embryonic development. Currently, funding provided by several agencies is supporting our efforts to test this and related hypotheses.

Conclusions & Future Directions

Taken together, the research conducted over the last 12 years in D-area suggests that surface impoundments containing CCW can attract amphibians from other sites and as a result of direct, indirect, and/or maternal effects have significant impacts on their reproductive success. However, many questions remain unresolved and require rigorous attention from scientists and regulators. For example, do the observed effects ultimately affect local amphibian population dynamics? Do similar effects occur at other CCW contaminated sites (i.e., is D-area a good model system)? What species are most sensitive to CCW and what traits contribute to their relative sensitivities? These and other questions must be addressed in order to fully appreciate the impacts of CCW aquatic disposal practices on amphibians.

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Session 3

ENVIRONMENTAL DAMAGE CASES

Session Chairperson:
Peter Michael
USDOJ Office of Surface Mining
Pittsburgh, Pennsylvania

Chisman Creek Superfund Site: A Retrospective Review

Robert J. Williams, Dominion Resources Services, Inc., Glen Allen, Virginia

Ground Water Impacts from Coal Combustion Ash Disposal Sites in Wisconsin

Michael W. Zillmer (Milwaukee) and Philip Fauble (Madison), Wisconsin Department of Natural Resources

Leaching of Arsenic from a Fly Ash Dump in Beverly, Massachusetts

David Lang, Ground Water Consultants, Inc. and Jan R. Schlichtmann, Beverly, Massachusetts

Environmental Concerns and Impacts of Power Plant Waste Placement in Mines

Charles Norris, Geo/Hydro Inc., Denver, Colorado

Impacts of Aquatic Disposal of CCBs on Vertebrates on the Savannah River Site

Dr. William Hopkins, University of Georgia, Savannah River Ecology Laboratory, Aiken, South Carolina

LOSING THE OPPORTUNITY TO BENEFICIALLY USE CCPS WHEN POLICY IS NOT BASED ON SITE-SPECIFIC CONDITIONS OR CURRENT MANAGEMENT PRACTICES

Stephen B. Dixon
National Gas & Energy Transmission
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Introduction

National Energy & Gas Transmission operates power generation units throughout the United States. Eight of them are coal-fired, including two circulating fluidized bed (CFB) units that burn waste coal – one in eastern Pennsylvania and the other in the western part of the State. The operation of these two CFBs, which burn the material to generate steam and electricity, has removed millions of tons of waste coal from the environment. The ash generated during this process is alkaline in nature and has been used to reclaim hundreds of acres of abandoned mine land in Pennsylvania.

When I first joined the Company, I kept getting calls from our folks out in the field telling me that just seeing the water quality data collected from the reclamation sites wasn't nearly as impressive as what was actually occurring in the field. They were correct. What I saw at the sites was indeed impressive, the classic "before and after" --- a "moonscape" where the waste coal had yet to be removed, adjacent to a vegetated site that had been reclaimed using coal combustion products.

I believe that it is this combination of analytical data and the actual results in the field that should provide the strongest positive response to concerns relating to using coal ash for mine reclamation in Pennsylvania.

Therefore, to Pennsylvanians and others, a real concern should be that of losing the opportunity to remove millions of tons of waste coal from the environment and to reclaim damaged sites using coal combustion products. In fact, these sites ---- those that have yet to be reclaimed and are a continual source of AMD that ravage miles of streams throughout the Commonwealth --- are the "damage cases" that desperately require our attention.

So with that in mind, it is very concerning to hear calls for broad-brush changes relating to the regulation and management of CCPs – like calling for moratoriums or calling for coal ash to be regulated as hazardous waste. Such calls could result in losing the ability to beneficially use coal combustion products for mine reclamation. And to take this a step further, it is especially concerning when the calls for change are based on older sites where ash management practices once conducted are not representative of today's practices.

Site Specific Examples

I've got a specific example that addresses this point that I would like to share with you. My example involves two sites – one discussed in the earlier session of this forum, the Vitale Site or Wenham Lake Site (I'll be using the names interchangeably) and another site in Massachusetts located around 70 miles away – the Copicut Road structural fill in Freetown, Massachusetts.

Vitale Site

First the Vitale site --- and I'd like to take my hat off to all the stakeholders involved with the site, it sounds like it was a real team effort in addressing the challenges posed by this site. As you may recall, the Vitale site was operated by two brothers back in the 1950s, 60s, and early 70s. As you have heard, the selection of this particular location, how the site was managed, and how it was closed, unfortunately lead to the migration of ash from the site.

However, even though there were numerous shortcomings related to this old site, the picture is not as gloomy as once depicted. From what I heard during the earlier presentation on this site, there were actually some very positive findings:

- Monitoring wells located down-gradient of the site show very low to non-detectable concentrations of arsenic.
- None of the compounds associated with fly ash have been detected in surface water in the Lake at concentrations greater than one-tenth of the current drinking water standards.
- The site is currently being evaluated for trespassers/recreational users. The likely conclusion will be “No Significant Risk.”
- The weight of evidence presently indicates “No Significant Risk” for ecological receptors and pathways.

Copicut Road Site

Now, let’s move ahead to the Copicut Road site. The site became active in early 2000 (around 40 years after the Vitale site began operation). The site is around 10 acres in size, and was slated to be active for 2-3 years. Coal ash was to be brought to the site to be beneficially used as a structural fill. Two buildings were to be constructed at the site once it was brought up to final grade.

In early 2000, the owners of the site met with the local Board of Health and the Massachusetts Department of Environmental Protection to review the project. The Department concluded that the project met their criteria regarding coal ash structural fills based on the following:

- The site had not been previously used for landfilling purposes;
- Only ash from coal combustion was to be utilized for fill material;
- The structural fill will result in final contours consistent with surrounding grades;
- A final end use had been determined, construction scheduled, and the final occupant identified; and
- Duration of each phase of site construction was consistent with typical projects.

In addition, there are some additional facts about the operation of the site that I would like to point out:

- Conditioned ash (10-15% water added to minimize the generation of dust during handling) was brought to the site and spread/compacted in two-foot lifts. The ash was compacted to 90% dry density with in-place permeability numbers of around 1×10^{-6} cm/sec.
- A water truck was operated on the site in order to control fugitive dust. Wood mulch was placed on the inactive areas to control fugitive dust as well.
- Groundwater monitoring wells were installed prior to placement of coal combustion products and were sampled before and during placement of the ash. There were five wells, one up gradient and four down gradient. Samples collected were analyzed for metals, volatile organic compounds, semi-volatile organic compounds, PCBs, ammonia, total dissolved solids, nitrate, and pH.
- When final contours were reached, 18 inches of soil cover was to be placed and two buildings were to be constructed on the ash.

Operations then began in early 2000, but soon the local residents began hearing the warning “what happened at Wenham Lake will happen here in your town.” In addition, the same folks that drew the comparisons between the Vitale site and Copicut Road labeled Copicut Road a coal combustion product “damage case” and submitted its name to EPA for inclusion on EPA’s list of “damage cases” – which it was.

Then, after a series of rallies and public meetings, an “article” was drafted by a few residents of Freetown, calling for a ban on storing, disposal, stockpiling, or using coal ash as fill material for any and all purposes. A town meeting was held and the article was passed and became a law.

In October of 2000, less than a year after operations started at Copicut Road, the site was shut down, even though analytical data collected from monitoring wells at the site showed that placement of the coal ash at the site had absolutely no impacts on groundwater quality.

Conclusion

Broad-brush positions can result in poor policy, and generate concerns where they may not exist. In fact, earlier in the year Pennsylvania's Joint Conservation Commission studied whether the call for a moratorium on the use of coal combustion products in mine reclamation was warranted. Their conclusion was that the call for the moratorium was asking for protection from a danger that does not exist.

It is essential that the regulation and management of coal ash be based on facts and reflects site-specific hydrogeologic conditions, the characteristics of the coal combustion products, and ash management practices at a particular site. Closing down the Copicut Road site based on what happened at another site decades earlier is the perfect example of how deviating from site-specific considerations can led to questionable decisions.

And by the way, this is not to say that there are not cases where concern is warranted. If there is indeed a site that meets EPA's definition of a "damage case" then aggressive efforts must be adopted to address the situation. But, continuing to throw out names of sites as "damage cases" in order to generate concerns when the facts don't back up the allegations is an inefficient use of all of our efforts. For example, last fall at a symposium in Lexington, KY, I heard someone say there were over 122 "damage cases." Well, if you include sites like Copicut Road on the list you may get such a number. Incidentally, EPA reviewed the facts on Copicut Road site and has now designated the final review complete and it has been rejected as a "damaged case."

So let's continue to aggressively reclaim the abandoned mine sites using coal combustion products - the damage cases that are a concern throughout the Commonwealth.

Steve Dixon is Water, Waste, and Hazardous Materials Manager with National Energy and Gas Transmission (NEGT) located in Bethesda, Maryland. NEGT owns and operates power generation facilities throughout the United States, including coal-fired stations in Pennsylvania, Florida, New Jersey, and Massachusetts. NEGT has consistently beneficially used nearly 70% of the CCBs generated by its facilities. Steve has over 15 years experience with environmental issues related to power generation. His responsibilities include environmental permitting, technical report preparation, environmental auditing/due diligence, regulatory tracking, and conducting site assessments. A significant portion of his efforts involve identifying and evaluating alternatives for managing coal combustion by-products, such as mine reclamation, as a raw material used in the production of cement, and as an ingredient in flowable fill. Steve holds a Bachelor of Science in Environmental Resource Management and a Master of Environmental Pollution Control from the Pennsylvania State University. He also has an MBA from the University of Kentucky.

PENNSYLVANIA'S ABANDONED MINE LAND HAZARDS

Larry LaBuz
Electric Power Generation Association
Harrisburg, Pennsylvania

Introduction

Good afternoon. My name is Larry LaBuz and I am here today speaking on behalf of the Electric Power Generation Association (EPGA). EPGA is a regional trade association of nine electric generating companies with headquarters in Harrisburg, Pennsylvania.

EPGA member companies own and operate more than 130,000 megawatts (MW) of electric generating capacity in the U.S., approximately half of which is located in Pennsylvania and surrounding States. Nearly 50% of the generating capacity in Pennsylvania uses coal or waste coal as its primary fuel.

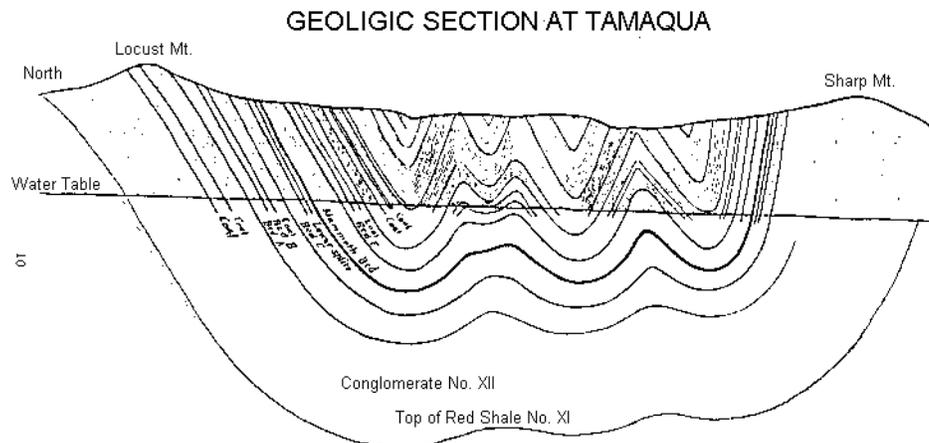
In Session One, Dr. Scheetz discussed the chemical and mineralogical properties of coal ash that make it a promising part of the solution to the chronic acid mine drainage problem in Pennsylvania. Acid mine drainage is the number one cause of groundwater and surface water pollution in Pennsylvania.

Pennsylvania Abandoned Mine Hazards

My talk is going to focus on a different, but equally serious problem. Many of Pennsylvania's abandoned minelands pose severe, and oftentimes hidden, hazards to public safety. Precipitous highwalls and actively subsiding deep mine openings present dangerous threats to hikers, hunters, mountain bikers and other outdoor enthusiasts. Eliminating these threats is a real challenge where backfill materials are generally unavailable or undesirable due their acid-producing potential. Coal ash that has been properly tested and certified for use in mine reclamation can virtually fill this void.

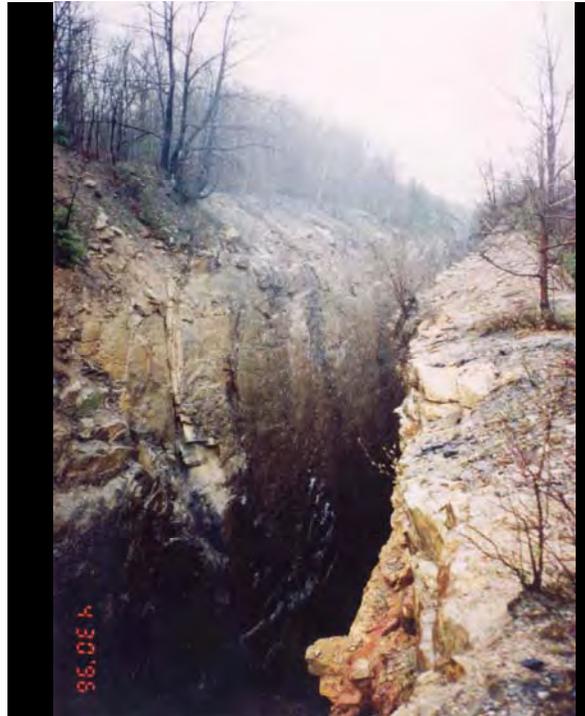
EPGA member companies produce approximately 9 million tons of coal ash per year, less than 40 percent of which is beneficially used. The proximity of many of these coal-fired power plants to the anthracite coal fields makes the coal ash that they produce a valuable resource to local communities and organizations seeking ways to remove these hazards from their communities.

A classic example of where backfill material is almost always lacking is crop falls. In the Anthracite Region, coal seams often curve and dip, and sometimes lay vertical as seen in the following geologic cross section.



Geological section of anthracite beds near Tamaqua, Pennsylvania. From a United States Geological Survey of Pennsylvania in 1883.

Where coal seams outcropped vertically, they were mined out from below, and eventually collapsed producing crop falls as shown in the following photograph.



Crop falls can be up to 100 feet wide and 400 feet deep. The hazards they pose are obvious, but how to deal with them is a major dilemma because of their size, depth, location and numbers. The fact is, there is no material available to backfill them because material from the crop fall area has subsided into the deep mine as shown in the following photograph.



Crop falls can extend for miles. You can imagine the danger they present to hunters or hikers in the area who are unaware of their presence. Crop falls may be enticing to explore but the adjacent ground is extremely unstable, especially during the spring of the year after freeze/thaw conditions weaken the underlying rock.

Fortunately, a promising remedy to the crop fall problem was recently demonstrated on Sharp Mountain in the city of Pottsville. State and local government along with several community-minded companies partnered together to test a process to seal the crop falls using coal ash, concrete panels and recycled steel as seen in the following photograph.



The materials were used to create a tapered "plug" to prevent continued subsidence into the underlying mine openings as seen in the following photograph.



Coal ash was then used to backfill the area to its original grade, so that precipitation can run over the sealed crop fall, rather than into the already contaminated mine pool.



Soil was then placed over the compacted fill and vegetated as seen in the above photograph.

While the primary goal of projects such as these is to address public concerns associated with abandoned minelands, we are also sensitive to the public concerns associated with the remedy itself.

Today, rules governing the placement of coal ash for mine reclamation are designed to protect, or more often, to improve the environment. This is especially true in Pennsylvania where our regulatory program for these projects is viewed as a model for the nation.

And while we are confident that Pennsylvania's regulations protect public health, EPGA-member companies have committed to go beyond what is required. We have listened to the recent concerns raised by residents living near active mine reclamation projects. And in response, EPGA is developing a best management practice standard for the use of coal ash in mine reclamation.

The measures in this standard are intended to ensure that projects are well-managed and that public concerns are addressed before, during, and after a project. They address routing of truck traffic, tarping standards, dust control, and testing. The practices are currently being used by companies who have established positive relationships with neighboring communities. We know they are effective. Member companies involved with mine reclamation are expected to implement these best practices within their individual programs by the end of the year.

But we know that "best practices" are not static. They evolve. And EPGA stands ready to work with community leaders and environmental interest groups to ensure that the practices guiding coal ash mine reclamation projects in the future continue to address community concerns as we work together to conserve natural resources and eliminate public health and safety hazards.

Larry LaBuz is Supervisor of Ash Operations at PPL Generation where he is responsible for developing and managing PPL's coal ash use projects. When he joined PPL's Environmental Management Division, he oversaw the development of PPL's ground water monitoring program. He also supported the siting and permitting of PPL's coal ash disposal facilities and chaired PPL's corporate solid waste management team set up to develop strategies for pollution prevention, recycling, reuse and disposal. Between 1996 and 1998, Larry worked at PPL's Martins Creek Steam Electric Station as Plant Environmentalist. He was responsible for environmental compliance activities including spill response, air and water quality data evaluation/reporting, and solid waste minimization and disposal. External to PPL, he chaired the Pennsylvania Electric Association's Solid Waste Subcommittee during the development and implementation of the PaDEP's residual waste regulations. In the early 1990's, he also chaired EPRI's Land & Water Quality Studies Program directing research on the leaching behavior and environmental fate of coal ash constituents. He graduated from the University of Notre Dame in 1980 earning a Bachelor of Science Degree in Civil Engineering, specializing in wastewater treatment and control.

A RESPONSE TO ALLEGATIONS OF DAMAGE TO WATER QUALITY DUE TO PLACEMENT OF FLUIDIZED BED COMBUSTION MATERIAL AT ABANDONED MINE SITES IN PENNSYLVANIA

Dennis Noll, P.G.
Earthtech
Johnstown, Pennsylvania

Introduction

Good afternoon. My name is Dennis Noll. I am a Pennsylvania Registered Professional Geologist. I have been working in environmental geology since 1969 and I have managed projects that deal with the environmental aspects of coal combustion byproducts since 1988. Those of you who attended Session 1 know that Dr. Rose and I collaborated on a study, completed in September, 2000 that addresses the occurrence and fate of hazardous trace elements in combined cycle fluidized bed combustion (CFBC) ash, the waste coal burned to make that ash, and the limestone consumed in that process. Those of you that attended that session also know that water-monitoring results over a 16 year period were examined at 14 of the most significant ash placement sites in Pennsylvania. A total of 854 water sample results, containing hazardous trace element analysis, were scrutinized for that study. After that study was completed, an additional 3870 water sample results have been analyzed to update through 2003 the trace element data and acidity, iron, manganese, aluminum, and sulfates.

