

USE OF COAL COMBUSTION PRODUCTS IN MINE-FILLING APPLICATIONS: A REVIEW OF AVAILABLE LITERATURE AND CASE STUDIES

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EXECUTIVE SUMMARY

Background

Surface and underground mining combine to produce over 800 million tons of coal annually, and the vast majority of this coal is burned in utility boilers to generate electricity. Combustion of coal in the utility boilers produces coal combustion residues or products (CCPs) in the form of fly ash, bottom ash, boiler slag, and flue gas desulfurization (FGD) sludge. In the year 2003, U.S. electric utilities produced approximately 122 million tons of CCPs. Coal fly ash constituted about 70 million tons, the bottom ash/boiler slag accounted for about another 20 million tons, and the remaining 30.8 million tons was FGD material. Approximately 62% of the CCPs generated are land disposed. About 38% of the fly ash generated was used within cement/concrete/grout, structural fill, waste stabilization/solidification, mining applications, and road base/subbase, accounting for most of the fly ash volume utilized. About 45% of bottom ash was used mostly in structural fills, road base, snow and ice control, mining application, and cement/raw feed for clinkers. About 96% of the boiler slag was utilized mostly for blasting grit and roofing granules. About 70% gypsum was used mostly for wallboard production. FGD material generated by wet scrubbers was utilized only about 2.75% and dry scrubber-generated FGD material accounted for about 13.5% use.

Use of CCPs in Mines

CCPs possess several physical and chemical properties/characteristics that are beneficial both from environmental and economic standpoints. Mine reclamation represents a potential beneficial use of CCPs that has been receiving increased attention in recent years. Coal mining operations have produced both open pits and deep underground cavities that can be filled by CCPs. Placement of CCPs into deep mines can provide structural support to abate subsidence, and placement of CCPs in surface mines or other open pits can aid in restoring mined land to beneficial use. The use of CCPs as mine backfill may provide the additional potential benefit of limiting impacts of acid mine drainage (AMD). Many CCPs are alkaline materials that can neutralize acidic groundwater and/or inhibit production of acid. Placement of CCPs also may reduce the permeability of mine strata and divert water away from acid-generating materials. Although CCPs possess several beneficial physical and chemical properties, there are concerns from regulators and environmental groups regarding potential for release of toxic chemicals in the leachates from the CCPs. Therefore, scientifically sound information is needed so that environmental concerns can be adequately and reliably identified and addressed.

CBRC and Cofunders

Ish Inc. and GeoTrans, Inc., were selected by the Combustion Byproducts Research Consortium (CBRC) to conduct research that will compile and synthesize information from several case studies involving CCP usage for mine applications for dissemination to inform regulators,

environmental interest groups, and the generators of CCPs about the benefits and impacts of CCP use for mine filling. CBRC provided funding for this research project, with additional cofunding by the Public Service Co. of Colorado, McDonald Farms, American Coal Ash Association, Utility Solid Waste Activities Group, GeoTrans, Inc., and Ish Inc.

Report Content and Results of Literature Search

This report contains information gleaned from literature on chemical and physical characteristics of CCPs produced in the United States along with some information on mine spoil material. This report also contains background information on coal mines and a brief discussion of the geochemistry of coal mines particularly to describe the formation of AMD. A summary of available data on water quality characteristics of mined areas is presented. A brief description of the geochemical interactions between the AMD water and CCPs is also presented in this report.

A summary list is presented of mine sites where CCPs are being utilized for filling the mined land and/or for abating AMD conditions.

Beneficial use of CCPs for coal mine reclamation occurs in varying degrees across the United States. Injection of CCPs into deep mines has been performed to provide structural support for subsidence abatement, and placement of CCPs in surface mines has been utilized to reclaim mined land to original grade and to mitigate AMD. Such practices have been employed at both active and abandoned coal mines.

Several surface and deep mines in the midwestern and western United States that have utilized CCPs for reclamation are identified in this report. These mines include the Keenesburg and Trapper Mines (CO); Wyodak, Glen Rock, and Rock Springs Mines (WY); Midwestern Abandoned Mine and Arnold Willis City Mine (IN); Big Gorilla Pit (PA); Winding Ridge Frazee Mine (MD); Savage Mine (MT); and Universal Mine (IN). Active mining operations continue to occur at the Trapper, Wyodak, and Savage Mines. Types of CCPs placed at these mines include fly ash, bottom ash, and FGD materials.

Thirteen mines were selected based on availability of site-specific data required to perform a reasonable evaluation of the benefits and impacts of CCP placement on groundwater quality. Tables and graphs are included, when available, to illustrate important aspects of each case study. These case studies represent a large range of CCP uses from filling of mine pits to using CCP grout mix to minimizing/eliminating acid mine drainage. A brief summary of each case study follows:

1. Placement of fly ash, bottom ash, and scrubber ash in mined areas at Wyodak Mine began in 1978. Approximately 5,000,000 yd³ of ash has been placed in 13 separate pits. Results indicate that the average groundwater quality throughout the Wyodak site compares favorably with the Wyoming Department of Environmental Quality (DEQ) Class III (livestock use) standard. Mean concentrations for all of the measured constituents in wells are at or below Class III standards.

2. Reclamation at the Keenesburg site is being performed using fly ash and bottom ash derived primarily from combustion of Keenesburg coal. Ash was placed at least 5 feet above the premining groundwater table, and reclamation includes placement of a vegetative final cover consisting of at least 5 feet of compacted overburden and 3 feet of topsoil material. Upon comparison of water quality in upgradient versus downgradient wells and for sampling events prior to ash placement (1978–1986) relative to sampling events following ash placement (1988–2000), there is little evidence indicating that elevated levels of regulated constituents in site groundwater are a direct result of leaching from the ash.
3. Trapper Mine is a surface coal mine located approximately 6 miles south of Craig, Colorado. Trapper Mine began operation in 1977 and produces up to 2.8 million tons of coal annually. Deposition of CCPs in the mined out areas of A Pit began in 1984. Trapper Mine has managed approximately 390,000 tons of CCPs per year since 1984 with total disposal quantities approaching 7 million tons to date. Current ash placement practices involve deposition of sufficient overburden material into the pit bottom such that the ash is placed above the expected postmining groundwater table. Overburden materials are placed above the ash and revegetated as part of the final reclamation plan. Comparison of the historic groundwater concentration data with Colorado standards indicates little or no evidence of groundwater impacts associated with most of the analyzed constituents.
4. Savage Mine has been in operation since 1958 as a surface lignite mine and currently produces 250,000 tons of lignite annually. Savage Mine began utilizing fly ash and bottom ash as backfill in 1987. The annual volume of ash received by the Savage Mine is variable and dependent upon the amount of coal burned by Holly Sugar in relation to the quantity of sugar beets processed. The estimated cumulative volume of ash placed between 1987 and 2000 is approximately 130,000 cubic yards.
5. The Mount Storm case study is about placement of coal ash in an active coal strip mine. The downgradient groundwater-monitoring data reveal no evidence of contamination over the 10-year period of operation. Coal ash (both bottom ash and fly ash) is being placed at a rate of about 800,000 tons annually in an active strip mine near an electric power plant in West Virginia. The strip mine provides coal for the power plant and is located on the plant property. Ash placement began in 1987. No ash has been placed below the water table because the groundwater table is deeper than the mine floor.
6. In 1988, PSI Energy, Inc. (now Cinergy Corporation) acquired a portion of the Universal Mine site containing the final cut pit for the express purpose of coal ash deposition and surface mine reclamation. Indiana Department of Natural Resources (DNR) issued a permit to PSI Energy to dispose of fly ash/bottom ash from its nearby Wabash River Station to fill and reclaim the mine pit. Between April 1989 and the end of October 2001, Cinergy placed approximately 1.6 million tons of coal ash from a nearby power plant to completely fill the open pit. The monitoring data to date indicate that the alkaline coal ash leachate has been effective in improving AMD water

quality that was present at the site. The coal ash leachate neutralized the acidic pH, increased alkalinity, essentially eliminated acidity, and significantly decreased manganese, iron, and sulfate concentrations. There were no indications of any other trace metal migration via the mine seep. However, the coal ash leachate did significantly increase boron concentrations in the mine seep water.

7. The Midwestern Abandoned Mine is a case study where a state agency elected to place CCPs with a Poz-O-Tec (a mixture of FGD sludge, fly ash, and quicklime) cap, which resulted in reduced infiltration and improvement in water quality by neutralization. The Midwestern Abandoned Mine consists of approximately 550 acres of previously mined land, which in some instances intersects with abandoned deep mining of the same coal seam. Based on the monitoring results, Branam et al. (1999) it was concluded that using CCPs to reclaim the Midwestern Abandoned Mine has resulted in the reduction of AMD leaving the site. This response is ascribed to the reduction in vertical recharge of oxygenated water by the fixated scrubber sludge cap and the neutralization provided by the alkaline CCPs.
8. The Arnold Willis City Underground Coal Mine in Indiana is an example in which fixated scrubber sludge (FSS) has been successfully injected into an abandoned underground coal Mine for stabilization by filling mine voids. Groundwater-monitoring data indicated that trace metals and sulfide remained unaffected by the placement of FSS. An FSS grout consisting of a mixture of FGD scrubber sludge, fly ash, lime, and water was developed for injection into the abandoned deep mine to abate surface subsidence and reduce acid mine drainage. A total of 12,502 m³ of FSS was injected over an 8-week period, resulting in filling of about 5 acres of the mine.
9. The Harwick Mine Complex includes the Monarch, Old Harwick and Cornell Mines covering approximately 7000 acres. This coal mine is a deep mine and was operated from about 1932 through 1970. The mine disposal operation consists of a wet ash-handling system to pump 10% solids slurry for a distance of approximately 8000 ft to two operating injection boreholes at the Harwick Mine Complex. Approximately 3 to 4 million gallons per day of the slurry is conveyed. Approximately 150,000 tons of coal ash is injected annually into the mine along with millions of gallons of water. The water quality data from samples of the mine water indicate no adverse effect on the water in the Harwick Mine Complex.
10. The Clinton County, Pennsylvania, Mine provides an example of the placement of fluidized-bed combustor (FBC) ash in a closed surface coal mine resulting in beneficial effects on water quality because of the favorable geochemistry that occurs. The alkaline FBC ash neutralizes the acidic AMD waters, resulting in precipitous decreases in arsenic, cadmium, and aluminum concentrations as a result of lower solubility and precipitation of solids. Results indicate that the injection of grout caused a temporary increase in pH from about 2.3 to about 9 as the alkaline FBC ash neutralized the acidic AMD waters. However, within a short time the pH again became acidic, with sulfate and aluminum returning to the pregrouting concentrations, although arsenic and cadmium remained at much lower levels.

11. In eastern Pennsylvania, there are several preact stripping pits in the middle of an anthracite coal basin where active strip and deep mining for coal was practiced since the 1800s. The strip mined pit known as Big Gorilla was one such location. The Pennsylvania Department of Environmental Protection's Regional Mining office in conjunction with the Wilkes-Barre Regional office issued a demonstration permit for the placement of cogeneration-derived dry fly and bottom ash into standing water in the Big Gorilla Pit. Ash deposition has taken place since August 1997. Over 3 million tons of ash was used to completely fill the pit which contained an acid mine water pool. The Big Gorilla water has maintained a consistently high pH value in response to the placement of ash. Iron, manganese, magnesium, aluminum, and zinc all have decreased significantly. One long-term effect of ash placement in the former Big Gorilla Mine pool will be the prevention of acidic water production through the surface mine pool.
12. The Red Oak Coal Mine was operated from 1907 to 1925 utilizing the room-and-pillar extraction method. The mine which covers approximately 46.5 acres contains water pools or reservoirs of AMD. The University of Oklahoma and the Oklahoma Conservation Commission injected 418 tons of FBC ash in 15 hours into this acidic (pH 4.3) flooded mine to chemically alter the mine water.
13. The Maryland Department of Natural Resources Power Plant Research Program and the Maryland Department of Environment Bureau of Mines launched a joint effort with private industry to demonstrate large-volume beneficial uses of CCPs to create flowable grouts for placement in abandoned, underground coal mines to reduce acid formation. In April 1995, this multiyear project initiative started with the Winding Ridge Demonstration project involving injection of a 100% CCP grout into the Frazee Mine, located near Friendsville, Garrett County, MD. The Winding Ridge Demonstration project at the Frazee Mine has shown that CCP grout mixture can be beneficially used for abandoned underground coal mines to reduce acid formation as well to fill mine voids with a high-strength, low-permeability material that would control mine subsidence. The placement of the CCP grout appears to have not caused an unacceptable water quality impact either.