With respect to the trace elements, the results produced a strong conclusion that burning waste coal and reclaiming mine sites with the resultant CFBC ash nets a significant environmental benefit. We found that not only did stringent laboratory testing indicate that these hazardous elements were strongly bound in the ash, but empirical field data, collected at the ash placement sites, backed up the laboratory findings. With respect to the additional parameters studied at these sites there does not appear to be any pattern of negative environmental impact from release of the contaminants listed.

However, much to our surprise, some of these sites have been identified by consultants who represent Clean Air Task Force, Hoosier Environmental Council, and the Army for a Clean Environment as locations where the ash has released hazardous compounds that have adversely affected the chemistry of the surface and ground water. These consultants gave “sworn” testimony about these sites before the Joint Conservation Committee of the PA Legislature on July 9, 2003 in Tamaqua, PA. and presentations at the offices of EPA in Washington, D.C. in May 2003.

This testimony was totally inaccurate in that most of the facts in the public record were completely ignored and the consultants disregarded or misinterpreted the scientific factors that control these sites. At the time the testimony was presented, the consultants had not visited one of these sites. Instead, they selected data or portions of data in the public record that appeared to show water quality degradation and, on the basis of timing only, concluded that the placement of ash was responsible for degradation. This was done even at sites where an overwhelming amount of public data indicated that vast environmental improvement occurred after ash placement with no commensurate environmental degradation.

The sites these consultants identified as “damage sites” in their oral and subsequent written testimony were Ernest, Revloc, Colver, Big Gorilla, and McDermott, all located here in Pennsylvania. After witnessing their presentations I took a detailed look at these sites with respect to their allegations. I have concluded that this testimony is incorrect and misleading for the following reasons:

Site specific Analysis

Ernest Mine Site

The Ernest Mine Site is located in White and Rayne Townships, Indiana County, PA. Prior to 1965, this site received coal refuse for many years from the mining and beneficiation activities associated with the Rochester and Pittsburgh Coal Company’s mining activities. The Ernest #2 Mine, which deep-mined approximately 14,000 acres

of Upper Freeport coal under and adjacent to the coal refuse deposit, is known to be one of the mines that contributed to the refuse placed at this site.

The coal refuse was placed by Rochester and Pittsburgh Coal Company in a stream valley that contained a tributary to McKee Run. The material filled all of the valley except the upper headwater areas, covering approximately 90 acres and achieving a thickness in excess of 100 feet in places. The position of the exhausted Ernest #2 deep mine is located 45 to 110 feet below the base of the coal refuse.

The coal refuse now is being removed by RNS Services for use as fuel at the Cambria Co-Generation facility in Ebensburg, PA. The site is permitted by PA DEP for this type of activity as well as replacement of the coal refuse with CFBC ash from Cambria Co-Gen. Early estimates (re. Todd Coleman, Minetech Engineers) of the amount of coal refuse placed at this site were approximately 10 million tons, although additional information gained by site activities that began in 1996 led the managers of the site to believe that these estimates were low.

Since removal of the coal refuse reached full production in late 1996 - early 1997, 3.8 million tons of material have been removed from the site (Todd Coleman, Minetech). Reports filed with PA DEP indicate that 1.4 million tons of CFB ash have been placed at the site during that time. To date, the ash has been placed to the southeast of the coal refuse mining area because refuse removal has affected large areas in multiple pits due to varying fuel quality characteristics. Because of the large-scale disturbance of the coal refuse, ash has not replaced coal refuse in exactly the same areas, although this has begun and will occur more regularly now that operations at the site have matured.

The monitoring points illustrated by the consultant's report exhibits and overhead projections were not in locations where water affected by the ash could have appeared during the time frames selected. Water that was in contact with the ash and readily available for sampling bears no resemblance to the results illustrated by the overheads. In fact, sampling of that water, conducted by the PA DEP, shows that the water quality meets Federal drinking water standards for all parameters subject to those standards. The water quality trends illustrated in the consultant's report exhibits and overhead projections were due to initial activities related to the removal of the coal refuse and are indicative of the type of water quality to be expected if, as suggested by the consultants, coal refuse were used as mine fill. At the time of the consultant's water quality time window, coal refuse removal activities were taking place much closer to the monitoring points than the ash placement activities. Also, water affected by the ash disposal activities at the time period occupied by his data window would not have flowed toward the monitoring points and, even if it could, it would have been diluted or completely masked by the water flowing from the much closer coal refuse removal activities.

I am not alone in my analysis of this site. Some of DEP's brightest scientists and engineers have examined this site twice in light of the allegations by these consultants. They also strongly disagree that this should be considered a "damage site." Their written findings, as well as, mine have been submitted to the Joint Conservation Committee.

Revloc Mine Site

Prior to 1965, this site, located in Cambria Township, Cambria County, PA, received coal refuse for many years from the mining and beneficiation activities associated with the Bethlehem Mines Corporation Mine #32 (also known as the Revloc Mine). This mine, developed on the Lower Kittanning coal seam and located approximately 350 feet below the elevation of the bottom of the coal refuse, underlies the entire coal refuse and planned ash disposal area and extends at depth, encompassing thousands of acres. Bedding dips from northwest to southeast at the site and the flow direction of the underlying deep mine pool is parallel to bedding.

In 1987, Ebensburg Power Company began the permitting process to utilize the coal refuse for fuel and place the CFBC ash at the Revloc site. The initial site to be developed was known as the Revloc #1 site because this site contained coal refuse that was unreclaimed. In 1997, the portion of the Revloc site that had been partially reclaimed, Revloc #2, was permitted. In 1991 Ebensburg Power Company began removing the coal refuse from Revloc #1 for their CFB facility near Ebensburg, PA. As of December, 2002, approximately 2 million tons have been removed and consumed by the plant. The coal refuse is screened at the site to remove the +2.0" material, resulting in an average recovery for fuel use of 50% to 65%. This means a maximum of approximately 3 million tons have been processed through December, 2002. Approximately 2 million tons of ash have been placed through December, 2002.

Removal of coal refuse began in 1991 on the northern portion of the Revloc #1 site, and progressed in a southerly

direction with ash and oversized coal refuse placement following the refuse removal. Removal of the coal refuse on the Revloc #2 site began in 1998 to the east of the Revloc #1 site on the opposite side of Black Lick Creek. Refuse removal began at the eastern extremity of the Revloc #2 coal refuse pile and has progressed in a westerly direction toward the portion of the Revloc #1 site where the refuse removal is presently taking place. The ash and oversized coal refuse placement trails the mining advance to the west at Revloc #2.

Water quality during the periods cited by the consultant at two monitoring points has nothing to do with ash placement. In the first case, water quality from 1992 through 1995 was examined in a well that can receive water flow from coal refuse and ash placement. At that point in time, the coal refuse occupied most territory upslope from the well for a distance of 1200 feet and ash was being placed at a distance 200 to 400 feet further uphill. Since then, coal refuse removal activities have eliminated most of the pile to a distance of 400 feet from the well and ash placement has now approached to within 600 feet of the well. The removal of coal refuse and the advancement of ash placement toward the well since 1995 has been accompanied by improvements in water quality at the well, compared to pre-ash placement levels. The second well cited by the consultant had virtually dried up by the time the data he examined had been generated, leaving only bottom sludge to sample. This trend has continued and the data he obtained is not applicable for any study.

Completely ignored by the consultant was the fact that significant water quality improvement has taken place at three of the remaining monitoring points at the site that could receive water from areas where the ash has been placed.

This site also has been extensively studied by DEP and is offered as an example of successful beneficial use of coal ash in the paper presented in Session 1 (“Remining and Reclamation of Two Large Bituminous Waste Coal Piles Using FBC Ash: A Watershed Perspective,” Timothy Kania, Pennsylvania DEP, Cambria Office, PA, 2004) and in “Coal Ash Beneficial Use in Mine Reclamation and Mine Drainage Remediation in Pennsylvania” by the Pennsylvania Department of Environmental Protection and the Pennsylvania State University Materials Research Institute, 2004.

Colver (Rail Yard/Coal Yard)

The “Coal or Rail Yard Site” is part of the coal refuse removal and ash placement site associated with Maple Coal’s Colver mine site, located at Colver, Cambria County, PA. This site is a typical coal refuse disposal site in that a valley and surrounding areas were filled with approximately 8 million tons of coal refuse from a large deep mine project. The coal refuse is being removed for use as fuel at the nearby Colver Power Plant and the CFBC ash is returned to the coal refuse site to replace that material.

The consultant examined data from two monitoring wells that are located down-slope from abandoned coal refuse and ash placement and concluded that certain water quality parameters were improving until ash placement activities began to take place at the site. As is the case with Revloc, during the years of data selection by the consultant (1994 through 2000), the coal refuse occupied the immediate area upslope from the monitoring wells for a significant distance. Ash placement began in a location that was further uphill with respect to the wells. It is true that, at the time coal refuse removal activities took place near the wells, the rate of improvement in water quality for certain parameters decreased from 1994 through 2000. It is also true that as more coal refuse has been removed, since the end of his selected data window, the rate of water quality improvement for these parameters has increased to the point that the overall rate of improvement has been greater than that which occurred prior to the placement of the ash. That, of course, was not mentioned.

Also ignored was the data from the downstream point on the receiving stream which shows that, since coal refuse removal and ash placement, there has been a dramatic improvement in water quality and an equally dramatic reduction in pollution loading at that point.

This site has also been extensively studied by DEP and is offered as an example of successful beneficial use of coal ash in the paper presented in Session 1 (“Remining and Reclamation of Two Large Bituminous Waste Coal Piles Using FBC Ash: A Watershed Perspective,” Timothy Kania, Pennsylvania DEP, Cambria Office, PA, 2004).

Big Gorilla

The Big Gorilla, a water-filled surface mine, is located in the Silverbrook Basin in northern Schuylkill County near the town of McAdoo. The Big Gorilla Pit represented approximately 25% of the water in the Silverbrook Basin and was a “class 2” health and public safety hazard. The outflow of the Silverbrook Basin is acid mine drainage that during low flow conditions constitutes the headwaters of the Little Schuylkill River, a significantly degraded river down stream of the discharge.

During August of 1997, a permit was granted by the PA DEP that allowed Northeastern Power Company (NEPCO) to place cogeneration (CFBC) ash directly into the water-filled pit. The objectives of the demonstration were to show that: (1) ash could be placed into standing water without creating significant turbidity; (2) the ash could be placed into standing water without resulting in environmental degradation; and (3) the resulting structural fill would be stable.

During the month of August 2003, the Big Gorilla demonstration was completed when the last of the pit was filled with ash. With the completion of the phase I demonstration, the project has advanced to phase II, during which the final ash platform will receive ash as a structural fill to bring the final contour of the mine to approximate original contour (AOC).

The consultant testified “the coal ash placement converted the pit water from an acidic pH supportive of at least some life to a caustic level lethal to all life, and corrosive enough to qualify as hazardous by characteristic.” Characterization of the acid level of the Big Gorilla pit water, prior to ash placement, as “supportive of at least some life” is extremely misleading. The pH level, along with the high metal content and sulfates, would have restricted life to a very small set of species with limited diversity which would not be characteristic of a healthy aquatic environment. The statement about the ash raising the pH to a level that is hazardous by characteristic is also false because the Federal regulations that define that characteristic require a higher pH than that which was reached in the pit before the ash absorbed all of the water. Public records are available that demonstrate beyond a shadow of doubt that at all of the monitoring points that check the aquifers in contact with this pit water, there was no significant change in pH. The consultant admitted to checking one of these monitoring points, the Silverbrook Mine discharge, stating “it (the pit water) has had zero effect on the acid discharge of the Silverbrook outlet.”

This site also has been extensively studied by DEP and is offered as an example of successful beneficial use of coal ash in “Coal Ash Beneficial Use in Mine Reclamation and Mine Drainage Remediation in Pennsylvania” by the Pennsylvania Department of Environmental Protection and the Pennsylvania State University Materials Research Institute, 2004.

McDermott Mine Site

The McDermott Mine is located in Jackson Township, Cambria County, PA. It is a site previously affected by surface and deep mining that degraded the headwaters of the receiving stream, Hinckston Run. The mine plan submitted for this site proposed complete removal of the previously deep-mined area, reclamation of abandoned spoil piles and highwalls on the site, and additional coal removal by the surface mining method. The mining plan also proposed alkaline addition in the form of CFBC ash from two power plants, located in Cambria County. The purpose of this material was to counter the high potential for AMD production from the strata to be encountered at this site.

The permit to implement this plan was issued in January, 1996 and activated in April of that year. As of this date, typical inorganic mining parameters have degraded at a number of monitoring points, without any corresponding increase in hazardous trace elements or other parameters that would indicate contamination by the CFBC ash. This site was extensively studied by DEP and they concluded that ash placement at this site was not responsible for any of the water quality contamination. This opinion is based upon the fact that the mining plan was not carried out in a timely fashion, missing the opportunity to correct those conditions that caused acid mine drainage to form prior to ash placement and leading to exacerbation of these conditions because the site was ultimately abandoned before reclamation was completed. This study is fully documented in “Coal Ash Beneficial Use in Mine Reclamation and Mine Drainage Remediation in Pennsylvania” by The Pennsylvania Department of Environmental Protection and The Pennsylvania State University Materials Research Institute, 2004.

Water Quality Data Analysis

The consultant testified in front of the LCC that water quality degradation occurred as a result of the placement of coal ash and that the cause of the degradation was the permeability characteristics of the ash that was placed. Permeability is a property that measures the ability of a material to allow liquids to pass through it. Some materials, such as coal ash or clay, can be less permeable than other materials; therefore liquids will pass through those materials much more readily than compacted coal ash. According to the consultant, the lower permeability of this ash allegedly altered the flow of water polluted by a source other than the ash to flow toward a spring. This claim has not been substantiated by DEP, but it should be noted that any material, even if it is totally pristine, can cause a problem with water flow if it is not properly placed.

In spite of the opportunities presented in Session 3 to share with this audience the claims made about these “damage cases,” only a slight reference to one of these sites was mentioned with no details presented. Instead, five new sites were presented, not as “damage sites,” but as sites with “negative impacts.” The new term, “negative impact,” was introduced in Session 3 as a replacement for the term “damage case.” Among the conditions proposed to identify a site with a “negative impact” are:

- When new contamination impairs or precludes a current or potential use
- Existing contamination increases or new contamination is introduced
- When alteration or remediation of an existing, contaminated water supply source has been delayed or deferred
- When ash placement has been used to justify a project that could not prevent the release of contaminants on its own merits and the ash placement fails to prevent the release of those contaminants (alkaline addition at surface mines)

While the term “negative impact” may appear to be more benign than the term “damage case” it creates the opportunity to make an unbalanced and biased point by presenting data collected during convenient windows of time or by ignoring the consideration of all available data. These are the techniques that have been employed to date. The burden required to substantiate a “damage case” has been found to be too difficult to meet, especially in light of the extensive studies by this author and DEP that have looked at all of the data for each of these sites.

The new cases with “negative impacts” that were presented in Session 3 by one of the consultants mentioned above offer the same opportunities for creating an impression that consideration of all of the data would lead to the conclusion that there is contamination being generated at a given site. For instance, the presenter said that four out of the five new sites mentioned had no record of any ground or surface water contamination outside the ash placement area and that the “negative impact” declaration was based upon the quality of water sampled in wells drilled into the ash fills. The “negative impact” designation for the fifth site was based upon wells in the ash fill and a “small seep.” To prove the point, we were shown time vs. concentration graphs for selected parameters for which none of the dates were legible.

None of these cases appear to meet the criteria offered during Session 3 to qualify as a “negative impact.” The fact that water quality found in wells developed in the fill itself do not meet a certain, arbitrary standard is immaterial. Some of the speakers in Session 1 explained the chemical and physical dynamics of ash hydration and dehydration and it was obvious from those presentations that ash pore-water quality and quantity changes with time. In fact, once the pozzolanic process is completed pore-water is practically non-existent and it is certainly not mobile.

The real measure of the potential for “negative impact” associated with a site should be measured at various distances from the ash fill areas. That is how PA DEP’s Module 25 water monitoring program operates. Monitoring points are established at all of the ash placement sites, many of them within 100 feet of the fills, to determine chemical changes over time. Receiving streams that support aquatic life and water supply sources, examples of protected uses, are located much further down gradient than the monitoring points. The purpose of the monitoring points is to provide an early warning to enable the protection of those uses should a contamination plume be generated by the fill. Monitoring points are not wells placed within the ash fills for good reason: the chemical and physical dynamics explained above and the fact that dilution by ground water and attenuation by soils are not taken into consideration.

Also criticized in Session 3, was the lack of sufficient time at existing sites to monitor the effects of the ash placement. The Module 25 monitoring program has produced laboratory analysis for samples taken from all of the monitoring points for 18 years from 1986 through 2003 and is continuing today with no plans to cease that practice. As an example, for the 17 sites that host at least 70% of the CFBC ash produced to date the following has been

accomplished during the period 1986 through 2003:

- 1,065 sampling events for trace elements: 646 from wells, 36 from receiving streams, 68 from ground water discharges, 277 from mine discharges, and 38 from runoff pond discharges.
- 3,659 sampling events for inorganic parameters: 1,037 from wells, 180 from receiving streams, 1,986 from ground water discharges, 203 from mine discharges, and 253 from runoff pond discharges

By contrast, nine quarters of data were offered in Session 3 to define one of the sites as exhibiting a “negative impact.”

In response to comments made during the EPA Listening Session last night by those conducting a study to determine negative impacts at ash placement sites in Pennsylvania, complaints were registered that research at PA DEP offices was made difficult because of missing data and unavailability of information to determine the types and dates of activities at the sites, inferring, perhaps, that the proper amount of cooperation is not being offered. Please be informed that all of the data necessary to evaluate any of the sites can be found at the PA DEP offices. At some district offices, the water monitoring data must be extracted from individual files, however, the Pottsville office has tracked this data for years and has entered most of it on spreadsheets. As for the times and dates of site activities, each file contains a copy of the inspection reports by the District Mine Inspector for the site. Additionally, the operators of the sites file an annual report of ash placement with the appropriate district office.