Conclusions

This literature review report on the use of CCPs in mine-filling activities provides a readily available resource for regulators, the general public, environmental interest groups, and potential users of CCPs by synthesizing technical information on a range of case studies. Each case study is different in several details and provides the readers insights into the use of CCPs from benefits and limitations standpoints. The technical information can be used to determine and decide on environmentally compatible uses of CCPs in surface and underground coal mines.

USE OF COAL COMBUSTION PRODUCTS IN MINE FILLING APPLICATIONS A REVIEW OF AVAILABLE LITERATURE AND CASE STUDIES

1.0 INTRODUCTION

Coal is a widely distributed fossil fuel across the United States. Surface and underground mining combine to produce over 800 million tons of coal annually, and the vast majority of this coal is burned in utility boilers to generate electricity. Combustion of coal in the utility boilers produces coal combustion residues or products (CCPs) in the form of fly ash, bottom ash, boiler slag, and flue gas desulfurization (FGD) sludge. In the year 2003, U.S. electric utilities produced approximately 122 million tons of CCPs. Coal fly ash constituted about 70 million tons, the bottom ash/boiler slag accounted for about another 20 million tons, and the remaining 30.8 million tons was FGD material (American Coal Ash Association [ACAA], 2004). Approximately 62% of the CCPs generated are land disposed. About 38% of the fly ash generated was used in cement/concrete/grout, structural fill, waste stabilization/solidification, mining applications, and road base/subbase, accounting for most of the fly ash volume utilized. About 45% of bottom ash was used mostly in structural fills, road-base, snow and ice control, mining application, and cement/raw feed for clinkers. About 96% of the boiler slag was utilized mostly for blasting grit and roofing granules. About 70% gypsum was used mostly for wallboard production. FGD material generated by wet scrubbers was utilized only about 2.75% and dry scrubber-generated FGD material accounted for about 13.5% use.

The CCPs possess several physical and chemical properties/characteristics that are beneficial both from environmental and economic standpoints. For example, fly ash has been used in the manufacture of portland cement, as a cement substitute in concrete, and as a stabilizing agent in road bases and subbases because of its wide availability and pozzolanic nature. Compacted fly ash also is considered a good candidate for structural fill on compressible ground since it is strong and durable but lighter than conventional fill materials. Gray et al. (1991) investigated the use of fly ash in slurry wall backfill mixtures and observed that these mixtures can exhibit high sorptive capacity for nonpolar organic pollutants. In addition, other researchers have investigated the potential use of CCPs in waste containment liners and have shown that it is possible to achieve hydraulic conductivity values below 10^{-7} cm/s using compacted fly ash alone or in conjunction with sand and/or bentonite (Gray et al., 1991).

Mine reclamation represents another potential beneficial use of CCPs that is receiving increasing attention in recent years. Coal-mining operations have produced both open pits and deep underground cavities that can be filled by CCPs. In addition, non-coal-mining operations (e.g., limestone, clay, sand and gravel) result in the creation of open pits. Placement of CCPs into deep mines can provide structural support to abate subsidence, and placement of CCPs in surface mines or other open pits can aid in restoring mined land to beneficial use. Use of CCPs as mine backfill may provide the additional potential benefit of limiting impacts of acid mine drainage (AMD). Many CCPs are alkaline materials that can neutralize acidic groundwater and/or inhibit production of acid. Placement of CCPs also may reduce the permeability of mine strata and divert water away from acid-generating materials (Kim and Cardone, 1997).

1.1 Scope of the Research

Mine reclamation has been identified as a long-term, large-volume beneficial use market for CCPs. Nonetheless, the use of CCPs in mine reclamation currently is performed on a limited basis relative to the overall quantity of CCPs generated each year. Only 0.68 million tons of fly ash, 1.2 million tons of bottom ash and about 0.39 million tons of FGD materials were used in mining applications in year 2003 (ACAA, 2004). Although CCPs possess several beneficial physical and chemical properties, there are concerns from regulators and environmental groups regarding potential for release of toxic chemicals in the leachates from the CCPs. Therefore, scientifically sound information is needed so that environmental concerns can be adequately and reliably identified and addressed.

Ish Inc. and GeoTrans, Inc., were selected by the Combustion Byproducts Research Consortium (CBRC) to conduct research that will compile and synthesize information from several case studies involving CCP usage for mine application for dissemination to inform regulators, environmental interest groups, and the generators of CCPs about the benefits and impacts of CCP uses for minefilling. CBRC provided funding for this research project, with additional cofunding by the Public Service Co. of Colorado, McDonald Farms, ACAA, the Utility Solid Waste Activities Group, GeoTrans, Inc., and Ish Inc.

1.2 Report Contents

This report contains compilation, synthesis, and limited evaluation of the available information from a number of mine sites where CCPs have been or are being used to fill the mined lands. This report contains a brief summary of physical and chemical composition of CCPs produced in the United States along with some information on mine spoil material. This information is presented in Section 2. Section 2 also contains a summary list of mine sites where CCPs are being utilized for filling the mined land and/or for abating AMD conditions. Section 3 contains a brief discussion of geochemistry in coal mines particularly to describe the formation of acid mine drainage. A brief description of the geochemical interactions between the AMD water and CCPs is also presented in this section. Section 4 contains summaries of case studies for which environmental monitoring information was obtained from the operators/owners of the sites. In Section 5, conclusions and recommendations are presented.

2.0 PHYSICAL AND CHEMICAL COMPOSITION OF CCPS

2.1 Physical Characteristics

CCPs include fly ash, bottom ash, FGD materials, and boiler slag. These different types of CCPs represent approximately 58%, 15%, 25%, and 1.5%, respectively, of the total CCPs produced in the United States in 2000 (ACAA, 2004). The smaller ash particles contained in flue gas are referred to as fly ash and are collected by air pollution control equipment such as electrostatic precipitators. The fly ash consists primarily of particles between 5 and 100 μm in size. Fly ash comprises spherical, glassy particles comparable in size to silt and/or fine sand. Larger particles that fall to the bottom of the boiler result in the production of bottom ash or

boiler slag. Bottom ash is a coarser-grained material, with particle sizes comparable to sands and gravels. Boiler slag and bottom ash consist of particles ranging from 100 μm to 10 mm. FGD materials are generated when some of the sulfur in the flue gas is removed. Wet FGD systems generate waste with a particle size range of 0.001 to 0.05 mm, whereas the dry systems produce waste with particle size in the 0.002- to 0.074-mm range. FGD gypsum is also produced at some of the power plants from sulfur removal activities.

The physical properties of CCPs have been well characterized. Fly ash particle morphology has been extensively investigated by scanning electron microscopy with four general categories describing most particles: 1) cenospheres, 2) plerospheres, 3) spherical particles with crystals, and 4) irregularly shaped solid particles. The fly ash exhibits light to dark colors. The bottom ash and boiler slag are black or gray in color.

Reported specific gravity for most coal fly ash and bottom ash falls between 2.2 and 2.6, and the bulk density ranges between the values of 80 and 90 lb/ft^3 . Crushed boiler slag has been reported to have high hydraulic conductivity (permeability) equivalent to that of fine gravel. Bottom ash permeability ranges from 10^{-3} to 10^{-1} cm/sec . Fly ash material exhibits permeability typically in the range of 10^{-6} to 10^{-4} cm/sec that is similar to that for silty-clay soils.

2.2 Chemical Composition and Leaching Characteristics

Numerous studies have been completed to provide data on the chemical composition of CCPs. Oxides of silicon, iron, aluminum, and calcium account for about 95% of the weight of the bottom ash, fly ash, and boiler slag. Magnesium, potassium, and sodium are also present in the ash. The FGD sludge is composed mostly of calcium and sulfur when limestone/lime is used to scrub the flue gas. All CCPs also contain trace amounts of many metals. A comprehensive elemental characterization study of samples of fly ash, bottom ash, and FGD sludge was completed by the Electric Power Research Institute (EPRI) in 1987. Table 2.1 provides the summary of the chemical concentration data from the EPRI study. This study did not contain data on Boron in CCPs. The presence of boron in CCPs, however, is well documented.

Western coal fly ash and FGD sludge are typically alkaline and possess acid neutralization properties which make them useful materials for abating acid mine drainage and for filling mined land for reclamation. Additional available chemical characteristic data and information on alkaline CCPs and their use in minefilling are included in the individual case study summaries later in this report.

2.3 Physical and Hydraulic Characteristics of Mine Spoils

Both underground and surface mines generate mine spoils of varying physical and chemical composition. In the surface mine setting, the mine pits are reclaimed by the replacement of the spill material as well as in permitted cases by CCPs. The mine spoil material is composed of excavated overburden sandstone, clays, and shale as well as coal remnants. The mine spoil material can contain pyritic material and/or high-sulfur reject material. The mine spoil is a heterogeneous and anisotropic material leading to the presence of variable hydraulic conductivity. Aljoe and Hawkins (1994) reported that increasing percentages of sandstone and

decreasing percentages of shale in mine spoil appear to yield higher-hydraulic-conductivity values. This may be due to the presence of larger-sized rock fragments in the sandstone-rich spoil zones than shale-rich zones. Shales break into smaller fragments and readily weather to create silt and clay particles which yield lower hydraulic conductivities. Aljoe and Hawkins (1994) also reported that hydraulic conductivity of the mine spoils may increase with spoil age. They observed that reclaimed mine spoil 30 months or less old had a significantly lower median hydraulic conductivity than reclaimed mine spoil that was over 30 months old. Rehm et al. (1980) reported that spoil permeability decreases with age. Their results are from the northern Great Plains region, compared to the Aljoe and Hawkins (1994) results from eastern coal fields. Hydraulic conductivity for surface mine spoils exhibits a broad range of values. A summary table (Table 2.2) prepared by Hawkins (2002) captures the available hydraulic conductivity data for surface mine spoils.

2.4 Premined Area Water Quality

Groundwater and surface water quality data for premined areas are sparsely available. Strippable coal and lignite deposits are prevalent in the Denver and Laramie Formations in the Denver Basin. Kirkham and O'Leary (1980) reported water quality data collected from monitoring wells in Watkins (Adams/Arapahoe County), Bijou Creek (Elbert County), Keenesburg (Weld County), and Matheson (Elbert County) areas. These chemical analysis results are shown in Table 2.3. Brady et al. (1996) and Hawkins et al. (1996) have given a summary of shallow groundwater quality in unmined regions of the northern Appalachian Plateau.

Examination of these premined area groundwater quality results indicates the presence of low to quite high concentrations of sulfate, sodium, iron, manganese, and total dissolved solids (TDS) in groundwater with a significant amount of alkalinity and neutral to alkaline pH.

2.5 Water Quality in Mined Areas

Numerous studies have been carried out around the United States to characterize water quality from surface and subsurface mined areas. Much of the water quality data deals with coal mine drainage. Coal mine drainage can be either acidic or alkaline. AMD is typically elevated in SO₄, Fe, Mn, Al, and a few other elements.

2.6 CCP Mine Placement Sites

A survey questionnaire was developed and sent out to solicit information on CCP mine placement sites. Follow-up phone calls and individual contacts were made to identify over 100 mine sites where CCPs have been or are being placed in the previously mined areas. As a result, Table 2.4 has been developed to list the mine sites where CCPs were or are being used to fill the mined areas. Most of the identified sites are either surface or underground coal mines. Environmental monitoring data were obtained for a subset of these sites and are summarized to the extent possible in this report.

3.0 GEOCHEMISTRY OF COAL MINE DRAINAGE AND INTERACTIONS WITH CCPs

3.1 Coal Mine Drainage

Coal mines represent an environmental setting where the mining of coal generates coal refuse and exposes pyritic material present in the coal. Coal mine drainage results when rainwater and drainage water undergo chemical reactions with the coal refuse, pyrites, and overburden material. AMD occurs by the oxidation of pyrite to release dissolved Fe^{2+} , SO_4^{2-} , and H^+ , followed by further oxidation of Fe^{2+} to Fe^{3+} , generating more H^+ . The oxidized Fe^{2+} precipitates as iron hydroxide solid giving rise to the orange–yellow color in the areas where the precipitates are deposited.

Coal mine drainage can be acidic or alkaline, typically with relatively high concentrations of sulfate, iron, manganese, and aluminum. The pH of the coal mine drainage water commonly is either in the range of 3 to 4.5 s.u. or 6 to 7 s.u. (standard unit). Obviously, more abundant pyrite material in the overburden correlates to a more acidic pH of the coal mine drainage.

The major source of acidity is, therefore, due to the oxidation of pyrite that gets exposed to humid air and/or aerated water when mining activities break rocks and move the overburden material. AMD water contains elevated levels of dissolved SO_4 , Fe, Al, and Mn, is typically low in pH, and may have elevated concentrations of arsenic, copper, nickel, and zinc (U.S. Bureau of Mines, 1994; Caruccio et al., 1994, Evangelou, 1995). Caruccio and Geidell (1978) reported that pyrite can range in grain size between 400 μm and 5 mm. Oxidation of pyrite is a complex set of reactions involving both abiotic geochemical and biotic microbial reactions (Figure 3.1). At low pH (<4.5), pyrite is oxidized by Fe^{3+} much faster than by oxygen.

Neutral or alkaline mine drainage has alkalinity that equals or exceeds acidity because of the neutralization reactions with carbonate minerals such as calcite and dolomite. The alkalinity imparted by the dissolution of carbonate minerals promotes the removal of Fe, Al, and other dissolved metals while neutralizing the acidity. The neutral or alkaline mine drainage water still contains elevated amounts of sulfate. Table 3-1 shows median postmining water quality data for several mine sites in Pennsylvania to illustrate the acidic and alkaline coal mine drainage water quality.

3.2 Interactions Between CCPs and Coal Mine Drainage

If acidic CCPs are placed in coal mines with AMD, then the low pH and acidity in the AMD water could increase leaching or release of some metals in the CCPs. When alkaline CCPs are placed in coal mines with AMD, then the AMD is neutralized by the CCPs. The primary base cations in CCPs that provide alkalinity are calcium and magnesium. The dissolution of the base cations from CCPs results in neutralization and increase in pH of the AMD, which in turn decreases the concentrations of other heavy metals in the AMD.

The most obvious effect of alkaline CCPs on AMD is the reduction of iron. As soon as the pH of the neutralized AMD water is about 6.5 and higher, iron solubility decreases and iron oxide or iron hydroxide precipitates are formed as red or orange flocculants. The formation of iron hydroxide further helps in the coprecipitation/adsorption of metals such as As, Ni, Cd, and Cr.

However, most of the sulfate found in the AMD water is likely to remain in solution, although some precipitation of sulfate may occur when enough calcium or barium is present to precipitate gypsum or barite.

The net effect of placing alkaline CCPs in abandoned coal mines is expected to be beneficial in most situations. The pH of the water is likely to increase, and concentrations of most metals in the AMD are expected to decrease.

4.0 CCPs IN COAL MINES

Beneficial use of CCPs for coal mine reclamation occurs in varying degrees across the United States. Injection of CCPs into deep mines has been performed to provide structural support for subsidence abatement, and placement of CCPs in surface mines has been utilized to reclaim mined land to original grade and to mitigate AMD. Such practices have been employed at both active and abandoned coal mines.

Several surface and deep mines in the midwestern and western United States that have utilized CCPs for reclamation are identified in Table 2.4. These mines include Keenesburg and Trapper Mines (CO), Wyodak Mine (WY), Savage Mine (MT), Midwestern Abandoned Mine and Arnold Willis City Mine (IN); Big Gorilla Pit (PA); Winding Ridge Frazee Mine (MD); and Universal Mine (IN). Active mining operations continue to occur at Trapper, Wyodak, and Savage Mines. Types of CCPs placed at these mines include fly ash, bottom ash, and FGD materials.

Several case studies to assess the advantages and disadvantages associated with beneficial use of CCPs in mine reclamation are presented in this section of the report. These mines were selected based on availability of site-specific data required to perform a reasonable evaluation of the benefits and impacts of CCP placement on groundwater quality. Tables and graphs are included, when available, to illustrate important aspects of each case study. Following is the list of case studies included in this report:

1. Wyodak Mine, WY
2. Keenesburg Mine, CO
3. Trapper Mine, CO
4. Savage Mine, MT
5. Storm Strip Mine, WV
6. Universal Mine, IN
7. Midwestern Abandoned Mine, IN
8. Arnold Willis City Mine, IN
9. Harwick Mine Complex, PA
10. Clinton County, PA
11. Big Gorilla Pit, PA
12. Red Oak Mine, OK
13. Winding Ridge Frazee Mine, MD

4.1 Wyodak Mine (WY)

4.1.1 Background

The Wyodak Mine is an active, above-ground coal mine located approximately 5 miles east of Gillette, Wyoming. Approximately 69 million tons of coal was mined at Wyodak between 1922 and 1990, and the estimated remaining minable quantity of coal within the permit boundary exceeds 100 million tons. The total permitted area of the mine is 5811 acres. The Wyodak coal seam is classified as subbituminous and ranges between 20 and 100 feet in thickness. The overburden thickness above the coal seam ranges from 20 to 200 feet, with thin overburden (i.e., 20–100 ft) in the southern area of the site and thicker overburden (i.e., 100–200 ft) in the northern area of the site.

Placement of fly ash, bottom ash, and scrubber ash in mined areas at Wyodak Mine began in 1978 and is expected to continue through 2009. The ash is received from three nearby power plants, i.e., Neil Simpson I, Neil Simpson II, and Wyodak. Annual quantities of each ash type generated at the three power plants are listed in Table 4.1.

Bottom ash currently is slurried to an on-site disposal pond that is dredged periodically to remove material for use as mine backfill. Fly ash from the Neil Simpson I power plant is mixed with bottom ash and either slurried to the disposal pond or placed directly in the pits. The commingled fly/bottom ash also may be intermixed with spoil material prior to backfilling. Scrubber ash (i.e., mixed fly ash and FGD sludge) is obtained from the Wyodak and Neil Simpson II power plants and is placed directly as pit backfill. Upon completion of ash placement, backfilled mine areas are covered with soil and revegetated. Approximately 5,000,000 yd³ of ash has been placed in 13 separate pits listed in Table 4.2.

Active ash placement operations are ongoing in Area 4 (1998–2009) near the center of the site at an annual placement rate of nearly 250,000 yd³. Ash-filling operations began at the base of each mined-out pit. As indicated in Table 4.2, pits filled from 1980 to 1982 in Area 2 represent “unencapsulated” cells in which no soil liner was constructed at the base of the cells prior to ash placement. In the southern area of the site, pits filled with ash prior to 1980 (Area 1) and after 1983 (Area 3) were lined with compacted clay materials selected from appropriate overburden, interburden, and/or underburden soils. Clay liners with a maximum hydraulic conductivity of 5×10^{-7} cm/s were required for these mines as a result of a relatively thin layer of overburden (i.e., less than 20 feet) between the mined-out pit floors and the groundwater table. The unlined pits (Area 2) and the lined pit filled in 1983 (Area 3) are located beneath the floodplain of Donkey Creek.

4.1.2 Local Geology

The Wyodak site is located within the Powder River Basin. The Eocene Wasatch Formation comprises the overburden at the site ranging from 10 to 200 feet in thickness and consisting primarily of fine-grained clays and siltstones with discontinuous cross-bedded sandstone and scoria beds. The scoria beds were formed from baked and fused shale and clay in the Wasatch. The Wyodak coal seam forms outcrops in the eastern side of the permit region and

generally comprises the uppermost Paleocene Fort Union Formation unit. Approximately 400 feet of interbedded claystone, siltstones, and shale underlies the coal. The coal and underlying sedimentary units form outcrops in the eastern portion of the site.

4.1.3 Local Hydrogeology and Hydrology

The shallow hydrologic system in the Powder River Basin primarily comprises five permeable units: 1) lenticular sands in the Fort Union Formation, 2) the Wyodak coal seam, 3) lenticular sands in the Wasatch Formation, 4) scoria beds in the Wasatch formation, and 5) Quaternary alluvium. Each of these water-bearing units produces water at the site. The most significant source of shallow groundwater within the Fort Union Formation at the site is a sandstone unit that is less than 100 feet thick and occurs 360 to 380 feet beneath the Wyodak coal seam. Interbedded claystones, siltstones, and shale are located between the coal seam and sandstone unit. The Wasatch Formation is saturated in the mine permit area, although it is not considered to be a significant source of groundwater because of its relatively low permeability. Secondary permeability created by fracturing has produced yields for some domestic and stock wells. However, groundwater from the coal beds on-site typically is not suitable for domestic use.

Surface water hydrology within the mine permit boundary is controlled by Donkey Creek, which is located in the southern portion of the permit region and has a drainage area of about 77 square miles upgradient of the permit area. Donkey Creek flows to the east and is a tributary of the Belle Fourche River.

4.1.4 Ash-Leaching Characteristics

Laboratory tests on fly ash and bottom ash samples were performed on several occasions over the past two decades in support of reclamation activities at the Wyodak Mine. In 1983, a laboratory column leaching study was performed on crushed Wyodak bottom ash and overburden material (i.e., mine spoils). The test was performed first by permeating site groundwater through a column of overburden material and analyzing the effluent for the constituents listed in Table 4.3 to establish representative postmining water quality conditions in the absence of CCP placement. The effluent from the overburden column then was passed through a column of crushed bottom ash and analyzed for the same chemical constituents.

The results provided in Table 4.3 indicate that effluent from the overburden meets Wyoming Class III livestock use standards for all constituents, but exceeds Class I domestic use standards for TDS, Mn, NH₃, and SO₄. Permeation through the bottom ash resulted in an increase in some of the regulated constituents, including pH, TDS, B, Cl, Cr, Cu, NO₂/NO₃, and SO₄. Effluent from the bottom ash meets Class III standards for all constituents with exceedances occurring for B (6.97 mg/L relative to 5.0 mg/L) and SO₄ (3060 mg/L relative to 3000 mg/L). Concentrations of several constituents decreased upon permeation through the ash. For example, Mn exceeded the Class I domestic use standard (0.13 mg/L relative to 0.05 mg/L) in effluent from the overburden column but was not detected in the effluent after exposure to the ash.

Additional laboratory leaching analyses of Wyodak bottom ash and scrubber ash were conducted in 1997 and 1998, respectively, using a sequential batch extraction procedure in which ash samples were mixed with site groundwater at an ash-to-water ratio of approximately 1:20 (i.e., 100 g ash:2000 mL groundwater) (Aqua Terra 1997, 1998). This ash-to-water ratio in the batch test is not representative of expected field leaching conditions as compared to the ash-to-water ratio in a column test. However, the batch method of testing is considered to yield more conservative leaching results than column testing, since more of the ash surface area is exposed to the groundwater.