Conclusions

1. Laboratory testing of CFBC ash and real-world water testing at the major CFBC ash-placement sites in Pennsylvania produce a strong conclusion that burning waste coal and reclaiming waste coal and abandoned mine sites with the resultant CFBC ash nets a significant environmental benefit.
2. Consultants for the Clean Air Task Force, Hoosier Environmental Council, and the Army for a Clean Environment have made allegations through sworn testimony and written reports to the LCC of water contamination at some of the sites studied by this author and PA DEP. These allegations are baseless in that most of the facts in the public record were completely ignored and the scientific factors that control the sites were disregarded or misinterpreted. At the time the testimony was presented, the consultants had not visited one of the sites and they selected data or portions of data in the public record that appeared to show water quality degradation and, on the basis of time periods only, concluded that the placement of ash was responsible.
3. The proposal to replace the term “damage case” with “negative impact,” introduced in Session 3, provides an opportunity to make an unbalanced and biased point by presenting data collected during convenient windows of time and/or by ignoring the consideration of all available data. Such an opportunity leads to the creation of an impression that consideration of all the data would lead to the same conclusion. This was demonstrated in Session 3 when the analyses of water, extracted from wells installed in ash fill cells at several sites, were used to create the impression that one would expect contamination of streams and water supplies off the site. This “negative impact” was postulated in spite of the fact that no additional on or off-site water analyses were offered for consideration.

Dennis Noll is a Registered Professional Geologist in Pennsylvania and has practiced for a total of 35 years in the field of environmental geology. In 1980, he co-founded Earthtech, a firm largely dedicated to dealing with the environmental requirements of mining companies and energy producers. Since 1988, he has managed projects for Earthtech that involve the environmental aspects of coal combustion by-products. Most of these projects have dealt with those by-products generated by circulating fluidized bed boilers. In 2000, Earthtech was awarded a contract, managed by Noll, to produce a study for ARIPPA that characterized the occurrence and fate of trace elements contained in the CFB fly and bottom ash generated by all 12 of that organization’s member power producers in Pennsylvania. Since then, he has guided Earthtech’s environmental oversight work at the ash placement sites studied in that report.

CFBC ASH AND MONITORING WELL CHEMISTRY COMPARED WITH NORMAL ROCKS, SOILS, AND WATERS

Arthur W. Rose
Professor Emeritus of Geochemistry
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Evaluation of Potential Damage Cases

In examining the significance of possible damage cases from disposal of coal ash, a number of factors should be evaluated. These include:

- **Placement technology.** Was the ash placed in accordance with current regulatory standards, in a well designed placement site?
- **Character of ash.** Was the ash tested for harmful leachate, using a TCLP, SPLP or similar test? How does the ash compare in composition to common soils and rocks?
- **Monitoring well significance.** Does water from monitoring wells evaluate only runoff from ash, or do they include drainage from coal refuse or abandoned mines?
- **Monitoring wells vs. background and MCL's.** How does the chemistry of groundwater and surface water flowing from ash placement areas compare with normal waters and with drinking water and other standards?
- **Comparison of environmental effects vs. coal refuse sites.** How does the groundwater adjacent to the ash placement site compare with water flowing from the coal refuse pile which was burned to generate the ash?

DOE Savannah River Facility

At sites such as the DOE Savannah River facility where Cherry et al. (1984) describe deleterious impacts on biota, the ash placement method definitely does not meet current regulations in Pennsylvania. For at least a year, the ash was discharged into a small pond, from which turbidity values and sediment character indicate that considerable fines overflowed into adjacent small streams and swamps (Cherry and Guthrie, 1977). It was from this ash-choked environment that Cherry et al. (1984) described deleterious effects on biota. The power plant in question burned coal with a high concentration of sulfur and heavy metals, as indicated by the experiments of Thomas et al. (2004) on highly contaminated runoff from the coal pile at the plant. This runoff had pH 2.4 and contained 142 mg/l Fe, 84 mg/L Al, 29 µg/L As, 6 Cd, µg/L 383 µg/L Cu, 356 µg/L Co, 716 µg/L Ni, and 1647 µg/L Zn. The extremely high concentrations of trace elements indicate the use of very pyritic and metal-enriched coal. The ash at sample sites in the drainage system averaged 1.7 ppm Cd, 38 ppm Cr, 6 ppm Zn, 81 ppm Cu, 2400 ppm Ti, 0.8 ppm Hg, 11 ppm Co, 20 ppm As, 96 ppm Mn and 6.1 ppm Se (Cherry and Guthrie, 1977). The pH averaged about 5.5 with excursions to 3.5. The stream was also affected by thermal pollution.

Glen Lyn Plant in Virginia

At the Glen Lyn plant in Virginia investigated by Specht et al. (1984), the ash was similarly discharged into a pond with overflow into a small stream and then into the New River less than 100 meters away. The pond was nearly full during some periods, and suspended sediments in the effluent ranged from 35 to 60 mg/L for a period of about 9 months during the study.

Neither of these ash disposal procedures would be acceptable according to current regulations in Pennsylvania. Given the combination of metal-enriched coal and high suspended particulates in the pond overflows, it is not surprising that biota were affected. Other proposed damage sites violate other of the criteria for safe placement of ash.

Trace Elements in CFBC Ash versus Common Rocks and Soils

Today, I would like to present information on concentrations of trace elements in CFBC ash as compared to common rocks and soils, and on trace elements in acid drainage from coal refuse compared to water associated with ash placement areas. The CFBC ash in question is formed by burning coal refuse composed of approximately half coal and half rock, mixed with limestone to capture sulfur, in a fluidized bed at significantly lower temperatures than conventional pulverized coal plants.

In evaluating any chemical hazard, it is important to compare samples of the supposed hazardous material with concentrations of the hazardous chemical in natural environments. A professor of mine once said that the First Law of Geochemistry is that "There is a little bit of everything in everything." That is, with a sensitive enough analytical method, any hazardous element or compound can be found in any material.

In this regard, it is relevant to compare levels of arsenic, cadmium, chromium, lead, mercury, nickel and selenium in CFBC ash with concentrations in shale and soils. Shale is the most common type of sedimentary rock. Many shales are "black shales," which contain above average amounts of carbon of organic origin. Black shales are reasonably common, occurring in many parts of Pennsylvania. Table 1 lists average values commonly found in shale, black shale, soil, and ash from CFBC plants in Pennsylvania. Sources for this data are Marshal and Fairbridge (1999), Wedepohl (1969), Turekian (1977), Krauskopf (1967), Vine and Tourtelot (1970), Quinby-Hunt et al. (1988), Connor and Shacklett (1975), Rose (2003), Rose et al. (1978), and Noll et al. (2000). Values for CFBC ash are given for both the median and the mean.

As seen in the table, concentrations in CFBC ash from plants burning coal refuse are similar to common rocks and soils in concentration of these elements. Values are less than or equal to the black shale value, except perhaps for mercury, for which no valid black shale value is established. Note that nearly all mercury at CFBC plants is retained in ash rather than being vaporized (Noll et al., 2000).

Table 1. Mean concentration of some elements in common rocks, soils and CFBC ash (in ppm).

	Arsenic	Cadmium	Chromium	Lead	Mercury	Nickel	Selenium
Shale	12	0.3	90	25	0.18	68	0.6
Black shale	29	1	100	50	>0.2?	50	5.6
Soil	7.5	0.5	43	17	.06	17	0.3
Soil (E PA)	18	1.5	42	11	.06	31	12
CFBC ash med., mean	18, 36	0.5, 1	33, 36	26, 27	0.3, 0.4	22, 29	2.5, 4

Based on this data, CFBC ash is not unusual in its content of these elements. It is no more hazardous than common rocks.

Another important comparison is between water flowing from coal refuse piles that furnish the fuel for CFBC plants vs. water flowing from ash placement areas. Table 2 shows data on water from monitoring wells down gradient from refuse vs. wells down gradient from CFBC ash, using all relevant data from Noll et al. (2000). In terms of number of samples, there are 125 samples down gradient from refuse, and 35 samples down gradient from ash. Note that many samples in Noll et al. (2000) are down gradient from both ash and refuse.

Table 2. Average concentrations of elements in water from monitoring wells at ash placement sites (in micrograms/liter)

	Arsenic	Cadmium	Chromium	Lead	Mercury	Nickel	Selenium
DGRefuse	176	41	106	102	1.8	638	41
DGAsh	9	3.9*	35	25	0.7	119	12
MCL	10	5	100	(15 ⁺)	2		50

DG = Down gradient, MCL = Maximum concentration limit (EPA drinking water). *Not detected taken as 2.5 ug/l = 50% of typical detection limit. ⁺No MCL for lead, only a guideline.

This table leads to two clear conclusions. First, the coal refuse piles are generating acid mine drainage with high concentrations of these elements. The above concentrations for the refuse sites are similar to 128 analyses of acid mine drainage from coal mines in eastern US compiled by Hyman and Watzlaf (1997). The refuse piles are therefore sources of severe and potentially hazardous water contamination, just like the acid drainage flowing from abandoned mines.

In contrast, the waters from monitoring wells down gradient from ash are uniformly much lower in concentration than the refuse waters, and are lower than the drinking water standards. Based on this data, the burning of the coal refuse in CFBC plants not only produces electricity and heat, but also removes a major source of water contamination. Unsightly piles are removed, and the ash furnishes alkalinity to neutralize any nearby acid drainage from other sources.

Another relevant type of information is in the form of trace elements in the ash, as determined by sequential selective extractions. Rose et al. (2001) and Noll et al. (2000) show that the trace elements are very firmly bonded in CFBC ash. Even pH 3 sulfuric acid, similar to some acid mine drainage, leached only a small proportion of the various elements from ash. We interpret the trace elements to be incorporated in aluminosilicates formed during the relatively lengthy circulation of ash in the circulating fluidized bed. These aluminosilicates and their contained trace elements are resistant to attack by natural waters.

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Dr. Arthur W. Rose is Professor Emeritus of Geochemistry in the Department of Geosciences, Penn State University. He has experience in a wide range of problems involving geochemistry of natural and contaminated environments, especially on trace element behavior. Currently he continues research on passive treatment of acid mine drainage, and consults on problems related to coal, coal ash, landfills, and groundwater. From 1964-67 he mapped geology and geochemistry for the State of Alaska before coming to Penn State in 1967. At Penn State he taught exploration and environmental geochemistry, environmental geology, and economic geology, and conducted research on Pennsylvania mineral deposits including uranium, copper, zinc, and chromium. Other major research involved oil well brines, radon and uranium decay products in soils, and acid mine drainage. He received a PhD in Geology and Geochemistry from the California Institute of Technology in 1958 and joined Kennecott Copper Co as an exploration geologist in Arizona and later as research geochemist investigating trace element and mineral halos around porphyry copper deposits.

A BRIEF HISTORY OF THE DOE SAVANNAH RIVER SITE

Dr. Barry Scheetz
Pennsylvania State University
University Park, Pennsylvania

I would like to take this time to supplement the excellent presentation of Dr. Hopkins by placing Area D into historical perspective. The Savannah River Site [SRS] is a governmental facility that produced fissile materials for nuclear weapons from the early 1950's through the 1990's.

This is a very unique site and I would like to take a few moments to attempt to put into perspective the nature of the secrecy under which the SRS operated and under which this electrical generation facility was constructed and operated.

Only 18 months after Enrico Fermi demonstrated on the campus of the University of Chicago that a nuclear chain reaction could be sustained, B reactor at Hanford, Washington went into service containing 70,000 enriched uranium fuel elements. These fuels produced the second of the bombs used against Japan in 1945. The first bomb was a uranium device and the second bomb was a plutonium device. The construction of the Hanford site during the second world war was the largest construction project ever undertaken in human history and it was all conceived and executed in the most extreme of secret conditions.

After the war in 1946, the Smyth Commission released details of the program with enough information that the Russian government was able to take what understanding they had received from spies [remember the Rosenbergs were executed for releasing U.S. government secrets relating to the nuclear program to Russia] and initiate a plutonium production program that culminated in the detonation of their first atomic bomb in 1949.

SRS was constructed in response to this event, which marked the onset of the "Cold War." SRS was a second generation, weapons-grade production facility that produced primarily plutonium and tritium. The secrecy that surrounded Hanford also surrounded SRS. It was decided to build the facility away from the Hanford site to ensure that a single attack would not compromise both facilities. An area was needed that had ample cooling water for the nuclear production reactors, a low population density and availability of ample electrical energy. The Site was selected along the Savannah River for these reasons and the only small rural farming community in the area appropriated along with 310 square miles and all of the residents moved.

This was a time of grave public concern. As children in elementary school, we practiced ducking under our desks in the class room and covering our heads to protect ourselves from the nuclear blast. It was a time of public fall out shelters with the large supply of yellow and red barrels containing water. It was also the time that many American citizens constructed their own family fall out shelter and were concerned about how to exclude their neighbors because of limited food supplies in the event they ever had to use the shelter.

The philosophy toward the environment at the time was much different than today's. The production of plutonium was THE ONLY objective. And this objective was to be accomplished in complete secrecy. SRS operated to produce plutonium from nuclear fission and to extract it from the fuel elements. For every atom of fissile plutonium produced, one million atoms of fuel element atoms had to be processed. In all, SRS was responsible for the production of nearly 47 tons of plutonium – all in the strictest of secrecy. The large site was used to dilute the environmental release of both industrial chemicals and radioactive materials. Point source discharge was not a consideration, only discharge at the boundary of the 310 square mile site was considered. *Dilution was the Solution to Pollution* under this philosophy. I don't have figures for releases to the Savannah River but as an example, Hanford released 110 million curies of radioactive arsenic, manganese, sodium and zinc to the Columbia River. This quantity of radioactivity amounts to about one third of all activity produced on the site.

The definition of 'coal ash' was not the same definition that we are familiar with today. The definition that the facility operated under in the 1950's through the 1970's, when it ceased operations, was all encompassing. All waste materials that were generated at the facility including: fly ash, bottom ash, boiler scale solutions, cooling tower blow downs, solvents, grease, and even coal that was purchased but did not meet the rudimentary specifications of the time, end up in the Area D pond. The disposal of off-spec coal into this site warrants a few

words. It must be kept in mind, from a historical perspective, that coals were not washed or heavy-media treated to remove acid producing components. There were no emission regulations: consequently, the coals had the characteristics of producing significant acid drainage, which was also diverted into the Area D impoundment. Furthermore, SRS was constructed in 1951. The electrical generation facility was likely based on proven technology available to the contractor, which means 1930's vintage technologies.

Area D can serve as a good example of what might be expected to be the consequences of sub-aqueous disposal of a mixture of materials, most of which we have no idea as to the chemical and mineralogical nature nor to the amount, with the exception that they were all products of the facility and that can certainly be called into question as well. It can serve as a lesson based not on today's technology but on a **75 year old** technology. It can serve as a lesson based on an environmental philosophy that typifies the operation of a super secret national defense site that is **50 years old**. Furthermore, Area D, like the rest of the SRS site, is in the process of being environmentally remediated and it is a tribute to the current Federal administration that is opening up our nations deepest and darkest secrets for international scrutiny and as an environmental laboratory to study the consequences of these dark and desperate times.

After examining the background of the Area D, its history and the philosophy under which it operated, it is very clear that those groups opposed to the burning of coal-based fossil fuels and attempting to link Area D to damage cases based on present day technologies and environmental philosophies are misrepresenting the excellent work of Dr. Hopkins as well as deceiving the EPA and the public.

Dr. Barry E. Scheetz, Professor of Materials, Civil and Nuclear Engineering, is a faculty member at the Materials Research Institute at the Pennsylvania State University. His research deals with the chemistry of cementitious systems including environmental remediation by the use of industrial by-products, focusing on large-volume fly ash-based cementitious grouts. The program activities approach the utilization of these by-products from a materials aspect; that is, by knowing the mineralogy and the bulk chemistry of the by-product; hydrated end products with specific characteristics can be engineered. Field activities have focused on the long-term elimination of acid mine drainage and the elimination of public health and safety issues associated with abandoned mines. He is a member of The President's US-INDO Energy Consultation serving in the Coal Advisory Group. He also has been active in the Acid Drainage Technology Initiative where he serves as the program chairman of the Avoidance and Remediation Committee for the Coal Sector. He has also served on several National Research Council committees reviewing nuclear waste management activities at various National Laboratories. He received an M.S. in 1972 in Geochemistry from The Pennsylvania State University followed by a Ph.D. in Geochemistry and Mineralogy in 1976, also from The Pennsylvania State University. He has been an active research faculty member for 28 years.

Session 4

RESPONSE TO RECENT PUBLIC CONCERNS ABOUT CCB PLACEMENT AT MINE SITES IN PENNSYLVANIA

Session Chairpersons:
Billie Ramsey and Dave Martin
Anthracite Region Independent Power Producers Association
Camp Hill, Pennsylvania

Losing the Opportunity to Beneficially Use CCPs when Policy is not Based on Site-Specific Conditions or Current Management Practices

Stephen B. Dixon, National Gas & Energy Transmission, Bethesda, Maryland

Pennsylvania's Abandoned Mine Land Hazards

Larry LaBuz, Electric Power Generation Association, Harrisburg, Pennsylvania

A Response to Allegations of Damage to Water Quality Due to Placement of Fluidized Bed Combustion Material at Abandoned Mine Sites in Pennsylvania

Dennis Noll, P.G., Earthtech, Johnstown, Pennsylvania

CFBC Ash and Monitoring Well Chemistry Compared with Normal Rocks, Soils, and Waters

Arthur W. Rose, Professor Emeritus of Geochemistry, Penn State University, University Park, Pennsylvania

A Brief History of the Doe Savannah River Site

Dr. Barry Scheetz, Pennsylvania State University, University Park, Pennsylvania

WHERE DO WE GO FROM HERE? FORUM PARTICIPANT RECOMMENDATIONS

At the conclusion of the forum on May 6, 2004, the participants provided the following recommendations concerning issues or concerns deserving attention and efforts by the Coal Combustion By-Products Steering Committee.

1. There needs to be more effort and research into how we test these materials to determine if placement in a particular hydrogeologic setting is appropriate.
2. There needs to be a closer look at what constitutes a damage case or negative impact.
3. There needs to be a CCB data repository (both monitoring data and research findings) so that people can access what has been done and also to avoid having the same work done more than once.