The results of the Wyodak ash batch leaching tests, summarized in Table 4.4, indicate that pH values tend to be elevated relative to the groundwater prior to testing. The results further indicate that maximum postextraction concentrations of all constituents in the bottom ash test, except for Mn and NH₃, are below current Wyoming Class III livestock-watering standards. Concentrations of both Mn and NH₃ in the groundwater also exceeded the Class III limits prior to ash exposure. As observed in the column test, the concentration of Mn in the groundwater decreased from 1.29 mg/L before ash exposure to 0.07 mg/L after ash exposure, a value that is only slightly higher than the Class III standard of 0.05 mg/L.

The postextraction groundwater in the scrubber ash test was above Class III standards for pH, Cr, F, Fe, Mn, and NH₃. However, concentrations of Fe, Mn, and NH₃ in the groundwater exceeded Class III standards prior to ash exposure. In both tests, postextraction concentrations exceed Class I domestic use standards for TDS, B, Mn, NH₃, and SO₄. Of these constituents, only B was below the Class I standard in the groundwater prior to ash exposure.

4.1.5 Groundwater Quality

Groundwater quality at the site is present within alluvium, upper and lower coal, Wasatch, scoria, Fort Union Formation, and sandstone (beneath the coal) units. Groundwater quality from the coal is classified as calcium sulfate-type water with significant bicarbonate and magnesium content. Groundwater derived from the coal is used for livestock purposes and is considered to be of better quality in the northern and eastern portions of the site where the coal seam is not split. Groundwater derived from scoria at the site is magnesium–sodium sulfate-type water, and alluvial groundwater is magnesium–sodium–calcium–sulfate-type water.

Monitoring wells are distributed within the different units both inside and outside of the permit boundary and also within or near ash-filled pits. Well LF-5 is located in close proximity to unlined pits in Area 2, and Well M-11 is located within the lined pit in Area 3 that received ash in 1998 and 1999. Well M-9 is located in Area 4, which comprises the largest unlined pit and is scheduled for ash placement until 2009. It is unclear which portions of Area 4 have already received ash. Wells M-1, M-1A, M-2, M-3, M-4, M-5A, and M-8 are located in an unmined region within the permit boundary to the north of Area 4. This area may be filled with ash in the future. Wells M-6, M-6A, M-16, M-17, M-18, M-19, M-22, M-23, and M-29 are located along the southern perimeter of the site outside of the mine permit boundary.

Results of groundwater quality analyses for samples collected from several of the Wyodak mine area monitoring wells are available for the period 1981 to 1999. Metal analyses also were

performed on a more limited basis from 1977 to 1981. These results are utilized herein as a basis for evaluating the groundwater quality at the Wyodak site. Groundwater quality information prior to mining operations is not available.

A summary of groundwater quality for regulated analytes in Wyoming is presented in Table 4.5 for selected wells. Wells were selected on the basis of data availability and well proximity to ash-filled pits in order to provide a representative cross section of groundwater quality across the site (e.g., mined versus unmined regions, upgradient versus downgradient locations). Local groundwater gradients appear to follow local topographic trends northward across the site. Monitoring Wells M-1A and M-5A were selected to represent the northern unmined area downgradient of ash backfill areas. Monitoring Well M-12 is located upgradient of ash deposition activities in the southern portion of the site. Off-site monitoring Wells M-17 and M-22 are located upgradient of mining and ash backfill areas. Monitoring Well LF-5 is located adjacent to unlined ash fill in Area 2. Although numerous wells are located within Areas 1, 2, and 3, limited analytical data are available for these wells.

Historical maximum and mean value analyte concentrations in groundwater samples from the selected wells are provided in Table 4.5. The Table 4.5 results indicate that the average groundwater quality throughout the Wyodak site compares favorably with the Wyoming Department of Environmental Quality (DEQ) Class III (livestock use) standard. Mean concentrations for all of the measured constituents in wells are at or below Class III standards. The only possible exception is mercury (Hg). Unfortunately, no valid conclusions can be made regarding whether or not mean Hg concentrations exceed the Class III standard of 0.00005 mg/L, since the reported detection limit for Hg in most of the sampling events was 0.001 mg/L. However, maximum Hg concentrations (i.e., maximum of all samples shown in Table 4.5 – Wyodak Mine groundwater quality summary for selected monitoring wells analyzed over time for a particular well) in excess of the Class III standard were observed in all of the selected wells, including the off-site, upgradient wells (M-17 and M-22). These results indicate that elevated Hg levels may be present in upgradient groundwater for this area.

The results in Table 4.5 also illustrate that periodically elevated concentrations of some constituents were observed in wells located near the southern ash-filled pits. For example, the maximum concentrations of TDS and chromium (Cr) measured in Well LF-5 from 1987 through 1999 exceed current Wyoming DEQ Class III standards. Concentrations of As, Cl, Mn, and SO₄ do not exceed Class III standards but appear to be elevated in Well LF-5 relative to the other wells. Well LF-5 is located in the backfill near the unlined ash-filled pits.

Elevated maximum concentrations of Cr and B were observed from 1977 to 1981 in Well M-12, located upgradient of the ash-filled pits in the southern portion of the site. However, these elevated Cr and B concentrations likely are not due to ash placement, since pits closest to M-12 were not filled until 1985 or later. Also, historically high Cr concentrations in off-site, upgradient Well M-17 indicate that elevated Cr may be naturally occurring in this area.

Temporal groundwater quality trends observed in Wells M-1A, M-5A, LF-5, M-17, and M-22 from approximately 1981 to 1999 for B, SO₄, Mn, and TDS shown in Figures 4.1–4.4. The results indicate that SO₄, Mn, and TDS concentrations tend to be elevated in Well LF-5, the

nearest well to the unlined pits in the southern portion of the site, relative to the upgradient and downgradient wells. However, LF-5 concentrations in excess of the Wyoming Class III standards were observed only for TDS. Also, lower values of these constituents in downgradient Wells M-1A and M-5A indicate that ash placement has not resulted in significant impacts to regional groundwater quality within the permit boundary to date.

4.2 Keenesburg Mine (CO)

The Keenesburg Mine is located approximately 7 miles north of Keenesburg, Colorado, in Weld County, Sections 25, 26, 35, and 36 of Township 3 North, Range 64 West. The permit area was recently extended to include Sections 2, 11, and 14, Township 2 North, Range 64 West, in 2000 (Correspondence dated 10-30-00 from Byron Walker of the State of Colorado DNR to Colorado Department of Health HMWMD). Coors Energy Company operated coal extraction at the Keenesburg Mine between 1980 and 1988. Approximately 1000 tons was mined from the site daily (Correspondence dated 11-14-86 from John Althouse of Coors Energy Co. to Weld County Commissioners).

Reclamation at the Keenesburg site is being performed using fly ash and bottom ash derived primarily from Keenesburg coal and generated by the Coors Brewery in Golden, Colorado. Ash placement operations began at the Site A pit that is configured in an east–west direction approximately within the mining boundary established in 1985. Ash also was deposited in the north and south ends of B pit after completion of mining operations in 1988. As of 1993, approximately 172,000 tons of ash had been placed in A and B pits at a rate of nearly 30,000 tons/yr. Additional ash placement sites were also utilized between 1987 and 1989. Ash was placed at least 5 feet above the premining groundwater table, and reclamation plans also include placement of a vegetative final cover consisting of at least 5 feet of compacted overburden and 3 feet of topsoil material.

4.2.1 Local Geology

The Keenesburg Mine is located within the Denver structural basin. Documented geologic units at the site include the following, in descending stratigraphic order: quaternary deposits, the Laramie Formation, the Fox Hills Sandstone, and the Pierre Shale. The total thickness of these sedimentary units at the site is estimated at 10,000 feet (Doty and Associates, 1998). Quaternary deposits include Ennis Draw alluvium and dune sand. The Laramie Formation is considered the uppermost bedrock unit at the site and is comprised of yellow–brown and gray to blue–gray, soft, carbonaceous shale and clay–shale interbedded with sand and shaley sand. The lower portion of the Laramie contains thick, continuous coal seams. The Fox Hills Sandstone consists of calcareous marine sandstone with dark gray to black sandy shale and some massive white sandstone. The thickness of the Fox Hills Sandstone reportedly ranges between 60 and 300 feet (Doty and Associates, 1998). Pierre Shale is considered the deepest water-bearing unit of interest at this site. This unit comprises fossiliferous marine shale, silt, and clayey sandstone and is estimated to be over 8000 feet thick.

4.2.2 Local Hydrology

The major aquifer at the Keenesburg site is the Laramie–Fox Hills Aquifer, with sandstone at the base of the Laramie and the upper portion of the underlying Fox Hills Sandstone. Groundwater flow across the site prior to mining was to the northeast at a prevailing hydraulic gradient of 0.002. Recharge to aquifers within the Denver Basin primarily occurs via outcrop infiltration, while shale deposits in the upper Laramie Formation largely prohibit deep vertical recharge. The estimated average hydraulic conductivity for the Laramie–Fox Hills Aquifer is 0.05 feet/day, with a transmissivity of 8 ft²/day (Doty and Associates, 1998). The average hydraulic gradient across the site is 0.002 ft/ft, resulting in an average groundwater flow velocity of about 0.2 ft/yr (assuming 15% porosity).

Sedimentary units that are considered to be of hydraulic significance at the site include the sand and alluvial deposits, overburden, coal, and clayey backfill material (mine spoils) (Doty and Associates, 1998). The sand deposits above the shale are unsaturated and have an estimated hydraulic conductivity of 1.0×10^{-3} cm/s. The Quaternary alluvium is saturated at the site with an estimated hydraulic conductivity of 10^{-4} cm/s. The water-bearing silty claystones and sandstones above the coal deposit in the upper Laramie Formation appear to be hydraulically connected to the alluvium. Coal deposits at the site are saturated, and the estimated geometric mean hydraulic conductivity of the coal is 9×10^{-5} cm/s. The hydraulic conductivity of the spoil material was not available.

4.2.3 Ash Chemical Characteristics

Chemical analyses were performed on fly ash and bottom ash samples prior to disposal at the Keenesburg Mine. For example, the results of a 1989 extraction analysis on saturated fly ash and bottom ash pastes with an ash-to-water ratio of approximately 1 g to 0.42 g are shown in Table 4.6 (Hazen Research Inc., 1989). The results indicate that the ash samples contain appreciable amounts of alkali/alkaline-earth metals (Ca, Na, K, Mg), sulfate (SO₄), and trace metals (e.g., B, Cr, Cu, Fe, Pb, Mo, Se, Zn). The fly ash appears to contain greater amounts of many of the analyzed constituents, most notably As, B, Cd, Cr, Mo, Se, and SO₄. The toxicity characteristic leaching procedure (TCLP) tests performed on fly ash and bottom ash samples indicate that the leached concentrations of the Resource Conservation and Recovery Act (RCRA) metals (Ag, As, Ba, Cd, Cr, Hg, Pb, and Se) would classify these ashes to be a nonhazardous solid waste as shown in Table 4.7 (Hazen Research Inc., 1991).

4.2.4 Groundwater Quality

Groundwater quality data were collected at the Keenesburg site since the late 1970s, primarily for monitoring Wells AMW-1, AMW-2, DH-96, DH-122, DH-133, DH-172, FPW, and SMW-2. Wells FPW, DH-96, DH-122, DH-133, and DH-172 were completed in the Ennis Draw alluvial aquifer. Wells SMW-2 and AMW-2 were completed in the reclaimed spoil, and Well AMW-1 was completed in undisturbed overburden. Assuming a prevailing northeastern groundwater flow direction, monitoring Wells DH-122 and DH-133 are located upgradient (south to southwest) of ash placement areas, whereas monitoring Wells AMW-1, AMW-2, FPW, DH-96, and SMW-2 are located downgradient (north to northeast) of ash placement areas.