SURVEY RESULTS
STATE REGULATION OF CCB PLACEMENT AT MINE SITES:
A TECHNICAL INTERACTIVE FORUM
PARTICIPANT COMMENTS AND RECOMMENDATIONS

CATEGORY OF PARTICIPANTS	# OF REGISTRANTS	% OF REGISTRANTS
TOTAL REGISTRATION	175	
TOTAL COMPLETING THE SURVEY	61	100
LEVEL OF SATISFACTION WITH THE FORUM		
EXTREMELY SATISFIED	31	53
VERY SATISFIED	25	42
SATISFIED	2	3
DISSATISFIED	1	2
VERY DISSATISFIED	0	0
LEVEL OF SATISFACTION WITH THE FIELD TRIP		
EXTREMELY SATISFIED	35	70
VERY SATISFIED	15	30
SATISFIED	0	0
DISSATISFIED	0	0
VERY DISSATISFIED	0	0

COMPLIMENTS:

1. This was an informative well organized forum!
2. Great job!
3. This has been an enjoyable, informative, and well run/organized forum that I would highly recommend to anyone!
4. Great field trip, well organized and outstanding support from host sites.
5. The conference was great! Very informative. I learned a lot.
6. Very good interactive forum! Great job! Great location!
7. Excellent conference!
8. The field trip was incredible!
9. Very comprehensive with excellent range of real life situations and topics. Appreciated field trip, damage cases, and many real life cases.
10. Excellent job pulling together a very complicated field trip!
11. Good job overall! A worthwhile conference. Time well spent!
12. Use of the speaker timer was excellent! A very disciplined approach was used to ensure no overruns on presentation time.
13. The field trip was well executed and very informative. It gave me a totally different perspective on ash utilization.
14. A very fine conference!

**WHERE DID THE PARTICIPANTS COME FROM?
AND
WHO DID THEY REPRESENT?**

PARTICIPANT AFFILIATION	# OF REGISTRANTS	% OF REGISTRANTS
State Agency	47	27
Electric Utility	36	21
Consultant	22	13
Mining	20	11
University	13	7
OSM	12	7
DOE	8	5
U.S EPA	4	2
Environmental Group	3	2
Geologic Survey	3	2
AML Group	3	2
CCB Organization	2	1
Railroad	1	0

REGIONAL REPRESENTATION	# OF	% OF
WEST	14	8
EAST	120	69
MID-CONTINENT	39	22
INTERNATIONAL	1	1

PARTICIPANT RATING ON USEFULNESS OF TALKS

4.0=EXCELLENT; 3.0=GOOD; 2.0=FAIR; 1.0=POOR

SESSION 1 FBC MATERIAL IN MINE RECLAMATION

<u>PRESENTER</u>	<u>AVERAGE RATING</u>	<u>RATING RANGE</u>
Francois Botha	3.5	4-2
Timothy Kania	3.4	4-2
Dennis Noll	2.3	4-1
Barry Scheetz	3.6	4-2
Caroline Loop	3.4	4-2
Stephen Skidd	2.6	4-1
OVERALL SESSION 1 AVERAGE	3.1	

SESSION 2 STATE PROGRAM REGULATION OF CCBs AT MINE SITE: CASE STUDIES

<u>PRESENTER</u>	<u>AVERAGE RATING</u>	<u>RATING RANGE</u>
James Ashby	2.8	4-2
Steve Potter	2.9	4-2
Andrew Voros	3.4	4-2
Bret Sholar	2.6	4-2
Stephen Tillotson	3.1	4-2
Ishwar Murarka	3.4	4-2
OVERALL SESSION 2 AVERAGE	3.0	

SESSION 3 ENVIRONMENTAL DAMAGE CASES

<u>PRESENTER</u>	<u>AVERAGE RATING</u>	<u>RATING RANGE</u>
Robert Williams	3.3	4-1
Phillip Fauble & Michael Zillmer	2.8	4-1
David Lang	2.7	4-1
Charles Norris	2.0	4-1
William Hopkins	3.5	4-1
OVERALL SESSION 3 AVERAGE	2.9	

SESSION 4 RESPONSE TO PUBLIC CONCERNS ABOUT CCB PLACEMENT AT MINE SITES IN PENNSYLVANIA

<u>PRESENTER</u>	<u>AVERAGE RATING</u>	<u>RATING RANGE</u>
Barry Scheetz	3.8	4-1
Art Rose	3.5	4-1
Dennis Noll	2.9	4-1
Stephen Skidd	3.1	4-1
Stephen Dixon	3.4	4-1
Larry Labuz	3.6	4-2
OVERALL SESSION 4 AVERAGE	3.4	

SUGGESTIONS FOR IMPROVEMENT

APPLICATIONS OF CCBS AT UNDERGROUND MINES

- I would like to hear more about placement of CCBs in underground mines.
- Need more on utilization of fly ash to fill mine voids for subsidence and water quality.
- Need more on the use of FBC ash to improve deep mine acid mine drainage.

CCB CHARACTERIZATION AND LEACHING

- Need a short overview of the chemistry of Boron, Arsenic, and sulfates and their relative mobility.
- Need more on coal ash testing criteria and the difference between AMD and ash leachate.
- Need more on fly ash chemistry and interactions.
- Need more on the toxicology of Boron.

ECONOMICS

- Would like to know about the economic feasibility of CCB placement.

REGULATORY

- Need more on regulatory status.
- Need a comparison of existing State regulations on CCB placement at mine sites.
- Need more on the permitting decision making process.

OTHER

- Have David Hassett hold a workshop on good disposal practices.
- Have the speakers provide more in depth abstracts.
- Would like to have a hard copy of speaker talks.

- Would like to hear about the legal aspects of CCB placement from attorneys.
- All speakers should be required to use power point. Poor overheads were hard to read and understand.
- Would like to see more State specific case studies.
- Would like to see additional studies on biotic communities.
- Need more western U.S. region case studies that differentiate between disposal and reuse.
- Need more presentations on ash disposal in sand and gravel mined areas.
- Need to discuss more recent damage cases.
- Need more studies on the effect of CCB placement on wildlife.
- Use predetermined questions for panel discussions.

APPENDIX I: RECORDED DISCUSSIONS

Edited by
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The following are the edited discussions that took place at the end of each speaker presentation and at the end of each topic session. The actual comments have been edited to translate the verbal discussion into a format that more effectively and efficiently communicates the information exchange into a written format. The organization of the discussion follows the same progression as that which took place at the forum. A topical outline has been developed to aid in accessing the information brought out in the discussions.

The topic of each question is shown in alphabetical order in **bold**. The individual speaker questions are listed in outline format under the appropriate topic session and presentation title. Questions during the interactive discussions are listed at the end of the session in the following format.

SESSION # AND TOPIC AREA

1. Presentation Title

- **Subject of Question or Comment**

SESSION #: INTERACTIVE DISCUSSION

Subject of Question or Comment

OUTLINE OF DISCUSSION TOPICS

SESSION 1: FLUIDIZED BED COMBUSTION MATERIAL AND MINE RECLAMATION

1. Fluidized Bed Combustion Material Overview
 - **Burning Tires**
 - **Loss on Ignition**
2. Remining and Reclamation of Two Large Bituminous Waste Coal Piles Using FBC Ash: A Watershed Restoration Perspective
 - **Effect of Removing Coarse Refuse Fragments**
 - **Land Use Objectives for Reclaimed Land**
 - **Total Solids Analysis on Trace Metals**
 - **Total Solids Leachate Analysis**
3. 2003 Update—Occurrence and Fate of Selected Trace Elements in Circulating Fluidized Bed Combustion By-Products
 - **Source of Water Samples**
4. The Chemistry and Mineralogy of Coal Derived Fly Ash: A Basis for Responsible Reuse
 - **Silica Dioxide versus Quartz**
5. Lessons Learned from Full-scale, Non-Traditional Placement of Fly Ash
 - **Ettringite Formation**
 - **Location of Water Table**
6. Fuel and Ash Management for a Typical CFBC Facility
 - **Short Term Leaching**

SESSION 1 INTERACTIVE DISCUSSION

Beneficial Use Debate

Community Values

SESSION 2: STATE PROGRAM REGULATION OF CCB'S AT MINE SITES: CASE STUDIES

1. Permitting Issues Associated with wet FGD Placement at the Mettiki Mine in Maryland
2. Regulatory Hurdles Associated with Mine Placement of Coal Combustion Materials at Hanson Aggregates Inc., Clarendon Quarry, Clarendon, New York
 - **Variety of Ash Sources**
3. Beneficial Use of Dredged Materials Amended with Coal Fly Ash for Abandoned Mine Reclamation in Pennsylvania

- **Chloride**
 - **Dredging for Shipping**
4. Alternative Methods of Ash Placement at a Non Coal Mine Site in Oklahoma
 - **Chloride and Selenium Trends**
 - **Type of Ash**
 5. Historical Perspective of Coal Waste Management at Dakota Gasification Company, Antelope Valley Power Station and Coteau Properties Mine, Beulah, North Dakota
 - **Commercial Gasification**
 6. Universal Mine Fill Site in Indiana: A Case Study of Fly Ash Utilization
 - **Arsenic Absorption**

SESSION 2 INTERACTIVE DISCUSSION

Dredge Material Fixation
Oklahoma Power Plant Fuel
Use of Foundry Sand

SESSION 3: ENVIRONMENTAL DAMAGE CASES

1. Chisman Creek Superfund Site: A Retrospective Review
 - **Current Disposal Environment**
 - **Lessons Learned**
 - **Placement Recommendations**
 - **Superfund Listing**
2. Ground Water Impacts from Coal Combustion Ash Disposal Sites in Wisconsin
 - **Ash Type**
 - **Embankment Standards**
 - **Reuse of Coal Ash**
 - **Water Analysis**
3. Leaching of Arsenic and Other Metals from a Fly Ash Dump in Beverly, Massachusetts
 - **Mixed Wastes**
 - **Solid Waste Exemption**
4. Environmental Impacts of Power Plant Waste Placement in Mines
5. Impacts of Aquatic Disposal of CCBs on vertebrates on the Savannah River Site
 - **Organisms Found in Sediment Ponds**
 - **Synergy and Selenium**

SESSION 3 INTERACTIVE DISCUSSION

SESSION 4: RESPONSE TO RECENT PUBLIC CONCERNS ABOUT CCB PLACEMENT AT MINE SITES IN PENNSYLVANIA

SESSION 4 INTERACTIVE DISCUSSION

Backfilling Room & Pillar Mines
Blending FBC with Coal Waste
Coal Waste Impurities
Crop Fall Problems
Evolution of Coal Ash Placement
FBC Process Influence on Leachability
Leachate Tests in Alkaline Conditions
Measuring Success at Treating AMD
Subsidence

DISCUSSION BY SESSION

SESSION 1: FLUIDIZED BED COMBUSTION MATERIAL AND MINE RECLAMATION

1. Fluidized Bed Combustion Material Overview *Dr. Francois Botha, Illinois Clean Coal Institute, Carterville, Illinois*

Question: (Burning Tires) You talked about the burning of tires. What is the size the tires are shredded to and what do you do with the wire?

Answer: It is my understanding that they chop the tires to about the same size as the coal (about ¼ inch or less by 0). The tire wire comes out of the ash as wire and is separated magnetically and is sold as scrap.

Question: (Loss on Ignition) You showed a relative high loss on ignition. I don't think the plants here in Pennsylvania see the high values that you are recording. Can you explain this?

Answer: It is probably related to the type of coal that you are burning. I believe that the Pennsylvania Anthracite coal burns much cleaner than what we burn in Illinois therefore you will get lower carbon content.

2. Remining and Reclamation of Two Large Bituminous Waste Coal Piles Using FBC Ash: A Watershed Restoration Perspective *Timothy Kania, Pennsylvania DEP, Ebensburg, Pennsylvania*

Question: (Effect of Removing Coarse Refuse Fragments) Would removing the coarse fraction of the refuse, by either burning it or removing it from the site, have resulted in better water quality?

We have found that encapsulating the coarse refuse in the ash has been pretty effective in isolating it from the environment and aiding in water quality improvement at the site.

Question: (Land Use Objectives for Reclaimed Land) What was the long term land use objective for this reclaimed area?

The overall land use objective was to improve the area by eliminating the erosion, vegetating it, and returning it to a wildlife habitat that would not be developed.

Question: (Total Solids Analysis on Trace Metals) Is it possible that when you did your monitoring well samples early on, that you got your high values because you may have been disturbing a lot of suspended material that may have contained a lot of trace metals versus what was dissolved in the water and leaching?

Answer: No. I don't believe so. Where we have compared our dissolved solids data to our total solids data, we found a big influence on iron and aluminum but not on trace metals.

Question: (Total Solids Leachate Analysis) Was the analysis done on dissolved solids or total solids? Did you use filtered or unfiltered water in your analysis?

Answer: Actually we did some of both. The data I have used in this talk were by total analysis.

3. 2003 Update—Occurrence and Fate of Selected Trace Elements in Circulating Fluidized Bed Combustion By-Products *Dennis Noll, Earthtech, Johnstown, Pennsylvania*

Question: (Source of Water Samples) Did the sampling represent both ground and surface water sampling?

Answer: It was mostly groundwater samples although there were some seeps and some deep mine discharges.

4. The Chemistry and Mineralogy of Coal Derived Fly Ash: A Basis for Responsible Reuse *Dr. Barry Scheetz, Pennsylvania State University, University Park, Pennsylvania*

Comment: (Silica Dioxide versus Quartz) In order for quartz to make it through the combustion process, it must be quartz to begin with. In the elemental analysis of ash, silica is reported as silica dioxide but silica dioxide is not necessarily quartz.

Answer: Yes. That is why you should do a mineralogical analysis. This is why you need to treat fly ash as a material. A bulk chemical analysis is just that, it is not a mineralogical analysis. A lot of people will look at the chemical analysis and see SiO₂ at 50% and say this is a carcinogen, but that would be wrong because it is probably

tied up in the metaclay or it is tied up in the glass. This is why you need to do a quantitative x-ray diffraction analysis.

5. Lessons Learned from Full-scale, Non-Traditional Placement of Fly Ash *Dr. Caroline Loop, Pennsylvania State University, University Park, Pennsylvania*

Question: (Ettringite Formation) What is the pH range that needs to be in the pore space for the ettringite reactions to continue?

Answer: You need a pH of at least above 9. The ettringite itself is between 10.5 and 11.5. The calcium oxide has a maximum pH of 12.45. We have not seen the water higher than 12.1 to 12.2.

Question: (Location of Water Table) What is the water table in the ash of the Big Gorilla?

Answer: The water table where we drilled the well was about 17 feet from the top of the lower platform. It is approximately the same place it was prior to ash placement. Initially, we thought placement of the ash in the water would cause the water to fill up and discharge but we have not seen that. We think that part of this water is being taken up by the ash and part is moving out into the culm and being reacidified.

6. Fuel and Ash Management for a Typical CFBC Facility *Stephen Skidd, Cambria CoGen, Evansburg, Pennsylvania*

Comment: (Short Term Leaching) The selenium and possibly chromium results that you have are probably very misleading. You can not do a short term leaching test on CFBC ash.

SESSION 1 INTERACTIVE DISCUSSION

Question: (Beneficial Use Debate) How can we determine the main outstanding questions concerning the beneficial use of coal ash and answer those questions in a definitive scientific way?

Answer: In Illinois, we have been trying to find applications that always work. This has not been successful. Instead we need to pick one project at a time and obtain the best data available. We now have enough small projects that have been completed successfully that we need to develop some sort of Internet based database so that people can access this information and determine what projects have worked successfully and how they were accomplished. This would be the sort of thing that could be taken on by the American Coal Ash Association.

Question: (Community Values) What are the values of the pits to the communities? Are they used for fishing or anything else?

Answer: There are few pits in the Anthracite region that I am aware of that have fish in them. The Little Gorilla has fish in it and some other small isolated pits have fish. The vast majority of the 1,000 + pits that are out there have water that is too contaminated to support fishing. The Anthracite communities are in general very old communities. These people do not desire change and are content to live just as they always have including the waste culms and contaminated pits. This is even the case where people have drowned in the pits. The city of Minersville supplements their water supply with a mine pit lake that has low pH but is used in dry periods.

Answer: A big issue in Anthracite communities is the relationship between the residents and the local coal companies where there may be long standing animosity. Conducting projects in these areas requires getting the confidence of the residents up front.

SESSION 2: STATE PROGRAM REGULATION OF CCB'S AT MINE SITES: CASE STUDIES

1. Permitting Issues Associated with wet FGD Placement at the Mettiki Mine in Maryland *James Ashby, Mettiki Coal, LLC, Oakland, Maryland*
2. Regulatory Hurdles Associated with Mine Placement of Coal Combustion Materials at Hanson Aggregates Inc.,

Clarendon Quarry, Clarendon, New York

Question: (Variety of Ash Sources) It is my understanding that you used a variety of ash sources. Did that have any influence on the performance?

Answer: We had criteria for ash quality and each truck load of ash was identified and sampled prior to use. It was all coal boiler fly ash just from different sources.

3. Beneficial Use of Dredged Materials Amended with Coal Fly Ash for Abandoned Mine Reclamation in Pennsylvania *Andrew Voros, NY/NJ Clean Ocean and Shore Trust (OCEAN), New York, New York*

Question: (Chloride) You seemed to have a lot of chloride. Did you have something to balance that?

Answer: The chloride did balance with other ions. It clearly originated from salt water. Municipal waste incinerator ash can have huge amounts of sodium chloride much more than salt water. Fresh water dredging would not have a chloride issue. Pennsylvania DEP determined not to use municipal waste incinerator ash because of the chloride as a result of the findings of this project.

Question: (Dredging for Shipping) Due to the changing of the size of container ships, do you also anticipate dredging in the Great Lakes?

Answer: I am not sure that you have container ships in the Great Lakes. I can tell that the work horse of the Great Lakes is a 1,000 foot long work barge. For every inch of draft they do not have, they must decrease their load by 250 tons. This gives you an idea about how a few inches can make a huge difference. It is my assumption that everywhere dredging will get deeper than it has been getting.

4. Alternative Methods of Ash Placement at a Non Coal Mine Site in Oklahoma *Bret Sholar, Oklahoma Department of Mines, Oklahoma City, Oklahoma*

Question: (Chloride and Selenium Trends) What sort of selenium and chloride trends have you seen downstream of the major sites?