Historic maximum and mean groundwater concentrations measured in monitoring wells prior to ash placement (i.e., 1978–1986) and after ash placement (i.e., 1988–2000) are compared to drinking water and agricultural standards from the Colorado Department of Public Health and Environment (CDPHE) Regulation No. 41 “Basic Standards for Groundwater” in Table 4.8. Regulated analytes for which site data are available include As, Ba, Cl, Fe, Mn, NO₃, Se, SO₄, and TDS. The results in Table 4.8 indicate that, on average, concentrations of these analytes historically have exceeded the regulatory standards in at least one of the site wells. Ba is the only analyte that has consistently remained below Colorado Groundwater Protection Standards both prior to and after the placement of ash.

However, upon comparison of water quality in upgradient versus downgradient wells and for sampling events prior to ash placement (1978–1986) relative to sampling events following ash placement (1988–2000), there is little evidence indicating that elevated levels of regulated constituents in site groundwater are a direct result of leaching from the ash. For example, elevated arsenic concentration (0.11 mg/L) was observed in both upgradient Well DH-122 and downgradient Well DH-96. Elevated concentrations of Fe, Mn, NO₃, and SO₄ also have been measured in upgradient Well DH-122. The results in Table 4.8 also indicate that average concentrations of Cl, Fe, SO₄, and TDS in downgradient Well SMW-2 for the years prior to ash placement are similar to, if not higher than, concentrations of these constituents measured in the years following ash placement. Thus the results for SMW-2 likely represent impacts to groundwater quality as a result of mining activities rather than placement of ash, which also may be responsible for the elevated concentrations of SO₄ and TDS measured in Wells AMW-1 (downgradient of B pit) and AMW-2 (downgradient of A pit) after 1988. Unfortunately, chemical analysis data prior to ash placement are not available for Wells AMW-1 and AMW-2.

Based upon the above analyses, the groundwater quality at the Keenesburg Mine does not appear to be impacted by placement of CCPs. Moreover, Coors Energy Company was chosen as the 1999 Small Operation Reclamation Award winner as a result of reclamation efforts at the Keenesburg Mine. Currently, the mine is operating under a reclamation-only permit.

4.3 Trapper Mine (CO)

The Trapper Mine is a surface coal mine located on the north slope of the Williams Fork Mountains, approximately 6 miles south of Craig, Colorado. The Trapper Mine began operation in 1977 and produces up to 2.8 million tons of coal annually. A majority of this coal is utilized by the Craig Generating Station, an electric power plant located near the mine.

Surface-mining operations were initiated in the Ashmore Pit (A Pit) in June of 1977. The Queen Anne, the first of three draglines to be put into service at Trapper during the late 1970s, began uncovering two coal seams in the westernmost portion of A Pit. The upper coal seam averaged from 4 to 6 feet in thickness and was typically encountered at overburden depths ranging from approximately 30 to 110 feet. The underlying coal seam varied from 5 to 9 feet in thickness and was separated from the upper seam by approximately 30 feet of overburden. Mining depths for the lower seam ranged from 50 to approximately 160 feet.

Deposition of CCPs in the mined out areas of A Pit began in 1984 with the first ash placement sites located in the Johnson drainage just east of the No Name watershed divide and the No Name drainage. CCP placement at these sites occurred from 1984 through 1986. Additional ash placement sites were subsequently developed just north of the original disposal sites in the Johnson drainage and along the eastern flank of the Johnson drainage. These locations served as ash deposition sites during 1987 and 1988. Ash deposition further south along the eastern flank of the Johnson Gulch drainage began in 1989, progressed to the top of the Johnson drainage in 1993, and followed the mine workings into the East Pyeatt portion of A Pit beginning in 1994 and continuing until today.

The Trapper Mine has managed approximately 390,000 tons of CCPs per year since 1984, with total disposal quantities approaching 7 million tons to date (personal communication with Karl Koehler, Trapper Mining Inc., 2001). Current ash placement practices involve deposition of sufficient overburden material into the pit bottom such that the ash is placed above the expected postmining groundwater table. Overburden materials are placed above the ash and revegetated as part of the final reclamation plan.

4.3.1 Hydrogeology

The geology in the vicinity of the Trapper Mine is dominated by the Williams Fork formation that represents the upper portion of the Mesa Verde Group in northwestern Colorado. The Williams Fork Formation is divided into a lower unit, Twenty Mile Sandstone (middle unit), and an upper unit. The Trapper Mine is located on an outcrop of this upper unit, which extends to a depth of approximately 850 feet in the vicinity of the Trapper Mine. Coal seams at the Trapper Mine are located within the upper 20 to 160 feet of the Upper Williams Fork Formation.

The Upper Williams Fork unit primarily consists of siltstones and shale and essentially acts as an aquitard to the Twenty Mile sandstone unit that represents the most productive aquifer in the area. Coal seams and sandstone units within the Upper William Fork Formation also may serve as aquifers for perched groundwater. The alluvial aquifer above the Upper Williams Fork Formation is associated with the Yampa River floodplain to the north and west of the mine site and has an estimated permeability of 67 to 670 ft/day (Radian Corporation, 1991). Based on groundwater levels measured in site monitoring wells in November 2000 (Trapper Mining Inc., 2001), the average groundwater flow direction is north to northwest across the site.

Groundwater recharge in the area occurs via direct infiltration, primarily from snowmelt. The rate of groundwater recharge is relatively low as a result of the climate, which is characterized by a lack of abundant precipitation and a high evaporation rate. Based on the results of a groundwater recharge study, recharge to shallow groundwater (assuming 4 feet of overburden) is estimated to be about 0.6 inches per year (Radian Corporation, 1991). Thus groundwater impacts resulting from infiltration through CCPs placed above the groundwater table are expected to be minor.

4.3.2 Ash Chemical Characteristics

Coal combustion products produced at the Craig Generating Station include fly ash, bottom ash, and scrubber sludge materials from combustion of bituminous coal in the boilers. Results of chemical analysis on composite samples of Craig Station fly ash and bottom ash are shown in Table 4.9 (Radian Corporation, 1991). These results illustrate that both the fly ash and bottom ash contain appreciable levels of several constituents, including Al, Ba, Ca, Fe, Mg, P, K, Si, Na, Sr, S, and Ti. Lesser amounts of other trace metals also were detected, including Ag, As, B, Cd, Cr, Cu, Mn, Mo, Ni, Pb, V, and Zn.

Analyses such as those given in Table 4.9 typically are performed by digestion of the ash samples in a strong acid solution (e.g., pH <2) and often are not representative of the expected leaching behavior of the ash, since many of the ash constituents may not be mobilized by leaching fluids that more closely approximate expected field conditions. For example, results of batch extraction tests with acetic acid (pH ~5.0) and distilled water are provided in Table 4.10. Although the results indicate that the fly ash and bottom ash tend to release measurable amounts of SO₄ and several trace metals (Ag, Al, B, Ba, Cr, Fe, Mn, Mo, Ni, V, Zn), comparison of constituent levels with current groundwater quality standards in Table 4.10 reveals that the fly ash tends to release elevated concentrations of only Al, B, Ba, and Cr based on distilled water extraction. No elevated concentrations were leached from the bottom ash. The alkaline nature of the ash materials is evident by final values of pH in the distilled water extraction tests that exceeded the acceptable pH range based on current standards (i.e., 6.5–8.5). Results of the RCRA extraction test indicate that the weak acid leaching tends to mobilize greater quantities of certain constituents from the fly ash. In particular, the concentration of SO₄ increased from <3 mg/L in the distilled water extraction to 90 mg/L in the RCRA extraction.

In order to better assess the potential impact of fly ash placement on groundwater quality, column leaching tests were conducted on fly ash and overburden materials. Column tests typically are more appropriate than batch tests for assessing potential groundwater impacts, since column tests are conducted at an ash-to-water ratio that is more representative of field conditions than the ash-to-water ratio employed in batch leaching tests. In this study, tests were conducted by passing deionized water through separate columns of Craig Station fly ash and unweathered overburden material. In addition, a third column test was conducted by passing deionized water through a fly ash column and subsequently passing the fly ash column effluent through an overburden column. The testing program provided the means to evaluate the individual leaching characteristics of the fly ash and overburden materials and to investigate the ability of the overburden to attenuate the migration of contaminants from the fly ash.

Effluent levels of pH, SO₄, Al, Ba, B, Cr, Mn, and V measured during the three column leaching tests are illustrated versus cumulative effluent volume in Figures 4.5–4.12. The results indicate that effluent from the fly ash exhibited consistently higher pH and higher concentrations of Al, B, Ba, and Cr than effluent from the overburden. Effluent concentrations of pH, Ba, and Cr from the fly ash column exceeded Colorado State drinking water standards throughout the test, whereas effluent levels of these constituents from the overburden material column essentially were below (or within) the standards throughout the test duration. The test results also indicate that subsequent passage of the fly ash effluent through the overburden resulted in

reductions in pH, Ba, and Cr to below drinking water standards. The overburden also appears to attenuate Al, but exhibits little ability to attenuate B released from the fly ash. Colorado drinking water standards currently do not exist for either Al or B, although regulated agricultural use limits of 5.0 mg/L and 0.75 mg/L, respectively, have been established. Based on these limits, placement of overburden in mined pits prior to placement of fly ash may not reduce the leaching of B from the ash.

The test results also indicate that concentrations of SO₄, Mn, and V were as high or higher in effluent from the overburden material column relative to the fly ash. The tendency for these constituents to be released from the overburden is illustrated further by the similar leaching trends exhibited in the overburden column test and the test in which the water is passed through the fly ash prior to passage through the overburden. Of these three constituents, SO₄ is the most difficult to assess in terms of groundwater impacts related to ash placement, since initial SO₄ leached from the fly ash and the overburden material are similar. The results in Figure 4.6 show that SO₄ levels may be expected to decline over time to levels below the Colorado drinking water standard (250 mg/L). However, leached concentrations of SO₄ in the early time were higher than the standard. In addition, SO₄ may be released in greater concentrations from the bottom ash based on the batch test data (see Table 4.10), as well as from the scrubber ash that tends to contain more sulfur than both fly ash and bottom ash (Radian Corporation, 1991).

4.3.3 Groundwater Quality

Groundwater quality monitoring has been performed at the Trapper Mine since 1982 but primarily since 1984, after ash deposition operations were initiated. Historic groundwater quality data are evaluated based on eight groundwater-monitoring wells identified as GD2, GD3, GF4, GF5, GF6, GF7, GF11, and GMP1. Wells GD2, GD3, GF6, and GF11 are upgradient of ash areas, whereas Wells GF4, GF5, GF7, and GMP1 are downgradient of ash disposal. Samples have been collected since 1982 in Wells GD2, GF4, and GF6. Sampling efforts were initiated in the remaining wells between 1984 and 1990. Groundwater samples were analyzed for Al, As, B, Ba, Cd, Cl, Cr, Fe, Mn, Mo, NH₃, NO₂+NO₃, Pb, pH, SO₄, and TDS. Historic maximum and mean concentrations of these constituents in each of the eight wells are listed along with the respective Colorado groundwater quality standards in Table 4.11.

Comparison of the historic groundwater concentration data with Colorado standards in Table 4.11 indicate little or no evidence of groundwater impacts associated with most of the analyzed constituents, including Al, As, B, Cd, Cl, Cr, NO₂ + NO₃, and Pb. Historic mean concentrations of iron (Fe) also are not elevated in any of the wells, although elevated individual values have been observed periodically in groundwater Wells GD2, GD3, GF4, GF5, and GF7. None of the historic Fe concentrations measured in any of the wells exceed the agricultural use standard (5.0 mg/L).