Answer: We look at a complete suite of trace metals and to date we have not seen any spikes of any parameters that would be a problem.

Question: (Type of Ash) Is this a Class C ash from pulverised coal?

Answer: Yes.

5. Historical Perspective of Coal Waste Management at Dakota Gasification Company, Antelope Valley Power Station and Coteau Properties Mine, Beulah, North Dakota *Steve Tillotson, North Dakota Department of Health, Bismark, North Dakota*

Comment: (Commercial Gasification) As a point of clarification, there are two commercial scale gasification plants that send power to the grid. There are several others that do not send power to the grid but commercially manufacture chemicals.

6. Universal Mine Fill Site in Indiana: A Case Study of Fly Ash Utilization *Ishwar Murarka, Ish, Inc., Sunneyvale, California*

Question: (Arsenic Absorption) Concerning the soil material where you determined the arsenic absorption coefficient, was that representative of the soils in the area of the pit?

Answer: I ran this on eight soils samples, I just showed you one. I am fairly confident that the iron content is the real driver. Any time your iron content is low your Kd is going to go lower. Aluminum oxide seems to have some role but I have not sorted that out yet.

SESSION 2 INTERACTIVE DISCUSSION

Question: (Dredge Material Fixation) Why do you add the lime and fly ash to the dredge material?

Answer: It is because we are using mildly contaminated dredge material. A significant portion of dredge material contains traces of contaminants that are way below hazardous levels but above the detection limit. The main issue about disposal of dredge material is that when you dispose of it in a marine environment or other water environment you fill the water with extremely fine particles that are consumed by aquatic organisms. This leads to the potential for bioaccumulation over time that will eventually get into fisheries for human consumption. In the effort to ban ocean disposal, the material has been vilified, in spite of the fact that much of it could be used as soil amendments. Given this history, it was determined that this type of solidification and stabilization of the material would sequester any of the organic contaminants.

Answer: In Pennsylvania, it is not allowable to fill open pits as a beneficial use unless it is with fly ash or blended fly ash. It is a possibility in the future that fly ash and dredge material could be blended at a different ratio and used as soil materials.

Question: (Oklahoma Power Plant Fuel) Could you tell us what type of fuel the power plant in Oklahoma was using and was it a fluidized bed plant or a PC boiler?

Answer: It is a PC boiler using pulverized coal. It is the only power plant in Oklahoma that burns Oklahoma coal.

Question: (Use of Foundry Sand) Concerning the New York project, at the first project at Clarendon they proposed to use a mixture of coal ash, foundry sand, and glass. The project that was actually initiated only involved coal ash. Why were the other constituents dropped from the mix?

Answer: I think they had some problems getting a consistent material from the foundry sand. It was a physical problem of just not having the material available when needed.

SESSION 3: ENVIRONMENTAL DAMAGE CASES

1. Chisman Creek Superfund Site: A Retrospective Review *Robert J. Williams, Dominion Resources Services, Inc., Glen Allen, Virginia*

Question: (Current Disposal Environment) Would this disposal site be allowed to be permitted under the current Virginia Solid Waste regulatory program?

Answer: No, it would not be.

Question: (Lessons Learned) What were the lessons learned from this experience?

Answer: First, keep your site off of the superfund list. Second, work with the community. Stay engaged with the project. Put a face on your work. I spent a lot of time talking to people one on one in their living rooms. Stay engaged with the agencies.

Question: (Placement Recommendations) Are you saying that we would never want to place ash in similar situations such as a very porous material (i.e. sand and gravel mine) or in contact with water?

Answer: I wouldn't want to say never. What we need to make sure of is that the material we are using is compatible with the geology of the site.

Question: (Superfund Listing) Why was this site chosen for the National Priority List?

Answer: The process for getting a site on the superfund list is a model used by EPA called the hazard ranking system. It is a scoring system based on the release of contaminants into the environment, ground water, surface water, proximity of the site to wetlands, endangered species, etc. You have to score above 28.5 points and Chisolm

Creek scored just above that.

2. Ground Water Impacts from Coal Combustion Ash Disposal Sites in Wisconsin *Philip Fauble (Madison) and Michael W. Zillmer (Milwaukee), Wisconsin Department of Natural Resources*

Question: (Ash Type) What type of ash was used in this project?

Answer: Type F ash.

Question: (Embankment Standards) Are there standards in place for the embankment projects to ensure the safety of the ground water?

Answer: Yes. We require embankments to be lined on the sides, bottom, and top.

Question: (Reuse of Coal Ash) What is the most commonly reused coal ash in Wisconsin?

Answer: Most is used as a concrete additive. Some is being disposed. Some are being used in DOT embankment projects.

Question: (Water Analysis) Were the water analysis done on the basis of total or dissolved metals?

Answer: We used totals from the 1970's to the early 1980's. Around the mid 1980's we switched to using dissolved water analysis.

3. Leaching of Arsenic and Other Metals from a Fly Ash Dump in Beverly, Massachusetts *David Lang, Ground Water Consultants, Inc. and Jan R. Schlichtmann, Beverley, Massachusetts*

Question: (Mixed Wastes) Ash Disposal began in the 50's and ceased in 1971. Were other wastes brought to the site?

Answer: There are mixed wastes at this site. Car bodies, white goods, and other unidentified piles of material. There could be industrial waste here as well but the present testing does not show a solvent problem.

Comment: (Solid Waste Exemption) Coal Ash in Massachusetts is exempt from the State Hazardous Waste regulations. It is only exempt from Solid Waste regulations if it is spread on the land or land filled. The exemption is applied for structural fill.

4. Environmental Impacts of Power Plant Waste Placement in Mines *Charles Norris, Geo/Hydro Inc., Denver, Colorado*
5. Impacts of Aquatic Disposal of CCBs on vertebrates on the Savannah River Site *Dr. William Hopkins, University of Georgia, Savannah River Ecology Laboratory, Aiken, South Carolina*

Question: (Organisms Found in Sediment Ponds) Are you working with organisms found in the sediment ponds themselves or those just exposed to the discharge but outside the ponds?

Answer: We obtain the organisms from the sediment ponds. A point I have been trying to make is that these ponds actually attract wildlife. We see that 10s of thousands of amphibians come into and breed in these ponds annually but we do not know how this impacts the larger populations outside the ponds. You may be creating a problem by attracting organisms away from more viable uncontaminated habitats.

Question: (Synergy and Selenium) You mentioned that one of the potential contaminants was selenium. Do you think there is a synergy between the trace elements you looked at? With respect to selenium are you in a normal nutrient range or above or below that?

Answer: In terms of a synergy, the possibility exists, but I have no idea where or not it does. We have 18 trace

elements in these systems that are elevated above background. A few are well above background such as cadmium, arsenic, and selenium. These elements appear to be in the potentially toxic range. These elements also appear to be accumulated in animal tissues. Vanadium has been mentioned today and I believe we need to take a closer look at the role of Vanadium. Sediment levels for selenium in the system run about 7 ppm on a dry mass basis which is above nutrient levels but do not even come close to the levels reported in western agricultural irrigation water. We do, however, see very high levels of selenium in the food chain. In various invertebrate species such as zooplankton, they have selenium concentrations up around 60 or 70 ppm. Some of the carnivorous salamander larvae rely on them for a food source and clearly they are getting a very high selenium level.

SESSION 3 INTERACTIVE DISCUSSION

Question: (Standards for Protecting Wildlife) Have you compared the elemental analysis data you are collecting with the U.S. or Canadian standards for protecting wildlife from contaminated water?

Answer: We have not looked at our data in terms of regulatory requirements. If you look at water concentrations at our site, they are not very impressive. What we are finding in most cases is that it is not the water concentration that is driving things, it is really trophic transfer. What is happening is that the sediments serve as a sink. Then you have benthic organisms and plants that take up a lot of these contaminants that are eaten by other organisms. It gets very difficult when you try to have sediment based criteria. Water criteria are easier to deal with but I don't think that water criteria necessarily give you a full picture and our site is a good example of that.

Question: (Cause of Biological Problems) Which specific elements do you think are causing the biological problems that you are observing?

Answer: This is really speculation, but I think that selenium is probably one of the most important. I say this based on tissue burdens that are accumulated. I am making the assumption that tissue burden is associated with effects. I think that cadmium, arsenic, and maybe vanadium contribute as well.

Question: (Relationship of Contaminants to Disease) Do you think that the effects of these contaminants increased the probability of disease or effect of parasites on these species?

Answer: I don't know the answer, but I have just started a new study to look at the relationship of these contaminants to fungal infections. We have not seen the type of deformities normally associated with parasitic infections.

Question: (Impact of Leachate) You presented information for the Universal Mine Site in Indiana where you compared the MW8 leachate data with the MW5 spoil well data. Have you considered that both the MW8 and MW5 indicate that the leachate did not migrate and impact the ground water in the spoil system for a variety of reasons and would you therefore interpret the data to indicate that the leachate in the ash management unit is not impacting the surrounding ground water?

Answer: The MW8 that I used as the spoil control relative to the fill control is not in the position that it should be in a migration path in any case. There is a negative impact to that site of the water quality in the fill area relative to water quality that one would expect there in that fill had it been the traditional spoil mass placed there. If that water is also migrating through the material and expanding the area of degraded water then that would be a negative impact.

Question: (Interstitial Water Quality) If I took the ash at Universal and placed it in an engineered landfill it would still have the same interstitial water quality that it had at the mine site. Are you suggesting that any water in contact with ash has to meet drinking water standards?

Answer: I have not said that it needs to be at drinking water quality. I am trying to propose that we look at it in comparison to other options. If that Universal pit had been reclaimed with spoil, it would not be at drinking water quality, which is clear from the spoil content. It is a negative impact that we have selected a fill that causes the water in that reclaimed area to be of lower quality that it would have been had we used a traditional technique. It may be that there are offsetting benefits of doing this but we need to be able to make a calculation to show that there

is a net benefit.

Comment: (SMCRA Requirement to Minimize Impact) I would like to point out that SMCRA requires that the impacts be minimized within the area itself. So that if you use a fill material that creates greater impacts than using the spoil material, then you are not minimizing the impact.

Comment: Concerning minimizing impacts under SMCRA, the Indiana courts ruled that Indiana was implementing SMCRA concerning ash placement correctly and this argument did not hold up.

Question: (Time Required for Water Monitoring) What should the criteria be in order to determine how long you should monitor water quality at a CCB mine placement site?

Answer: Based on the data I used, most of the sites had from 12 to 30 years of data and the water quality had not yet stabilized. I think a lot more work needs to go into what is happening with the chemistry at each of these sites.

Question: (Wisconsin Ash Quality) In Wisconsin, was your coal ash amended with calcium carbonate, and what was the ash pH?

Answer: Those with the greater impacts were filled in the 1960's and 70's and there is very little record of what actually happened. Since there is no coal in Wisconsin, the coal used reflected current market availability so we really don't even know the coal source. Concerning pH, it is really a non factor in the ground water. The buffering capacity of the soils and ground water in Wisconsin is so high that, shortly after contact, the water from the ash is back to neutral.

SESSION 4: RESPONSE TO RECENT PUBLIC CONCERNS ABOUT CCB PLACEMENT AT MINE SITES IN PENNSYLVANIA

SESSION 4 INTERACTIVE DISCUSSION

Question: (Backfilling Room & Pillar Mines) Is anyone in the U.S. backfilling room and pillar mines?

Answer: In the anthracite region of Pennsylvania, it was all deep mined with the room and pillar method with dips up to 90 degrees. They later robbed the pillars taking the last bit of coal out which incidentally killed a lot of miners. Later, when strip mining began, they went back in and stripped out the pillars closest to the surface.

Question: (Blending FBC with Coal Waste) Has anything been tried to blend coal waste material and the FBC material either underground or for stabilization?

Answer: Layer caking to encapsulate pyretic material is a very common practice.

Question: (Coal Waste Impurities) Is it true that waste coal has more impurities and potential contaminants than virgin coal?

Answer: I think this tends to be true in the sense that it contains more rock material than coal and it is the rock that contains most of the pyretic minerals.

Comment: (Crop Fall Problems) Crop falls represent two major problems. They represent a very significant public health and safety problem. The other feature we are trying to limit is due to the fact that all of the water that falls on the mountain ends up in the crop fall. It is then carried deep into the mine pool where it just creates more acid mine drainage. It ultimately ends up as contamination of the Delaware basin and estuary. Stabilization efforts that would seal crop falls would eliminate both hazards.

Comment: (Evolution of Coal Ash Placement) I think there has been an evolution in coal ash placement in Pennsylvania. Years ago, prior to beneficial use, most of the coal ash from the pulverized coal plants went into big slurry impoundments or large ash monofills on site. Then the co-gen plants came around and had to show about 25

years fuel supply and 25 years of ash storage in order to get financing. Now there has been an evolution to filling more and more mine land features. I think that ultimately we will even be able to fill crop falls as the technology progresses.

Question: (FBC Process Influence on Leachability) How would the FBC combustion process affect the ash constituents in terms of its potential leachability of trace elements?

Answer: Based on the data from our 2000 study, we have found that trace elements are strongly bound in the FBC material. This could be attributed to two factors. The temperature is not as high as it is with a PC boiler so that volatilization is reduced. For instance, Mercury does not go up the stack in an FBC unit. You are also forming very reactive dehydroxylated clays that bond with the trace elements that result in a form that is not readily leached.

Question: (Leachate Tests in Alkaline Conditions) Boron or selenium or other specific constituents can be mobile in a reducing environment. Most extraction procedures however, use a weak acid or a strong acid. Are there other methods that can give us a better estimate of a materials ability to leach in an anoxic environment?

Answer: The first of our leaching tests at Penn State was in distilled water so that the pH was determined by the material itself. The CFBC ash had a quite alkaline pH in the initial leach and even in the second stage when we were looking for exchangeable ions. This is an area that needs a lot more work. We need to figure out improvements on the leaching procedures in order to test all of the processes. Certainly all of the elements that exist as anions such as arsenic, boron, selenium are going to behave differently than cations.

Question: (Measuring Success at Treating AMD) Concerning the results of burning waste coal and reclaiming AML sites with FBC materials, do we have any numbers on the miles of streams where the water quality and biological productivity has improved and how much is still contaminated? Have you realized any benefits in public relations?

Answer: There are 4,200 miles of assessed streams that are impacted by acid mine drainage in Pennsylvania. There are a number of streams however, that have yet to be assessed. About 40 percent of the streams in Pennsylvania have not yet been assessed. Pennsylvania is second only to Alaska in the number of surface streams. It is hard to get a clear picture of this because the methods for measuring streams have been getting more accurate with time so that the total keeps getting higher even though we are reducing the impacts with better reclamation.

Question: (Subsidence) Is ash being used to backfill active mines in order to prevent future subsidence?

Answer: A mine in West Virginia tried to create a plug to seal of the mine from water but it did not work because of shrinkage. A longwall mine in western Pennsylvania wanted to be able to backfill with grout behind the long wall miner in order to have the structural support to maximize coal removal. Barry Scheetz developed a grout for this purpose but it has never been used.

Comment: Subsidence is a major problem in the anthracite and bituminous coal fields. The Pennsylvania DEP does a great deal of filling of these subsidences. They use an enormous amount of fly ash in their dry stoning of these subsidence areas. Our colleges at West Virginia University have some experience with super sonic dry stoning of fly ash into mines. The problem is that you never get a total seal because you will always have a gap around the top. Products are now in use where you can place a grout in flowing water. You haven't heard much here but I think you are going to hear more about this.

APPENDIX II

Editor's Note: The following paper was intended for Session 3 "Environmental Damage Cases" but none of the authors were able to attend to present the paper. Paper copies of the paper were made available at the forum for those attending. It is included in the appendix because there was no opportunity for the participants to comment on the paper or question the authors during the interactive discussion.

ECOTOXICOLOGICAL IMPACT OF COAL COMBUSTION BYPRODUCTS (CCBs) IN SOUTH CAROLINA/VIRGINIA STREAMS DURING THE 1970s TO EARLY 1980s

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Abstract

The toxic ramifications of coal ash effluent upon benthic macroinvertebrate assemblages in a stream/swamp drainage system (Savannah River Project [SRP], South Carolina) and mountain stream (Glen Lyn Plant [GLP], Virginia) were studied for 8 and 2 years, respectively, as both streams suffered from poorly managed settling ponds. At the SRP, aquatic biota were initially impacted by heavy ash overflow from the ponds into the receiving system, followed by acidic pH from fly ash addition, and high trace metal concentrations over 4 years until new holding ponds were developed. Ash siltation at the SRP significantly reduced all macroinvertebrate assemblages in the stream/swamp channels while acidic pH (mean change from 7.2 to 5.5 and extreme of 3.5) continued to impact the macroinvertebrates as well as the only remaining fish population (mosquitofish, *Gambusia affinis*) over 4 years of pond operation. Some trace metal concentrations (aluminum, cadmium, copper, selenium, zinc) were 1 to 2 orders of magnitude greater than US EPA Water Quality Criteria limits. The tolerant invertebrates and mosquitofish in the receiving system generally began to recover after 1-2 years when new, more efficient settling ponds were built.

At the GLP, aquatic biota was altered over several months as the pond reached ~97% filling capacity. The macroinvertebrate community in the mountain stream at the GLP was significantly reduced of mayflies, stoneflies and caddisflies while highly resistant coleopterans (*Psephenus herricki*) proliferated. Water quality was not substantially altered except for alkaline pH (mean of 8.5 over 2 months and extreme of 9.5). Trace metal concentrations were not substantially different between the upstream control and the downstream ash influenced sites. No laboratory acute toxicity of the ash effluent was detected but longer term chronic effects of ~28 days were suspected but not tested. Two to 10 months were required for most of the macroinvertebrate community to recover after correction of the ash pond effluent. Ash particle morphology and metal distribution analysis of the GLP ash, using electron microscopy and surface/subsurface ion microscopy, showed that metals could be either clumped or evenly distributed on fly ash particle surfaces. Surface enrichment on fly ash particles was found from electrostatic precipitators but not from heavy ash obtained from the plant furnaces.