Of all the constituents monitored at the Trapper Mine, Mn, SO₄, and pH have consistently been observed at levels in excess of drinking water standards in Colorado. For example, pH values in excess of the upper limit established by Colorado drinking water standards (i.e., 8.5) historically have been measured in several of the wells, most notably in downgradient Wells GF4 and GF5. Historic maximum and mean values of pH are 9.3 and 8.86, respectively, in Well GF4,

and 9.15 and 8.06, respectively, in Well GF5. Elevated pH values in GF4 and GF5 may represent a groundwater impact related to ash placement, considering the locations of these wells relative to ash areas and the tendency for leachate from the fly ash to exhibit pH values between 10 and 12 (See Figure 4.6).

Excessive mean historic concentrations of Mn have been observed in upgradient Wells GD2, GD3, and GF11 and in downgradient Wells GF4 and GF7. The presence of elevated Mn in both upgradient and downgradient wells suggests that Mn levels reflect the overall postmining groundwater quality at the site and not an impact attributable to ash areas. This observation also is supported by the column leaching test results for Mn in Figure 4.7 that show the tendency for Mn to be mobilized from the mine spoil rather than from fly ash.

Elevated mean historic concentrations of SO₄ are evident in upgradient Wells GD3 and GF11 and in downgradient Well GF7. Historically high SO₄ levels in both upgradient and downgradient wells indicate that elevated SO₄ is characteristic of the overall postmining groundwater quality at the site. However, the historic mean SO₄ concentration in downgradient Well GF7 (1,581 mg/L) is considerably higher than the historic mean SO₄ concentrations in upgradient Wells GD3 and GF11 (435 mg/L and 569 mg/L, respectively). Given that the ash represents a significant source of sulfur (see Table 4.9) and mobilization of SO₄ from ash samples has been observed in leaching studies (see Table 4.10 and Figure 4.6), the higher concentration of SO₄ in Well GF7 may be due, in part, to ash disposal operations. However, the column test results also show that significant SO₄ may be mobilized from the mine spoil.

4.4 Savage Mine (MT)

Knife River Corporation's Savage Mine is located approximately 20 miles southwest of Sidney, Montana, and about 4 miles northwest of Savage, Montana, close to the North Dakota border. The Savage Mine has been in operation since 1958 as a surface lignite mine, and it currently produces 250,000 tons of lignite annually. The mine primarily serves the Imperial Holly Sugar Corporation plant in Sidney, Montana.

The Savage Mine began utilizing fly and bottom ash generated by Holly Sugar as backfill in 1987. The annual volume of ash received by the Savage Mine is variable and dependent upon the amount of coal burned by Holly Sugar in relation to the quantity of sugar beets processed. The ash disposal permit does not allow annual ash disposition in the mine to exceed approximately 10,000 cubic yards (about 16,500 tons). The estimated cumulative volume of ash placed between 1987 and 2000 is approximately 130,000 cubic yards. In order to achieve the maximum dilution of ash with spoils, the ash is distributed over as much pit length as possible. The typical annual pit length requirement is 700 feet for storage and disposal activities. Ash placement at the Savage Mine is performed in accordance with the following procedures:

- Ash placement occurs above the saturated zone of the anticipated postmine groundwater table. This level is achieved by placing 25 to 30 feet of ash-free spoils material at the pit bottom prior to ash placement.
- The dragline distributes the ash over the length of the pit.

- The dragline alternates buckets of ash with as many buckets of spoil material as possible to achieve maximum mixing and homogeneous dilution.
- Ash disposal will be kept to a minimum depth of 8 feet below the final postmine surface.

4.4.1 Local Geology

The regional geology surrounding the Savage Mine is characterized by broad gravel-capped uplands between wide troughlike valleys. At least five terrace gravel deposits have been identified in proximity to the Savage Mine and are attributed to river deposits that were transported into the eastern portion of Montana from mountains in the western and central parts of the state. The permit area of the Savage Mine is located on the east-central portion of a remnant terrace from the Yellowstone River known as Breezy Flat.

Surficial sediments in the vicinity of the Savage Mine are derived from the Tongue River Member of the Fort Union Formation primarily comprising light yellow to gray, massive to cross-bedded sand and light gray, silty clay. These unconsolidated deposits ranging from about 30 to 60 feet thick form approximately one-half of the total overburden thickness at the site. The lower portions of the site overburden comprise mostly gravel and sand. The Tongue River Member also contains several thick and laterally continuous lignite beds and limestone. Lignite from the Tongue River Member is removed during mining operations. The lignite ranges in thickness from 15 to 25 feet.

4.4.2 Local Hydrogeology

Groundwater is derived primarily from sands and gravels that cap terraces in areas around the Savage Mine and serves both livestock and domestic needs. Typical shallow aquifer yields are on the order of 5 to 20 gallons per minute (gpm). Groundwater is considered to be of good quality, with sandstone aquifers producing groundwater with softer and more mineralized characteristics as compared to the lignite aquifers.

The permit area is recharged by direct infiltration during snowmelt and rainfall events and by lateral underflow from adjacent unmined areas. Transmissivities calculated from pumping test data collected from Wells OB-1 and OB-2 are from 0.13 to 16 ft²/day. The limited transmissivity of the Pust coal has been attributed to a limited recharge area, highly variable topographic relief of the coal bed, and limited fracturing in the coal.

The stratigraphy of the Fort Union Formation and the overlying sediments at the Savage Mine and adjacent regions dictate local groundwater flow patterns. Potential water-bearing units disturbed by mining operations include the surficial sand and gravel and the Pust coal. Postmining protocol involves replacing these units with a saturated zone at the base of the spoils. The surficial sand and gravel at the site do not serve as an aquifer because of the lack of adequate recharge area, thickness, and rainfall. The unmined coal in the western part of the site is thought to form a continuous aquifer with the base of the mined-out spoil. The prevailing groundwater flow direction at the site is to the south/southeast.

Groundwater elevations in wells completed in the Pust Coal upgradient of the mine operations have remained relatively stable since 1986. Groundwater levels in Wells 261 and 284 generally have fluctuated less than 1 foot over the period of record. Groundwater levels in Wells OB-1 and OB-2 located immediately west of the mine activities have been influenced by mine-dewatering operations and have declined approximately 3 to 6 feet, respectively, over a 7-year period. Groundwater elevation declines of approximately 2 to 3 feet have been observed over the past 8 years in wells screened in mine spoils. Once mine operations cease, groundwater levels are anticipated to rise as a result of discontinuation of groundwater-dewatering operations. It is also expected that the spoils will not serve as a dependable source of water supply under postmining conditions.

4.4.3 Ash-Leaching Characteristics

Batch-type laboratory leaching analyses on Savage coal ash are performed annually prior to placement in the surface pits. The tests are performed by mixing ash samples with water at an ash-to-water ratio of 1:1 (by weight). The test results from 1987 to 2000 and corresponding water quality standards for a range of constituents are summarized in Table 4.12. The results indicate that pH values of the leachate tend to be elevated relative to water, which is beneficial for limiting the generation of acidic leachate from the spoils. Also the average and maximum leached concentrations of most constituents, most notably As, Cr, and Pb, are below current water quality standards. On average, federal and/or state standards are exceeded for Cl, NO₂+NO₃, and SO₄, while elevated individual values for pH and Cd have been measured. Boron (B) is present in the leachate samples. However, federal or Montana water quality standards are not established for B.

4.4.4 Groundwater Quality

Approximately 36 wells have been identified both outside and within the mine permit boundary. All of these wells except 003, 262, OB-1, and OB-2 have temporal groundwater quality information, predominantly metals analysis. Wells 108, 108A, and 109 are no longer in service because mining operations required the removal of these wells. Wells located upgradient of the site include ETZEL NORTH, LANGE NORTH, LANGE SOUTH, 261, 262, 263, and 284. Wells located downgradient of the ash backfill areas include Wells 002, 003, 005, 286, 289, 290, 291, and ETZEL SOUTH.

General groundwater quality has been characterized in a report for ash placement in the mine. Pust coal groundwater is considered to be of very high quality for eastern Montana and ranges from slightly basic to slightly acidic (i.e., <6.4 to 7.7). The groundwater also is classified as calcium, magnesium bicarbonate type water. Sulfate levels in groundwater for spoils tend to be elevated in proportion to bicarbonate. Groundwater in contact with spoils is more acidic, with a pH range of 6.3 to 7.0 in comparison to the groundwater contained in coal aquifer.

Groundwater metals analyses have been conducted on an annual basis for monitoring wells both upgradient and downgradient of ash placement. Analyses have been conducted since 1979 for TDS, SO₄, Cl, NO₃, Ca, Mg, Na, Fe, and Mn, while trace metal analyses (i.e., As, B, Cd, Cr, Pb, Ni) have been performed for post-ash placement conditions since 1986. Trace metals were

not analyzed prior to ash placement. However, because trace metal analysis was performed in laboratory leaching studies on the ash (see Table 4.12), qualitative assessments can be made regarding ash constituent contribution to groundwater quality.

4.4.4.1 Trace Metals

Historical groundwater concentrations (maximum and mean) of As, B, Cd, Cr, Pb, and Ni for monitoring Wells 001, 106, 261, 263, 284, 286, 289, 290, and 291 from 1986 to 2000 are compared to Montana groundwater quality standards in Table 4.13. The results indicate that, on average, trace metal concentrations are below current Montana State maximum contaminant levels (MCLs) in all of these wells, with the exception of arsenic in Well 291. The historic mean concentration of arsenic in Well 291 (0.20 mg/L) is an order of magnitude higher than the current MCL (0.02 mg/L). The MCL for arsenic also has been exceeded on occasion in Well 290. Well 290 is approximately 700 feet downgradient (east) of Pit 3, and Well 291 is located immediately downgradient (east) of Pit 4. However, it is unclear whether the elevated arsenic levels in these wells are due to release of arsenic from the ash. Historic ash-leaching test results given in Table 4.12 indicate that the amounts of arsenic mobilized from the ash are not sufficient to account for the elevated arsenic concentrations in Wells 290 and 291, particularly considering that the ash is mixed with mine spoils during placement. Although no chemical analyses of the spoils are available for this study, the spoil material itself may contain arsenic that could be released into site groundwater.

Historic arsenic concentrations in Wells 286, 289, 290, and 291 versus time are plotted in Figure 4.13. Well 290 demonstrated slight arsenic exceedances of the Montana MCL in September 1995 (0.021 mg/L) and in October 2000 (0.023 mg/L), whereas elevated levels of arsenic has been observed in Well 291 for nearly all of the sampling events since 1986. Overall, arsenic levels in Wells 290 and 291 have fluctuated over time but generally have increased since ash placement. Surprisingly, arsenic levels in downgradient Wells 286 and 289 historically have remained below the Montana MCL, even though these wells also are located within 500 feet of the pits.

The results in Table 4.13 also indicate that concentrations exceeding the Montana MCL were observed periodically for Cd and Pb in nearly all of the selected wells, including upgradient wells. Thus elevated Cd and Pb concentrations in downgradient wells likely are not due to leaching and migration of these constituents from the ash. However, no conclusions can be made regarding the impacts of ash placement on groundwater quality since the reported detection limit (i.e., 0.01 mg/L) for Cd in most of the historic analyses is higher than the current Montana MCL (i.e., 0.005 mg/L). Elevated nickel was observed during a single sampling event in upgradient Well 284 and in downgradient Well 106. Chromium has not been detected in any of the wells since 1986.

4.4.4.2 Other Constituents

Analyses of SO₄, Cl, NO₃, Ca, Mg, Na, Fe (total and dissolved), and dissolved Mn were performed for most of the monitoring wells. Other analyses on groundwater from these wells included pH, temperature, electrical conductivity, and TDS. Consistently elevated historical

concentrations of TDS, SO₄, and Fe have been observed in both upgradient and downgradient wells. The state of Montana has not established MCLs for these constituents.

Maximum and mean historical concentrations of TDS and sulfate measured in selected wells prior to and after ash placement in the mined pits are compared with the federal secondary MCLs in Table 4.14. The majority of monitoring well data shown in Table 4.14 indicates that TDS and/or SO₄ levels are consistently above the secondary MCL before and after ash placement. Elevated concentrations of TDS and SO₄ also are evident in both upgradient and downgradient wells relative to the ash-filled pits. These results indicate that the high-TDS and SO₄ concentrations are not due solely to the release of constituents from the ash but are indicative of postmining groundwater quality that is commonly encountered at other coal mine sites. However, increases in TDS and/or SO₄ levels have been observed in certain downgradient wells since ash placement operations began in 1986, most notably in Wells 286 and 291 along the western edge of Pit 4.

Iron (Fe) was detected at levels above the national secondary MCL of 0.3 mg/L for monitoring Wells 001, 105, 106, 110, and 111 prior to ash placement. Although measured Fe concentrations in the years following ash placement are not available in many of the wells, Fe concentrations in downgradient Well 001 decreased from an average of 1.32 mg/L prior to ash placement to 0.17 mg/L after ash placement. Overall, no obvious groundwater impacts due to release of Fe from the ash are evident at the site based on the available data.

4.5 Storm Strip Mine (WV)

This case study involves placement of coal ash in an active coal strip mine. The downgradient groundwater-monitoring data reveals no evidence of contamination over the 10-year period of operation.

Coal ash (both bottom ash and fly ash) is being placed at a rate of about 800,000 tons annually in an active strip mine near an electric power plant in West Virginia. The strip mine provides coal for the power plant and is located on the plant property. Ash placement began in 1987. No ash has been placed below the water table because the groundwater table is deeper than the mine floor. Monitoring Well MW1 is located deep underneath the land surface and is regarded as an upgradient well. No strip mining of coal has taken place near this well, which is screened in rock and coal about 130 ft below the ground surface. GWM2 is located upgradient of the ash fill area, and Wells MW6 and MW7 are located downgradient of the ash fill area. GWM2 is screened 40 ft below ground surface also in rock and coal. MW6 and MW7 wells are screened 25 ft and 60 ft, respectively, below ground surface. Groundwater quality-monitoring data from the four wells have been collected since 1987 for several metals, pH, alkalinity, conductivity, and sulfate. The Solid Waste Division of the state of West Virginia has issued a permit for a landfill involving the placement of coal ash in the strip mine.

Summary statistics of the monitoring results for eleven chemical parameters for aluminum, iron, pH, sulfate, and arsenic data trends or changes in these water quality parameters which may be associated with the ash placement in the coal strip mine. Aluminum concentrations in Wells MW1 and MW7 were below detection limits. Well MW6 shows aluminum below detection

limits with occasional measured concentrations between 0.2 and 0.6 mg/L. Only Well GMW2 showed aluminum concentrations above detection limits that range between 0.1 and 0.8 mg/L. There is no time trend indicated for aluminum. These aluminum data indicate that the upgradient well is the only well that contains a measurable amount of dissolved aluminum. All other monitoring wells are unaffected for aluminum, demonstrating no effects from the placement of coal ash.

Iron shows considerable variability and is measured above detection limit in upgradient Wells GMW2 and MW1. Well GMW2 appears to show an increasing trend. Both downgradient Wells MW6 and MW7 show low iron concentrations, averaging about 0.35 mg/L compared to an average concentration of about 7 mg/L for Well MW1 and an average concentration of 5.8 mg/L for Well GMW2. The time-series graph for iron in Well GMW2 indicates an increasing trend over the monitoring period of more than 10 years. The pH in this well also appeared to be decreasing, which may be related to the increasing iron concentrations. The pH at Well MW1 is nearly constant at around 7 for the entire monitoring period. There is relatively more variability in pH measured at MW6 and MW7, with averages of 6.25 and 7.70, respectively. All these monitoring data show that placement of ash in the strip mine has not affected pH in the downgradient groundwater.

Sulfate concentrations in the four monitoring wells show four different behaviors. Background Well GMW2 has an average sulfate concentration of about 12 mg/L, with a range from about 4.5 to 18.5 mg/L. There is no time trend indicated by the monitoring data. Sulfate in MW1, an upgradient well, shows an average concentration of about 225 mg/L, with a range from 7.5 to 483 mg/L. It appears that there was an increase in sulfate concentration for a period of time, followed by a decrease. Sulfate at MW6 is also low in the range of 3 to 18 mg/L. However, the sulfate concentrations show a decreasing trend over the more than 10 years of monitoring. Monitoring Well MW7 has consistently shown measured sulfate concentrations to be near 5 mg/L. However, there are four values out of 47 observations that are in the range of 55 to 200 mg/L.

Arsenic data show that it is not above detection limits in all four wells. Arsenic concentrations above detection limits were observed only twice in over 160 measurements.

4.6 Universal Mine (IN)

The Universal coal Mine site is located about 5 miles north-northwest of Terre Haute, Indiana in Vigo County. The site is adjacent to the eastern boundary of the eastern region, Interior Province coalfields of Illinois and southwestern Indiana. These mine fields have typically produced medium-to-high-volatility bituminous coals of Pennsylvanian and Permian geologic age. The south embankment of the Universal site ridge was augur-mined during the 1940s and 1950s. Peabody Mining Company began and completed highwall mining operations at the Universal site along a north-south line in the mid-1980s, creating a final cut pit which was approximately 1920 feet long, 325 feet wide, and about 90 feet deep.

In 1988, PSI Energy, Inc. (now Cinergy Corporation), acquired a portion of the Universal Mine site containing the final cut pit for the express purpose of coal ash deposition and surface

mine reclamation. Indiana DNR issued a permit to PSI Energy to dispose of fly ash/bottom ash from its nearby Wabash River Station to fill and reclaim the mine pit. The permit allowed for approximately 1.448 million cubic yards of CCPs to be disposed of with ash deposit thickness ranging from 30 to 75 feet. Between April 1989 and the end of October 2001, Cinergy placed approximately 1.6 million tons of coal ash from a nearby power plant to completely fill the open pit. Table 4.15 shows the annual tonnage of coal ash and coal gasification slag placed in the mine pit at the Universal site during the 13.5 years of mine-filling operation. Cinergy has been conducting quarterly monitoring of groundwater and surface water at the Universal ash fill site since early 1988.

The coal ash was hauled by triaxle road-going trucks and placed dry in the mine pit. The coal ash was deposited in the mine pit until the height of the filled area corresponded to the approximate original topographic contour. A 5-foot noncompacted soil cap and vegetation are being established as a final cover. The reclaimed mine pit land is to be maintained as a wildlife refuge.

4.6.1 Hydrogeology

The geology of the premined consolidated strata consisted of a thin layer of fine clay overlain by an 18-inch-thick coal seam that was overlain by shaley limestone, shale, and a thick deposit of loess. The exposed high wall at the Universal site showed a mantle of about 12 to 16 feet of pre-Wisconsin till and Wisconsin loess over the bedrock. The Universal site area is part of an upland landscape north of Coal Creek and west of the Wabash River. The upland rises about 100 feet above the Coal Creek flood plain and was well dissected before surface mining altered the topography.

The water level elevation data from the groundwater-monitoring wells at the Universal site indicate that the groundwater from the unmined east side of the ash fill area is flowing to the west into the ash that has resulted in approximately 30 ft of saturated zone in the reclaimed mine pit. The groundwater then flows in a radial manner to the west, north, and south. This groundwater flow field implies that monitoring Well MW-4 is the upgradient well and that all other wells are hydraulically downgradient of the placed coal ash.

4.6.2 Chemical Composition of Coal Ash and Leachates

Between May 1988 and December 2001, 25 samples of coal ash were collected and analyzed for total chemical concentrations and for 18-hour and 30-day leachate concentrations. Table 4.16 provides a statistical summary given by mean, median, maximum, and minimum values based on the measured concentrations for the bulk ash samples. Table 4.19 provides a statistical summary for the 18-hour and 30-day leaching tests completed on the same ash samples. Table 4.18 contains a statistical summary of field leachate composition data based on 17 samples collected and analyzed from leachate monitoring Well MW-8 installed in 1997.

The coal ash deposited at the Universal site contains aluminum, iron, potassium, calcium, magnesium, sulfate, sodium, and boron at average concentrations of greater than 240 mg/kg. Manganese, zinc, barium, arsenic, vanadium, nickel, chloride, lead, chromium, copper, and

fluoride are present in the average concentration range of 18 to 128 mg/kg. Molybdenum, selenium, cadmium, silver, and mercury are found to range from nondetect to an average of about 5 mg/kg. The coal ash is alkaline, with an average pH above 9.0 has a net neutralization capacity of about 17.6T/1000T, and contains about 5.8% total organic carbon.

The laboratory-generated leachates as well as the field leachate data show that all leachates contain sulfate as the most abundant constituent released from the coal ash. Aluminum, boron, sodium, and calcium were the next most abundant constituents in the laboratory-generated leachates. The field leachates, however, showed that calcium, sodium, chloride, boron, and potassium were the next most leached constituents (Table 4.18). The laboratory-generated leachates contained no silver, mercury, nickel, or lead. The field leachates contained no dissolved cadmium, chromium, copper, iron, lead, mercury, nickel, selenium, silver, or zinc. Field and laboratory leachates also showed arsenic, with some variability in concentration.

Time-series plots for pH, sulfate, boron, and arsenic in leachate monitoring Well MW-8 are given in Figures 4.14 through 4.17. These plots show that ash leachate has a pH of over 9 throughout the monitoring period of 1997–2001, with no measurable acidity. The sulfate concentration in the ash leachate appears to be holding at about 1700 mg/L, and boron is present at a concentration of about 45 mg/L. The field leachate data indicate that ash leachate at the Universal site is highly alkaline and contains about 200 to 300 mg/L each of chloride and sodium. However, both sodium and chloride concentrations in the leachate show a rapid decrease over the monitoring period, indicating a near depletion of these constituents in the fly ash. MW-8 monitoring data show an initial increase in arsenic concentration in the ash leachate followed by a steady concentration ranging between 180 and 250 µg/L.

4.6.3 Water Quality-Monitoring Program

Both groundwater and surface water are monitored on a quarterly basis for the compliance-monitoring program by Cinergy. The Universal site originally installed four groundwater-monitoring wells (i.e., MW-1, MW-2, MW-3, and MW-4) in May 1988, approximately 1 year before the placement of coal ash began in April 1989. MW-4 is an upgradient well installed in the undisturbed bedrock formation and has been monitored for water level and water quality since May 1988 on a quarterly basis.

Monitoring Well MW-1 was installed to the northeast side of the ash fill and had a long screen covering both the bedrock and the overlying spoil material. The MW-1 well was replaced in December 2000 by two new wells designated as MW-1BR and MW-1UR. Monitoring Well MW-2 was installed at the edge of the mine fill to the west presumably in the downgradient direction of the groundwater flow. This monitoring well was replaced by a new Well MW-2A in 1997 which was properly screened in the mine spoil material to the west and downgradient of the ash fill. Original Well MW-3 was installed to the south of the ash fill area, approximately 30 ft downgradient in the mine spoil material. This well was replaced by MW-3R in December 2000. In 1997, additional compliance Wells MW-5, MW-6, and MW-7 were installed in the mine spoil material downgradient of the ash fill. In 1997, MW-8 was installed in the ash fill to monitor chemical composition of the ash leachate. Therefore, the longest series of groundwater-monitoring data is available from monitoring Wells MW-1, MW-3, and MW-4, with a somewhat

shorter time-series of monitoring data from Wells MW-2, MW-2A, MW-5, MW-6, MW-7, and MW-8. Replacement Wells MW-1BR, MW-1UR, and MW-3R are relatively new and have the shortest set of time-series data.