Introduction

In the mid-1970's, a coal combustion byproducts (CCBs) holding pond system was allowed to fill and overflow into the stream/ swamp receiving system at the Savannah River Project (SRP) near Aiken, SC. Over a 12-year period, 30 + studies by D.S. Cherry, R.K. Guthrie, J. Cairns, Jr. and others were published describing the initial environmental impacts of acidic pH (3.5-5.5), followed by ash siltation/ filling into the stream/ swamp channels, and then elevated water column elemental concentrations resulting in bioaccumulation in the resident fauna (Cherry et al. 1984; Cherry and Cairns 1986). After cessation of CCB impacts by development of two vastly larger holding pond retention

systems, it required 1-2 years for ecological recovery of the fauna after 4 years of gross biotic impairment in the receiving system (Cherry and Cairns 1986, Cherry et al. 1984). The initial impact of the physical appearance and filling of the stream/swamp channels was annihilation of benthic macroinvertebrates and most fish except for mosquitofish (*Gambusia affinis*), although acidic pH caused a greater loss of biota in the water column downstream beyond the confines of the ash filling in the channels of the swamp.

After the initial 2-year slug of coal ash toxicity to the resident fauna, longer-term, chronic toxicity occurred as elevated trace element concentrations accumulated in the Great Savannah River Swamp sediment and ultimately into the recovering biota there (Cherry et al. 1976; 1979 a,b; Cherry and Guthrie 1977, 1978, 1979; Guthrie and Cherry 1979a,b; Guthrie et al. 1981,1983,1986).

The high surface area of fly ash particles with a negatively charged surface, along with the proximity of the ash in the electrostatic precipitation process of air stack emissions, allows the cooling ash particle surfaces to become coated with trace elements (Fisher and Natusch 1979; Theis and Wirth 1977; Natusch 1978; Natusch et al. 1975). Cherry et al. (1987) used electron microscopy and surface/subsurface analysis with ion microscopy and reported that fly ash particles became extremely enriched by trace elemental condensation on particle surfaces, such as from cadmium, copper, chromium, nickel, lead, mercury, titanium, arsenic, and selenium. Conversely, hotter bottom ash from the furnaces is not exposed to the electrostatic precipitation cooling process and has lower to no trace elemental coatings on the particle surfaces.

A second over-loaded fly ash settling pond scenario occurred in the latter 1970's-early 1980's at the Glen Lyn Plant (GLP), Virginia whereby a mountain stream was impacted for several months when the holding pond reached = 90% filling capacity (Specht et al. 1984; Cherry and Cairns 1986). The stream received a fly ash coating in the channel for a few months as alkaline pH of the effluent increased to pH 9.5. Although no acute toxicity of biota was observed in the stream, population shifts occurred in the benthic macroinvertebrate community as mayflies and predatory stoneflies were significantly reduced in numbers while tolerant beetles flourished. Some elevation of trace metals (cadmium, chromium, zinc, arsenic, selenium) occurred in the stream sediment, but 2 to 10 months after cessation of fly ash effluent into the stream, the macroinvertebrate recovery was actually happening and appeared to be nearly complete after 12 months.

When fly ash holding pond systems had been poorly operated in the mid-1970's to early 1980's, toxic consequences were high at the SRP in S.C. to chronically so at the GLP, Virginia. By comparing the environmental consequences of these two holding pond systems, studied by D. Cherry with others, various ecotoxicological ramifications can be ascertained from these two types of settling pond system operations. The objective was to compare the acute and chronic fly ash impacts from these two holding pond systems in SC and VA, where the principal investigator was actively involved at both sites. Also, the subtle consequences of trace elemental bioaccumulation can be evaluated from the severe situation at the SRP. From these two vastly different environmental scenarios, at least both of them were studied to the point where ecological recovery of the receiving system biota had occurred. Other CCB impact studies elsewhere during this era were not funded to the point of studying recovery in the damaged ecosystems. From this rudimentary origin ~30 years ago about CCB impacts, perhaps we can become better equipped to understand the current CCB concerns that occur today.

Materials and Methods

There was a vast number of research methodologies utilized in these two CCB-impacted sites, but it is unnecessary to describe them in detail because that exercise has been done in many publications by D. Cherry on the subject (please see Lit. Cited section). Sampling efforts included qualitative sampling of benthic macroinvertebrates by dipnets to seining of fish. Instrumentation used at Virginia Tech for the elemental bioaccumulation studies was neutron activation analysis (NAA) from the SRP samples to atomic absorption spectrophotometry (AAS) from the GLP ash. D. Cherry at the SRP initiated *in situ* toxicity testing in 1972 using stainless steel bioboxes, while acute toxicity bioassays of aquatic insects and fish were carried out at the Cherry lab in the early 1980's. Trace elemental methodology for electron microscopy and surface/subsurface ion microscopy analysis can be best documented in Cherry et al. (1987), for example, and the other literature citations found at the end of this paper. For the two site locations, a summary of holding pond conditions is available below.

Savannah River Project

In the SRP ash pond holding system, the following sequences or periods developed between November 1973 and February 1982 is found in Fig.1 by Cherry et al. 1984.

- (1) Period 1: November 1973 to June 1974: heavy ash input into the original dual small holding pond system as the ponds were filling.
- (2) Period 2: July 1974 to December 1974: heavy ash input into the original, dual holding pond system whereby ash overflowed the ponds into the stream/swamp receiving system that smothered most resident biota.
- (3) Period 3: January 1975 to September 1975: heavy ash input after ash excavation occurred from ponds with introduction of fly ash.
- (4) Period 4: October 1975 to December 1977: fly/heavy ash input with acidic pH effluent (pH = 3.5-5.4) into the receiving system.
- (5) Period 5: January 1978 to August 1979: fly/heavy ash input into a newly expanded primary/secondary holding pond system.
- (6) Period 6: September 1979 to February 1982: fly/heavy ash input into an additional secondary pond adjacent to the new ones developed previously.

Glen Lyn Plant-Adair Run Stream

At the GLP, fly ash effluent was released into a mountain stream (Adair Run), and the following sequences were noted by Specht 1985 and Specht et al. 1984 in Fig. 2. Note that the fly ash holding pond was a single, linear-type pond with the influent entering at one end and the effluent flowing over the skimmer wall at the opposite end. The effluent then traveled down a hill for ~30 m into the mountain stream.

- (1) Period 1: September 1979 through February 1980 with ash entering the pond (filling content not known).
- (2) Period 2: March through early June 1980 with ~87% of pond full with ash.
- (3) Period 3: Late June through early August 1980 as pond became ~97% full.
- (4) Period 4: Late August through October 1980, pond full and effluent input terminated into stream.
- (5) Period 5: November 1980 through June 1981, stream recovery.

Results and Discussion

Savannah River Project

Water Quality Changes

Major changes in temperature, turbidity, pH, and sulfate content occurred at the skimmer wall of the holding ponds and eventually into the stream/swamp receiving system from periods I through VI. Temperature increased in the holding pond effluent as sedimentation potential decreased from the slurry of heated coal ash. Turbidity up to 250 Judson Turbidity Units escalated substantially in Period II when heavy ash filled the ponds and spilled into the receiving system. For several months after acidic fly ash was added into the original pond system in Period IV, acidic pH in the pond effluent and receiving system was reduced to an average of 5.5 and an extreme of 3.5. Sulfate concentrations were low in Periods I and II (20 mg/L) and then steadily escalated to 80 mg/L through Period V. Turbidity and acidic pH in the holding ponds/ receiving system generally stabilized in Periods V-VI.

Biotic Changes

Density declines of benthic macroinvertebrates and the near-surface dwelling mosquitofish sampled reflected the

changes in the degrading water quality conditions of the settling ponds (Fig. 3). The harshness in the receiving ponds/receiving system was most dramatic during Period III when most insects, clams, and fish were eradicated from the system. During this period, ash turbulence from the ponds into the receiving system was at its peak while acidic pH was adding an additional stressor. Only chironomids were found in the stream/swamp receiving system during Period III at a low level of 4.0/m² while all other fauna were near or at 0/m². Even the mosquitofish tolerated the high turbidity in Period II, but with turbidity declining with advancing extreme acidic pH of 3.5 thereafter; even these tolerant fish were eradicated from the receiving system.

After the completion of the new, expanded ash basin in Period V, several tolerant insect groups (*Platthemis lydia*, *Libellula* spp, chironomids, *Peltodytes* spp., *Gammarus* spp., and *Physa* spp.) started to recover (Fig. 3). Other organisms (*Enallagma* spp., *Procambarus* spp. and mosquitofish) recovered later in the stream/swamp receiving system in Phase VI by February 1982. Overall, the periods covered in the study started in November 1973 and ended in February 1982, which spanned a total of ~eight years whereby the biota in the receiving system finally recovered in general.

Initial factors influencing biotic density demise included coal ash scouring and smothering of benthic macroinvertebrates in the stream channel to the swamp and further into the swamp channels as the ash ponds filled and overflowed for several months and longer. Thereafter, acidic pH from fly ash addition added to the environmental impact and some tolerant water column organisms (e.g. *G. affinis*) were adversely affected and disappeared. Perhaps elevated trace elemental concentrations in the sediment and water column had an additional confounding impact upon some of the tolerant populations that eventually recovered there. In general, the profoundly visible impacts of coal ash entrenched into the receiving stream and swamp sampling stations from ~300 m in the stream to the swamp confluence, and then ~ 1200 m thereafter into the swamp were difficult to comprehend. For the first 100 m into the swamp to Station C, most biota were severely impacted to being eradicated. Only at Station E, 1200 m downstream into the swamp was recovery apparent (Fig. 3).

Elemental Bioaccumulation

The effects of trace elemental bioaccumulation were difficult to address because of the compounding problems of benthic macroinvertebrates and mosquitofish assemblages being eradicated from the stream/swamp receiving system. However, after the coal ash settled, and one could review the data years later, there was startling metal accumulation prevalent in the eight-year study.

Cherry et al. (1976) and Rodgers et al. (1978) published metals data for water column sediment, mosquitofish and duckweed (*Lemna* sp.) when the settling ponds were overflowing in June-December 1974 (Table 1). Samples were analyzed for 40 different elements; however, data selected for only ten are presented here. These elements were selected based on their toxicity and use in other recent studies conducted in the SRP. Data presented were mean values for six sampling stations in and below the coal ash basin. The two metals found in the highest concentrations back then were iron and aluminum, both at concentrations well above their acute and chronic Water Quality Criteria (WQC) limits today (16.9 and 13.0 ppm, respectively). The average concentrations of cadmium (Cd), chromium (Cr), copper (Cu), mercury (Hg), selenium (Se), and zinc (Zn) in the drainage system water column also were greater than their respective WQC limits (Fig. 4). For example, Cu averaging 0.401 mg/l or 401µg/L exceeded the acute (19 µg/L) and chronic (12 µg/L) WQC by 22 and 33 times. Zinc (371 µg/L) exceeded chronic WQC (47 µg/L) by eight times while Cd (123 µg/L) did so by 112 times. These levels for Zn and Cd exceeded the acute WQC by 1.1 and 31 times, respectively.

Table 1. Mean Metal Concentrations in Different Media for Six Sampling Sites in and below the SRP Fly Ash Basin during June-December 1974. The BCF or Bioconcentration Factor is the Concentration in *Lemna* Divided by that in Water. All Concentrations in ppm are from Cherry et al. 1976 and Rodgers et al. 1978.

Element	Water Column	Sediment	<i>Gambusia</i>	<i>Lemna</i>	Lemna BCF In Ash Basin
Aluminum (Al)	13.0	40,657	215.5	46,762	3,597
Arsenic (As)	0.058	19.7	0.5	72.7	1,253
Barium (Ba)	0.709	294.2	19.96	426	601
Cadmium (Cd)	0.123	1.7	1.3	17.4	141
Chromium (Cr)	0.16	38.4	2.76	65	406
Copper (Cu)	0.401	51.6	8.45	101.1	252
Iron (Fe)	16.9	20,912	154.7	13,885	822
Mercury (Hg)	0.036	0.8	0.22	5.6	156
Selenium (Se)	0.107	6.1	9.4	21.4	200
Zinc (Zn)	0.371	6.5	11.79	58	156

In addition, considerable deposition of metals in sediments was evident with nearly all metals mentioned above being at concentrations an order of magnitude higher than those in the water column, ranging from 0.8 ppm for Hg to 40,657 ppm for Al (Table 1). These high metal concentrations in the water and sediment resulted in accumulation of metals in both mosquitofish and duckweed tissue (Fig. 5). The mosquitofish tissues accumulated metals at concentrations ranging from 0.22 ppm for Hg to 215.5 ppm for Al. Duckweed accumulated even greater concentrations of the various metals, and the bioconcentration factors (BCF, the concentration of a selected metal in tissue divided by the concentration in the water column) were calculated for the individual sampling sites (Rodgers et al. 1978). In the ash basin site, BCF values were as high as 3,597 for Al, 1,253 for As, and 156 for Hg, suggesting that duckweed was a sink for metal deposition. The BCF is an environmentally relevant consequence because duckweed serves as a food source for aquatic and /or terrestrial organisms, and metals will accumulate at even greater concentrations in those organisms of higher trophic levels. If that bioconcentration potential is indeed true, chronic impairment consequences can occur.

Further studies were conducted at the SRP in subsequent years when changes in pH of the water occurred. The pH of the water was 7.4 for the above-described data due to deposition of bottom ash only (Table 1). From 1975 through 1977, the mean pH of the water in the settling basins was 5.4 due to deposition of both heavy and fly ash with acidic pH excursions, and from 1978 to 1982 it was 6.5 due to deposition of both heavy and fly ash. Table 2 presents metals data for water, sediment, plants, invertebrates and fish from these two subsequent periods of time for comparison to the previous table. In general, water column concentrations of most metals remained approximately constant over the three sampling periods (e.g., bottom ash siltration, acidic pH from fly ash addition, elemental impacts), except for decreased Se (0.197 to 0.05 and 0.06 ppm) and increased Zn concentrations (0.371 to 3.87 and 1.9 ppm). Zinc concentrations also increased in the sediments at pH 5.4 and 6.5 (48 and 26 ppm, respectively) as compared to pH 7.4 (6 ppm). Other sediment metals concentrations were fairly consistent except that at pH 6.5 Cd increased to 27.5 ppm, compared to 3.4 ppm at pH 5.4 and 1.7 ppm at pH 7.4. BCF values were highly variable but in general were highest at pH 5.4 for plants, and at pH 7.4 for invertebrates and fish. Plants also generally had the highest BCF values of the three groups followed by invertebrates and fish.

Table 2. Metals Concentrations in Various Media from the SRP Fly Ash Basin Drainage System. Bioconcentration Factors (BCF) are Presented in Parentheses as Concentrations in ppm from Cherry et al. 1984.

Data from 1975-1977, pH 5.4					
Element	Water	Sediment	Plants	Invertebrates	Fish
As	0.03	25.0	38.0 (1267)	6.0 (200)	1.02 (34)
Cd	0.11	3.4	1.39 (13)	1.29 (12)	0.51 (4.6)
Cr	0.16	40.0	17.0 (106)	0.5 (3.1)	0.5 (3.1)
Cu	0.58	81.0	29.0 (50)	39.0 (67)	13.0 (22)
Se	0.05	2.6	0.29 (5.8)	1.92 (38)	6.4 (52)
Zn	3.87	48.0	16.0 (4.1)	18.0 (4.6)	34.0 (8.8)

Data from 1978-1982, pH 6.5					
Element	Water	Sediment	Plants	Invertebrates	Fish
As	0.05	70.0	11.0 (220)	17.0 (340)	1.65 (33)
Cd	0.14	27.5	0.82 (5.8)	0.74 (5.3)	0.6 (4.3)
Cr	0.06	43.0	13.0 (217)	1.0 (17)	1.0 (17)
Cu	0.60	81.0	6.0 (10)	40.0 (67)	12.0 (20)
Se	0.06	5.13	1.27 (21)	3.18 (53)	5.26 (87)
Zn	1.90	26.0	42.0 (22)	19.0 (10)	20.0 (10)

From the eight-year study of CCB constituents released by fly ash pond effluent, three to four major toxic impact pathways were ascertained by Cherry et al. (1984) at the SRP. They found that the initial receiving system impact upon the aquatic biota was due to acidic pH of 3.5-5.5, ash siltation, elevated water column elemental concentrations and bioaccumulation. The high water column concentrations of multiple trace elements were a major concern from fly ash toxicity and chronic contamination. It took approximately eight years for the aquatic biota, including macrophytes, macroinvertebrates, and fish at the SRP stream/swamp receiving system to recover after the CCB stress was initiated in 1973 (Fig. 3). For example, some benthic macroinvertebrates started to recover in three years while snails did so after four years. Crayfish, some damselflies (*Enallagma* spp.), and mosquitofish did so after eight years. Hopefully, all of us would want to learn from the destructive impacts of CCBs and realize that the impacted receiving systems studied during the mid 1970's to mid 1980's took several years to recover.

Glen Lyn Plant, VA

Water Quality Changes

The environmental impacts of fly ash released from an over-loaded settling pond discharged into a mountain stream at the GLP were not on the same order of severity as that measured at the SRP. Water samples were collected upstream of the ash effluent (Adair Run –UP or AR-UP) and within the immediate mixing zone (Adair Run –Down or AR-DN). The pH in AR-UP ranged from 7.5-7.8 throughout the 2-year study while at AR-DN, it was 8.2-8.6 from Periods 1-3. The highest single pH measurement was 9.5 in Period 3, and after cessation of effluent into the stream, pH dropped to ~7.8. Total suspended solids (TSS) were low upstream (<20 mg/L) throughout and averaged 65 mg/L (100 mg/L maximum) during Period 3. Again, in Periods 4-5, TSS was nearly the same at both sampling stations. Sulfates reached 100 mg/L during Period 3 at AR-DN compared to 25 mg/L at AR-UP. Trace metal concentrations (Cd, Cr, Cu, Zn) were not appreciably different between AR-UP and AR-DN. Only arsenic and selenium were quite high at 38 and 26 ug/L downstream compared to 2 and 4 ug/L, respectively, upstream.

Biotic Changes

Benthic macroinvertebrate communities at AR-UP and AR-DN were generally similar during Period 1, relative to density, number of genera and diversity in Fig. 6 from Specht et al. 1984. During Periods 2 and 3, these parameters declined substantially downstream while density and abundance significantly declined in Period 3. In Period 4, these three parameters were lower in AR-DN, but in Period 5, some of these parameters were actually higher in AR-DN.

Mayflies (Ephemeroptera) were the most abundant insects collected at AR-UP throughout the study but they

declined substantially in all periods at AR-DN (Fig 7). In Period 3, mayflies comprised ~18% of the composition downstream compared to ~40% upstream. Beetles (Coleoptera) were minimal throughout the study at AR-UP but became the most abundant group in Period 3, comprising ~75% of the total composition.

The six genera with the greatest differences in abundance between the two sampling stations changed abruptly in Period 3 (Fig. 8). *Heptagenia*, *Paraleptophlebia*, *Stenomema*, *Chematopysche* and *Acroneuria* declined substantially at AR-DN, while the reverse occurred for the water penny (*Psephenus herricki*). The mayfly, *Heptagenia*, had the widest differences in decline between up versus downstream followed by the stonefly predator, *Acroneuria*. By Period 5, *P. herricki* remained much more abundant downstream while *Acroneuria* failed to show any substantial recovery. The other four genera were substantially more abundant downstream in Period 5. For most genera evaluated (except *Acroneuria*), 2 to 10 months were required for the insect community to recover after cessation of ash effluent.

Acute Laboratory Toxicity

Acute toxicity testing of the fly ash and trace metals (Cd, Cu, Zn) were conducted in the laboratory for 96 hr between *P. herricki* and *Stenomema pudium*, a mayfly. *Psephenus* was one to three orders of magnitude more resistant to the trace metals than *S. pudium*. No acute LC50 values could be generated with fly ash and at 4,000 mg TSS/L, 20 and 40% mortality occurred for the water penny and mayfly, respectively. Obviously, no acute toxicity from trace metals and fly ash occurred in the Adair Run stream as laboratory toxicity data were two to several orders of magnitude higher for trace metals and fly ash TSS, respectively, than that measured in the receiving system.

Trace Metal Toxicity from Fly Ash

Trace element concentrations vary widely among coal ashes from different sources (Theis and Wirth 1977). Although fly ash particles have less overall aqueous solubility, the outer 50-300 Å layer can be easily solubilized in water (Natusch 1978), and then released into solution depending upon pH, length of time in solution, ash solution ratio, temperature, complexing agents and particle surface areas and origin (Theis and Wirth 1977, Chu et al. 1978, Fisher and Natusch 1979, Dudas 1981). Acidic pH conditions create the highest dissolved heavy metal concentrations in fly ash ponds (Theis and Wirth 1977, Stern and Stern 1980, Turner et al. 1982).

In tests conducted by Cherry et al. (1987) from the GLP, trace elemental concentrations were evaluated from raw heavy ash collected at the settling pond influent pipe, raw fly ash obtained dry from the electrostatic precipitators, and leached fly ash by adding 0.1 N HNO₃ to the ash placed in 250-ml beakers with distilled water stirred for 1 hr. A CAMECA IMS-3f ion microanalyzer was used for determination of minor and trace elements below scanning electron microscope (SEM) detectability by C. Evans and Associates, Palo Alto, California. The incident ion beam literally burned the sample surface away (penetration rate was ~2-5 ?/sec), and an isotopic yield from the sample as a fraction of time, represented an approximation of the distribution of that isotope with depth below the particle surface.

In heavy ash, the concentration of elements was low or not substantially different from the surface to inner depths of the ash particles (Cherry et al. 1987). In raw fly ash, ion yields for 14 elements were monitored and several had a substantial surface enrichment (Cu, Ni, Cr, Se, Pb, Ti, As, Cd, Hg) while only Ca, Mg, Zn, Sr, and Ti did not vary with surface enrichment to the inner particle depth. In acid-leached fly ash particles, the surface ion intensity was considerably lower than that found for raw fly ash as the outermost particle surface was depleted of trace elements verifying their accumulation on particle surfaces.

To further substantiate trace metal availability on ash particle surfaces, heavy ash was not acutely toxic to rainbow trout (*Salmo gairdneri*) and bluegill sunfish (*Lepomis macrochirus*) up to 1500 mg/L TSS at pH 5.0, 7.5 or 8.5 and to bluegill at fly ash concentrations up to 1360 mg/L TSS (Cherry et al. 1987). Rainbow trout, however, were highly sensitive to fly ash at concentrations of 4.3 to 20.5 mg/L when dissolved metal availability was high but not sensitive at higher particle concentrations (58 to 638 mg/L TSS) when dissolved metals were low. No rainbow trout mortality occurred in acid-leached TSS concentrations up to 2,350 mg/L TSS. The authors went further to emphasize that if the pH of the fly ash effluent is contained within the range of 6.0-9.0, acute toxicity can be attributed to trace element availability from fly ash but not heavy ash. In essence, control of holding pond and effluent pH while maximizing pond residence time, were important strategies for minimizing ash pond discharges on fish because certain trace metal coatings on fly ash particles became available to the aqueous phase of the effluent. Even though

the US EPA (1982) requires dry ash fly ash disposal for new fossil-fuel power plants, all the older power plants currently using wet-ash pond disposal are exempt. Therefore, although these two environmental scenarios occurred 2-3 decades ago, there are a number of currently operated power plants across the country that use wet-ash pond disposal, so these environmental impact problems documented years ago may not be exempt from re-occurring somewhere else at a more chronic level of toxic consequences.

Overview of Toxic Consequences from CCBs

When having the opportunity to reflect upon the toxic consequences of coal ash, especially at the SRP receiving system after 30+ years of research experience, toxic consequences of CCBs became clearer to this researcher (D. Cherry). When initiating the research at the SRP in 1972, it became clear that the stream/swamp receiving system contained hardy populations of biota. In fact, only one fish species, the mosquitofish, was found in the ash-influenced watershed. Therefore, previous years of ash influence had created a tolerant assemblage of benthic macroinvertebrates and *G. affinis*, one of the most tolerant fish species known to elevated temperature (Carlander 1969, Cherry et al. 1976).

In 1973-1976, the visual impact of heavy ash overflowing the settling ponds and partially filling the stream channel into the swamp and then covering acres of swamp habitat, was an overwhelming incident for this fledging scientist. So the initial view was that heavy ash loading was “bad”, but it can be easily controlled with proper settling pond management today. When acidic fly ash was introduced into the original holding ponds after they were excavated, acidic pH of 3.5 was toxic enough to eradicate the near surface dwelling mosquitofish that escaped the benthic smothering of the heavy ash. Then they succumbed to the acidic pH exposures, as well as remnants of tolerant benthic macroinvertebrates. Again, acidic pH discharges are easily remedied today by alkaline neutralization.

Reflecting upon the eight-year study at the SRP, the subtle concentrations of various hard and soft metals are really the major impacting parameters of CCBs in aquatic receiving systems today. It was not until reviewing the initial 40-trace element database of 1974 recently that the elevated content of several elements (Fe, Cu, Zn, Cr, Cd, Sb, Se, Hg) were found to be extremely loaded in the water column sediment, and bioaccumulated in the mosquitofish (Cherry et al. 1976). For example, when reviewing the average aquatic metal concentrations of iron, aluminum, copper, zinc, and cadmium in the stream channels in 1974, iron concentration was 16,900 µg/L or 16.9 times greater than the US EPA WQC (Fig. 4). For aluminum, the concentration of 13,000 µg/L was 149 times greater than the WQC. Copper and zinc were 33 and 8 fold above the WQC but cadmium was 123 times higher. Then when reassessing the bioaccumulation data from Cherry et al. 1976 and Rodgers et al. 1987 for aluminum and iron from water to midge, crayfish, fish and duckweed, these trace metals were accumulating in the body burden of all four biotic groups, but the bioaccumulation was accentuated 3+ fold for both metals (Al and Fe) in duckweed (Fig. 5). For example, the aluminum concentration (13 mg/L) in water was multiplied 3,597 fold (46,762 ppm) in duckweed while iron (169 mg/L) was magnified 82 fold (13,885 ppm).

Controlling trace element contamination at the SRP was costly by expanding the size of the holding ponds, but the acute impacts of heavy ash siltation and acidic fly ash toxicity overshadowed the subtle, long-term consequences of trace element contamination from CCBs. Today, such acutely toxic problems at the SRP have been eliminated, but the longer term chronic toxic consequences of low-level trace element bioaccumulation is the real concern of CCB's and requires more research and possible federal regulation. For example, bioaccumulation of selenium (Se) from fly ash effluent has had major impacts in Belews Lake, North Carolina (Lemly 1985), and the real concern was the bioaccumulation potential upon fish sterility. Also, the water column concentrations of Se in the SRP receiving system were exceptionally high, 0.107 mg/L in 1974 to 0.05-0.06 mg/L in 1975-1982, which were orders of magnitude above current US EPA WQC limits of 0.005 mg/L (5 µg/L). Current WQC limits for Se may not be protective enough, based upon bioaccumulation studies by Lemly (1992, 1993), in that waterborne concentrations of 2 ug/L may be hazardous to some biota such as fish, predatory birds, and mammals. The CCB database at various landfill and ash holding pond sites may need to be readdressed because Se monitoring has been neglected at many of these sites in the past. From the above review, the potentially real to subtle toxic issues of CCBs is related to the longer-term chronic exposure of several trace elements in the 2004 arena and beyond. If not, we as scientists, those as industrialists, others as regulators, etc., are missing the “far sighted”, long-term picture of subtle chronic toxicity from CCBs.

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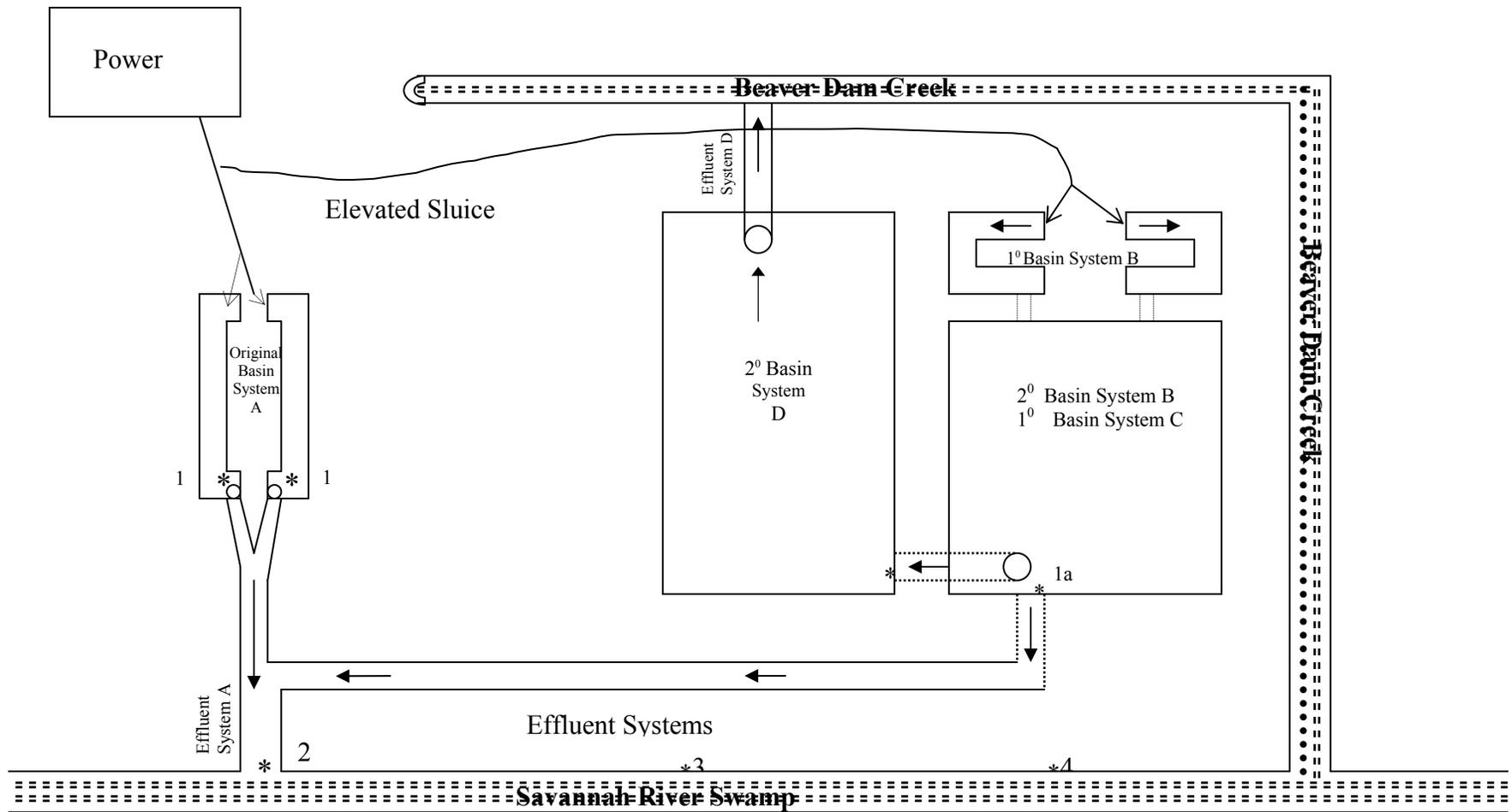


Figure 1. Evolution of the coal ash settling pond system from the original dual ponds (A), to the larger primary and secondary ponds (B and C), to a larger secondary pond (D) at the Savannah River Project.

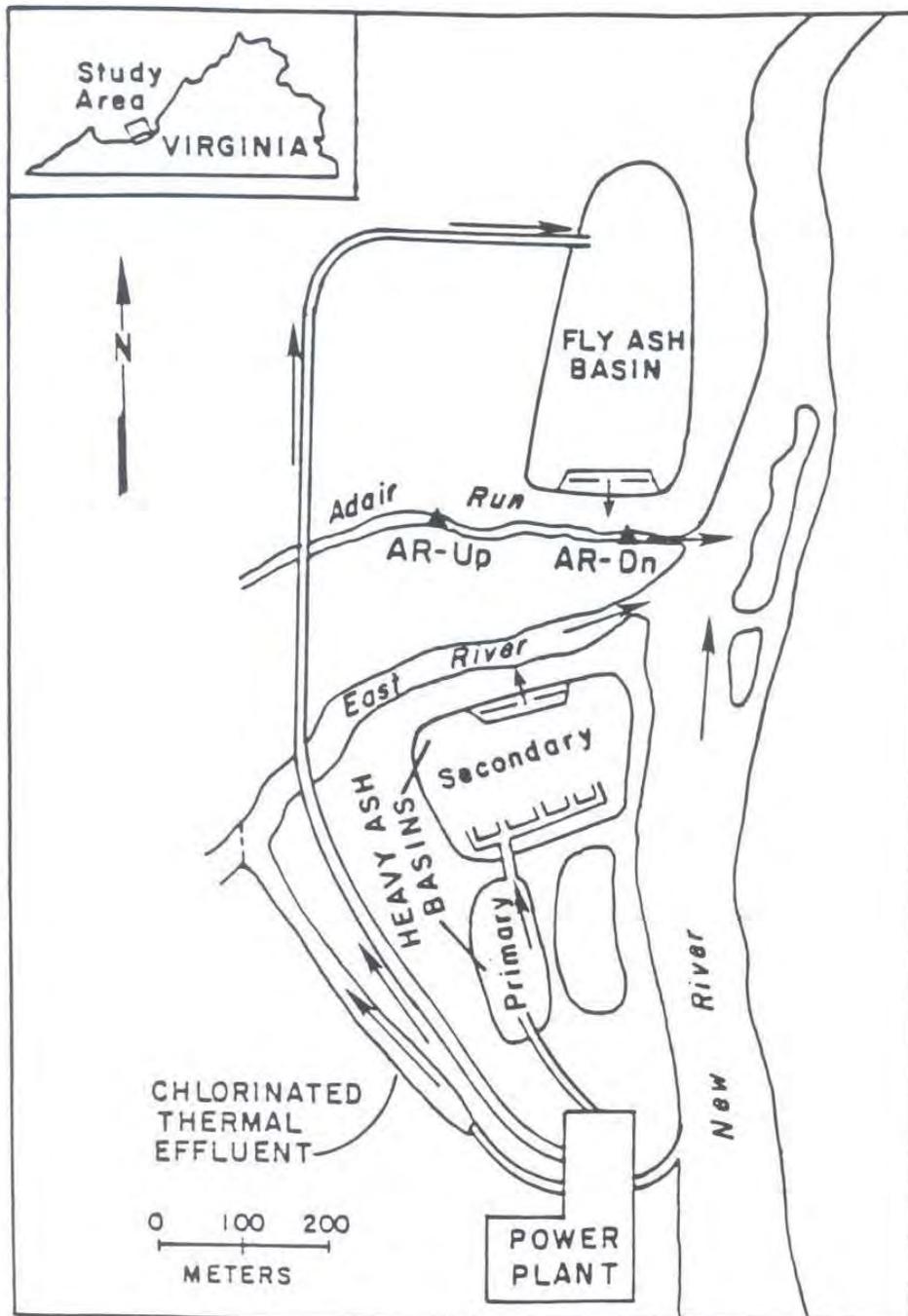


Figure 2. Heavy and fly ash settling ponds at the Glen Lyn Plant where heavy ash was released into the East River and fly ash into the Adair Run mountain stream.

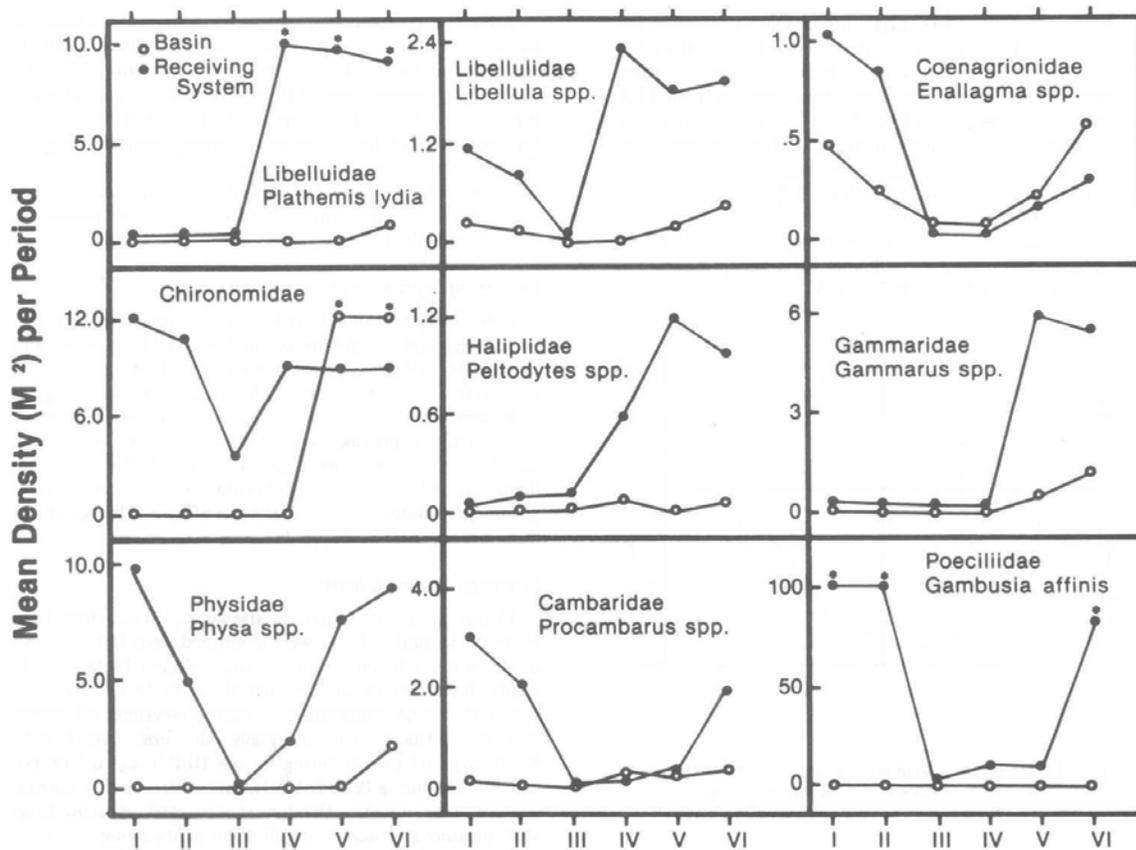


Figure 3. Changes in several benthic macroinvertebrate populations and the mosquitofish during periods I-V (November 1973-February 1982) at the Savannah River Project.

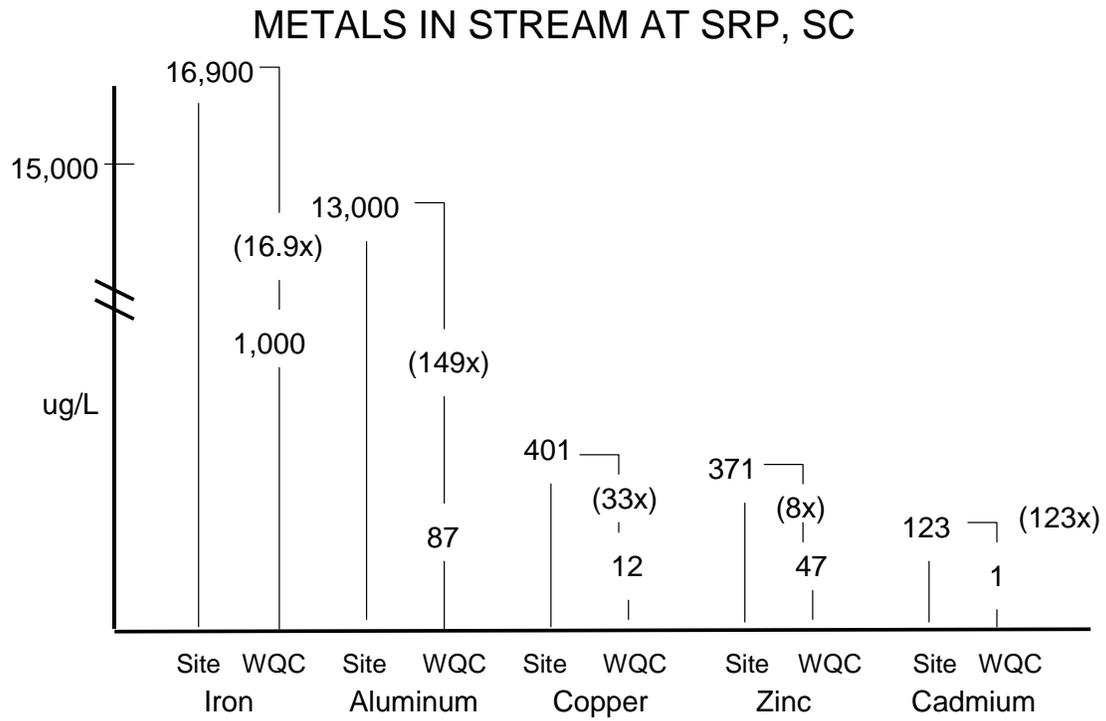


Figure 4. Average trace metal concentrations taken in 1974 from all sampling stations in the ash receiving system compared to the chronic US EPA Water Quality Criteria at the Savannah River Project.

BIOACCUMULATION IN SRP BIOTA

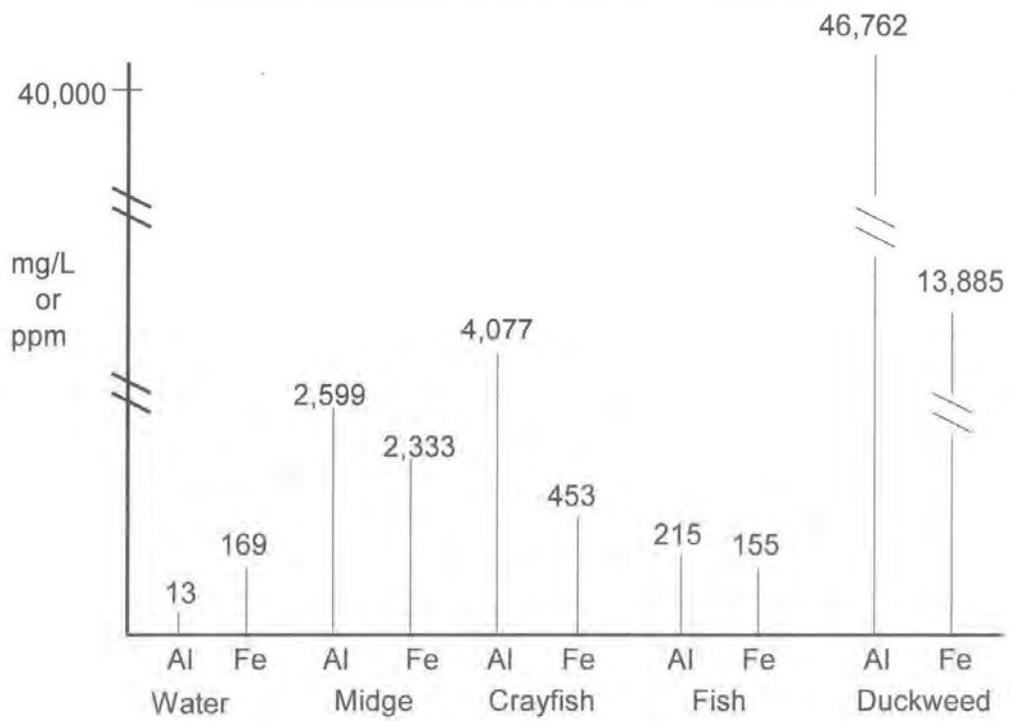


Figure 5. Bioaccumulation of aluminum (Al) and iron (Fe) from the water to selected portions of the food chain in the ash receiving system of the Savannah River Project.

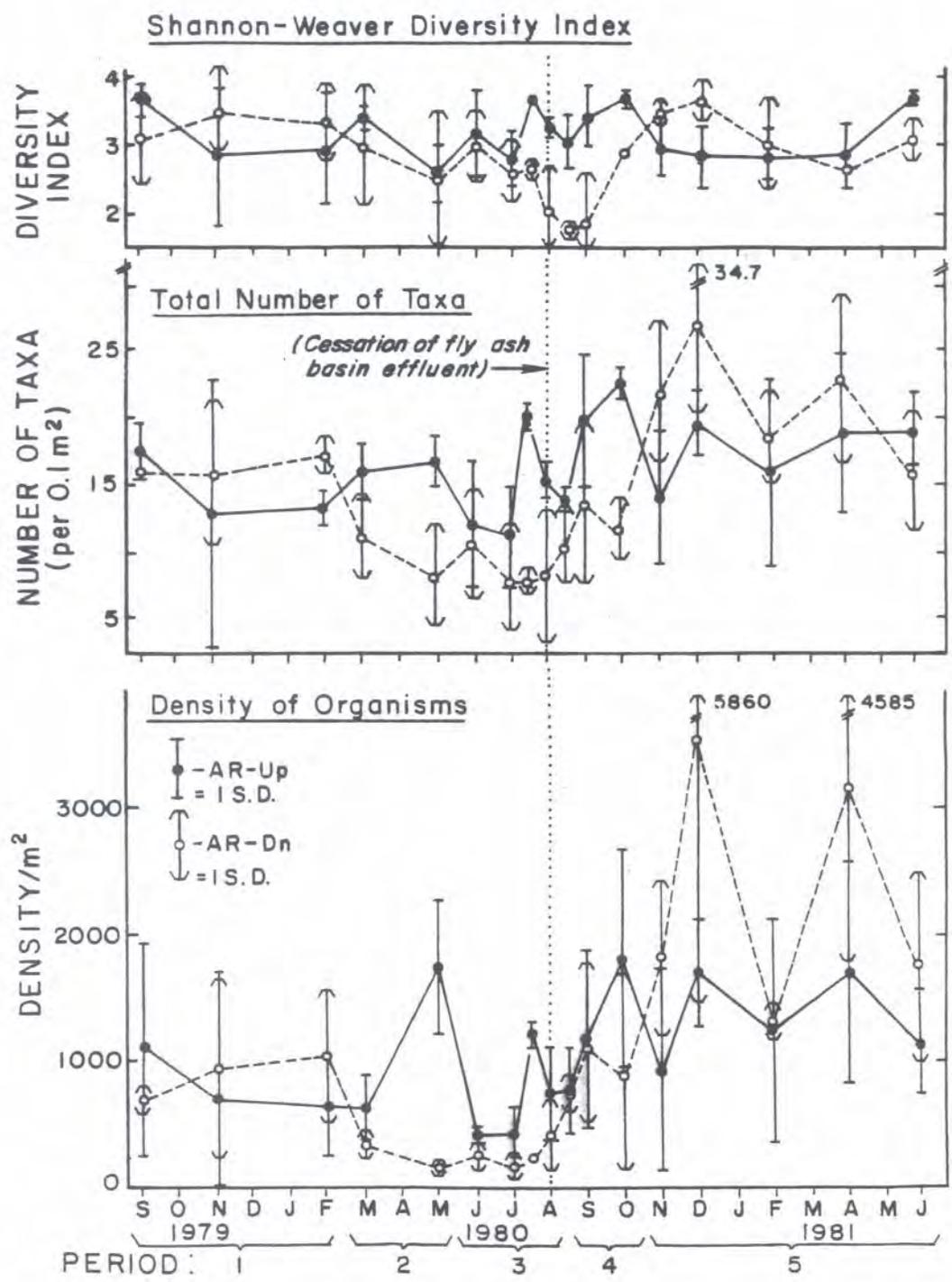


Figure 6. Fluctuations in the benthic macroinvertebrate community diversity index, taxa, and density from Period 1 (July 1979) through 5 (June 1981) in Adair Run upstream (AR-UP) and in the ash influenced area downstream (AR-DN) at the Glen Lyn Plant.

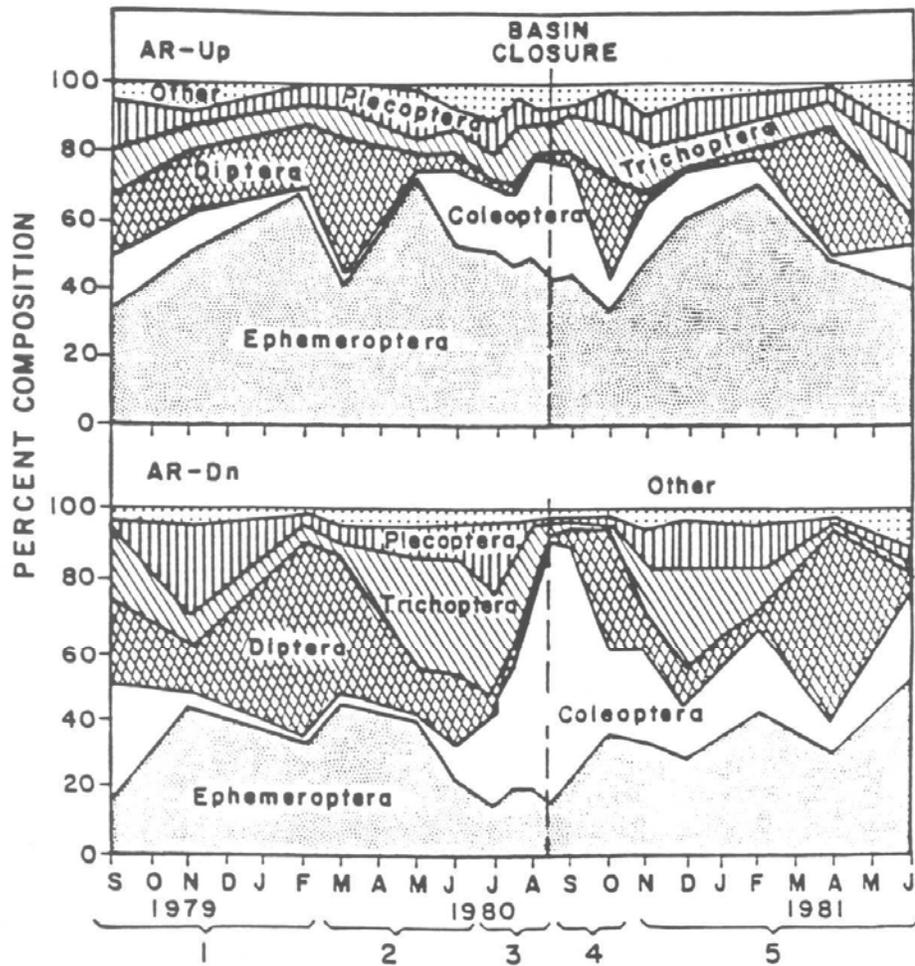


Figure 7. Fluctuations in the percent composition of insect orders in Periods 1-5 (September 1978-June 1981) in Adair Run upstream (AR-UP) and downstream (AR-DN) at the Glen Lyn Plant.

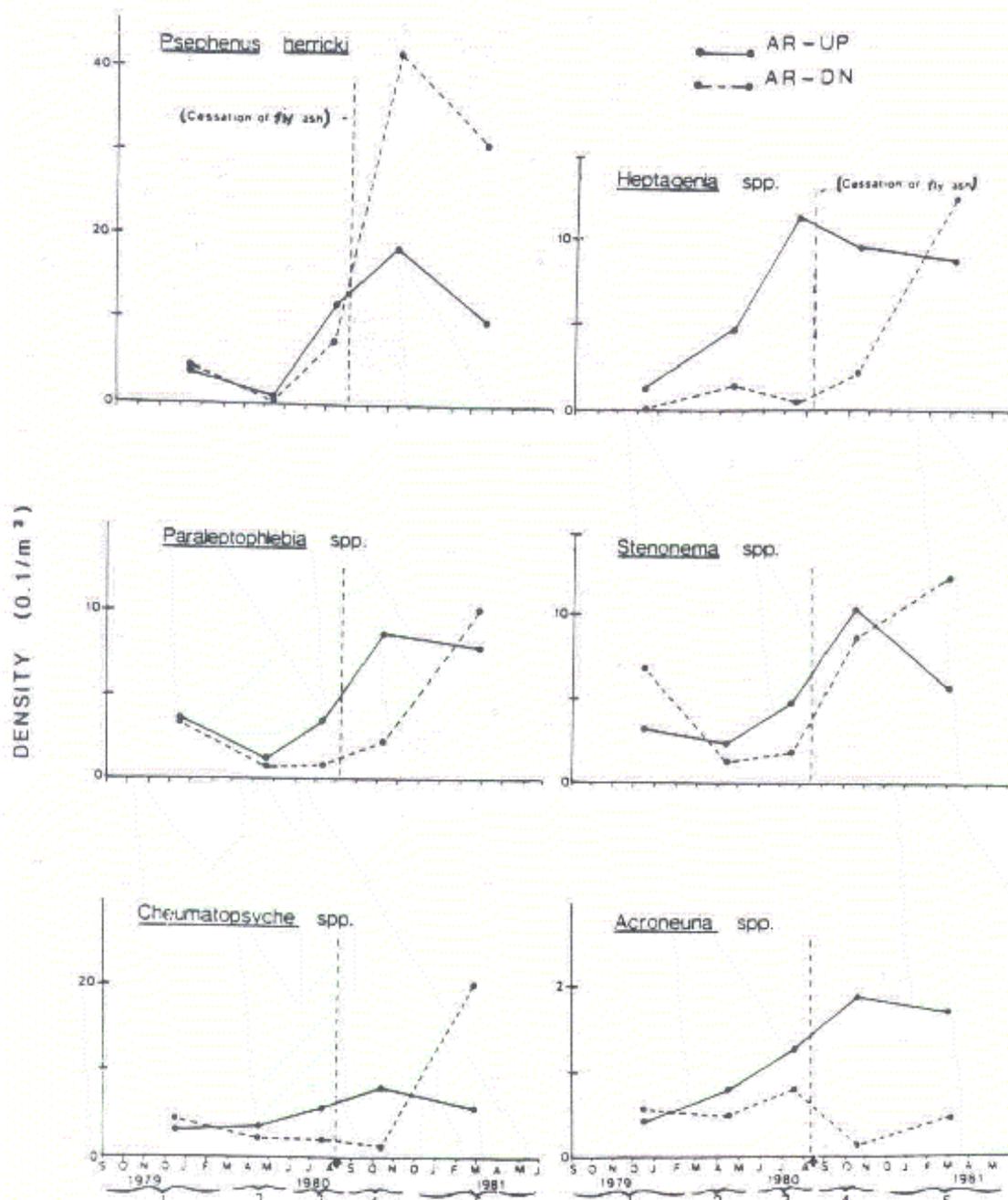


Figure 8. Fluctuations in the percent composition of insect orders in Periods 1-5 (September 1978-June 1981) in Adair Run upstream (AR-UP) and downstream (AR-DN) at the Glen Lyn Plant.

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