As part of the compliance-monitoring effort, Cinergy has also been sampling and analyzing surface water samples from an old mine seep location since 1988. Cinergy has since then added five more surface water quality-monitoring locations to its compliance-monitoring program. These sampling locations are designated as North Pond, Ash Pit-Water Pond, Plug Seep, and two locations in the Coal Creek. In this summary paper, we will only present and discuss the mine-seep water quality data to depict the benefits and impacts of coal ash placement on surface water at the Universal site.

Table 4.19 provides a list of parameters that have been measured in the water quality samples collected in the groundwater- and surface water-monitoring program at the Universal site.

4.6.4 Groundwater Quality Results

Groundwater quality-monitoring data from Wells MW-1, MW-2, MW-2A, MW-3, MW-4, MW-5, MW-6, and MW-7 were evaluated for time trends and changes in concentrations that may be related to the placement of ash. It was determined that aluminum, cadmium, chromium, copper, mercury, nickel, and silver concentrations in groundwater in these wells during the period 1988 through 2001 were at or below analytical detection limits. Therefore, no further discussion of these constituents is included in this report. The data analysis also indicated that sulfate concentrations in groundwater at Wells MW-2, MW-2A, MW-3, MW-5, MW-6, and MW-7 are significantly higher than those found at MW-1 and MW-4. Boron concentrations in monitoring Wells MW-1, MW-2, MW-2A, MW-3, MW-5, and MW-6 are significantly higher than those found in groundwater at Wells MW-4 and MW-7. The groundwater pH in most wells is near neutral, with a narrow range of 6.8 to 7.2. However, MW-3 groundwater has the lowest average pH, at about 6.4. The nearly 13 years of quarterly groundwater-monitoring data also show that the waters are alkaline, with notable concentrations of chloride, magnesium, and sodium. The groundwater at the site also contains several mg/L of total organic carbon.

The groundwater quality-monitoring data also show no significant differences in concentrations of barium, cadmium, chromium, copper, fluoride, lead, mercury, nickel, selenium, silver, and zinc. Sulfate is present at low levels in monitoring Wells MW-4 and MW-1, but is found at concentrations between 500 to 2,000 mg/L in Wells MW-2, MW-2A, MW-3, MW-5, MW-6, and MW-7. MW-3 contains higher levels of iron, manganese, and boron. Boron concentrations are also higher in groundwater at MW-1, MW-2, MW-2A, MW-5, and MW-6 compared to those found in Wells MW-4 and MW-7. The pH in groundwater wells at the site generally falls in the range of 6.8 to 7.2 except for MW-3, which showed a slightly more acidic pH of about 6.4.

Since MW-3 and MW-4 wells data contain water quality measurements both before and during ash placement from 1988 through the end of 2001, we focused on presenting and discussing those data in this report. The quarterly monitoring data for pH, acidity, alkalinity,

manganese, iron, sulfate, chloride, boron, and arsenic are presented in Figures 4.18 through Figures 4.26 as time-series plots. There are several observations that can be extracted from these time-series plots. The pH time-series plot (Figure 4.17) indicates that throughout the 13.5 years of monitoring, pH in groundwater at downgradient Well MW-3 is somewhat acidic compared to the near-neutral pH in upgradient Well MW-4. Groundwater in Well MW-3 contains about 200 mg/L of acidity, whereas MW-4 contains essentially no acidity (Figure 4.18). Both MW-3 and MW-4 groundwater contains over 350 mg/L alkalinity with MW-3 groundwater being more alkaline than MW-4 from the 1988 through 1998 time period. Both MW-3 and MW-4 have alkalinity of about 450 mg/L between 1998 through 2001 (Figure 4.20).

Figure 4.21 shows the time-series plot of manganese concentrations. The upgradient well shows no measurable amounts of manganese for the entire monitoring period, whereas MW-3 has shown manganese being present in several mg/L concentrations. There appears to be a decrease in manganese concentrations in MW-3 groundwater over the 13-year monitoring period. These monitoring results indicate that dissolved manganese was present at about 5 mg/L level in groundwater downgradient of the ash fill area before the ash was placed and, therefore, the observed manganese concentrations in MW-3 are not associated with the placement of ash in the mine pit.

Figure 4.22 shows the time-series plot of dissolved iron in groundwater at monitoring Wells MW-3 and MW-4. Upgradient Well MW-4 shows an absence of dissolved iron in the groundwater, whereas downgradient Well MW-3 shows an average of about 20 mg/L iron. Dissolved iron at this elevated concentration was present in Well MW-3 before ash placement. The monitoring data lead us to recognize that groundwater in downgradient Well MW-3 contains elevated dissolved iron concentrations that are not associated with the placement of coal ash in the Universal mine pit.

Figure 4.23 shows the time-series plot of sulfate in groundwater at Wells MW-3 and MW-4. The upgradient groundwater measured in MW-4 in an undisturbed geological setting has an average concentration of about 53 mg/L of sulfate, whereas the groundwater in downgradient Well MW-3 screened in the mine spoil material has sulfate concentrations of around 1500 mg/L. It is noted that sulfate was present at these elevated concentrations at MW-3 in water samples collected before ash placement and that subsequent to ash placement the sulfate levels has not increased in the period from 1989 through 2001. This time-series behavior leads us to conclude that even though sulfate is an ash leachate constituent, it has not impacted the downgradient groundwater at MW-3 at the Universal site.

Figure 4.24 shows the time-series plot of measured chloride concentrations in groundwater at MW-3 and MW-4. The average chloride concentration in upgradient Well MW-4 is about 11 mg/L, and downgradient Well MW-3 contains about 18 mg/L. Chloride in groundwater at these two wells is pretty low and requires no further discussion.

Figure 4.25 shows the time-series plot for boron concentrations in groundwater at Wells MW-3 and MW-4. This plot indicates that background Well MW-4 contains an average concentration of about 0.4 mg/L of boron. The boron concentrations in groundwater at MW-3 do show an increase after early 1990 lasting through 1997 before showing a decrease during 1998

through 2001. The average boron concentration in groundwater at MW-3 is about 2.1 mg/L. Boron is a known leachate constituent associated with coal ash. The field-scale leachate-monitoring data show that boron is present in the ash leachate at the site in over 40-mg/L concentrations. Therefore, it is inferred that the elevated boron concentrations in groundwater at MW-3 may be a result of ash leachate migration even though there is an approximate 20-fold decrease in concentration between the source and downgradient monitoring Well MW-3.

Figure 4.26 shows the time-series plot for dissolved arsenic in monitoring Wells MW-3 and MW-4. The average dissolved arsenic concentration in upgradient Well MW-4 is 5 $\mu\text{g/L}$, with over 95% of the data showing below-detection-limit values. The measured arsenic in MW-3 shows an increase during the monitoring period 1989 through 1996 and then a large variability in concentrations measured during 1997 through 2001. The maximum measured concentration of arsenic in MW-3 was 35 $\mu\text{g/L}$, with about 43% of the data showing below-detection-limit values. The ash leachate data from MW-8 and data from the laboratory leaching tests showed arsenic concentrations to range from about 200 to 300 $\mu\text{g/L}$. Further research at the Universal site is now evaluating the release and migration of arsenic in groundwater to accurately describe the fate of leached arsenic from coal ash.

4.6.5 Surface Water Quality

AMD was present at the Universal site. AMD was caused by the oxidation of pyrites because of coal-mining operations. An old mine seep is present to the southeast of the ash-filled mine pit. The mine seep water flows through a surface channel into Coal Creek, located at least 1500 feet further south of the mine seep area. This old mine seep water has been sampled quarterly and analyzed for chemical parameters listed in Table 4.18. We have prepared time-series plots for pH, acidity, alkalinity, manganese, iron, sulfate, chloride, and boron measured in the old mine seep water from May 1988 through the end of 2001.

Figure 4.27 is the plot of pH which shows that prior to coal ash placement, the mine seep water was highly acidic (i.e., $\text{pH} < 3.5$) which has been completely neutralized to pH 7 by the coal ash deposit in the mine pit. The neutralization of the acidic pH is consistent with the alkaline chemical characteristic of the coal ash.

Figure 4.28 shows the time-series plot for measured acidity in the mine seep water. The AMD water contained a large amount of acidity before coal ash placement in the mine pit. Within 1 year of ash deposition, the acidity in the mine seep water was nearly eliminated, again because of the acid neutralization capacity of the coal ash placed in the Universal Mine pit.

Figure 4.29 is a time-series plot of measured alkalinity in the mine seep water. This plot shows the absence of alkalinity during 1988 to 1990, followed by a gradual increase in alkalinity through the end of 2001. Therefore, the coal ash not only neutralized acidity and increased the pH of the groundwater feeding the mine seep, it has also generated additional alkalinity to provide further improvements to water quality.

Figures 4.30 and 4.31 are the time-series plots of measured concentrations of manganese and iron, respectively, in the mine seep water. Before the coal was placed in the mine pit, the

mine seep, an AMD water, was quite high in dissolved iron (over 100 mg/L) and manganese (over 7 mg/L). But within a year of ash placement, both iron and manganese concentrations decreased significantly. The long-term monitoring further indicates that dissolved iron has continued to decrease and has essentially reached to a nondetect level, with manganese also showing a gradual decrease to a concentration of below 2 mg/L in 2001. These decreases are due to the neutralization of acidity and increase in pH achieved by the alkaline ash. Increases in pH and alkalinity have precipitated the dissolved iron and manganese. It is also possible that the lack of oxygen may have stopped the generation of AMD through oxidation of pyrites.

Figure 4.32 is a time-series plot of sulfate concentrations measured in the mine seep water. This plot shows a notable decrease in sulfate concentrations as a function of time. The sulfate levels of over 2000 mg/L in 1988 are now reduced to about 1500 mg/L in 2001. The initially high sulfate concentrations most likely were a result of pyrite oxidation which produced sulfate, iron, and acidic pH waters with low levels of calcium and other base cations in the water. The alkaline coal ash provided large amounts of calcium as well as alkalinity through the leachate. This ash leachate mixed with AMD waters and resulted in the precipitation of iron hydroxide(s) as well as gypsum, thereby creating the modified water quality of the mine seep that is now low in iron and sulfate compared to the original AMD waters. Therefore, even though sulfate is a coal ash as well as mine spoil-derived constituent, in this case there is no distinguishable impact of sulfate on the water quality in the mine seep.

Figure 4.33 shows a time-series plot of chloride measured in the mine seep water. This plot shows a slowly increasing trend in chloride concentrations, but the chloride levels were still less than about 100 mg/L in 2001.

Figure 4.34 shows a time-series plot of boron concentrations in the mine seep water samples monitored since May 1989. For the first 2 years of monitoring, boron concentrations in the mine seep water were less than 1 mg/L. Beginning in May 1991, the boron concentrations increased steadily, with the highest concentration of about 5 mg/L recorded in 2001. This time-series behavior and the documentation that coal ash leachate at this site contains over 45 mg/L of boron led to the conclusion that boron contained in the ash leachate has resulted in the elevated boron concentration in the mine seep water samples.

4.6.6 Summary and Conclusions

This paper provides a limited summary of monitoring results on water quality at a coal ash-filled surface coal mine pit in Indiana. The over 13.5 years of quarterly water quality-monitoring data and the more recent extended spatial monitoring data provide insights on coal ash leachates and water quality changes when coal ash has filled and reclaimed a final cut mine pit at the Universal site. The monitoring data to date indicate that the alkaline coal ash leachate has been effective in improving AMD water quality present at the site. The coal ash leachate neutralized the acidic pH, increased alkalinity, essentially eliminated acidity, and significantly decreased manganese, iron, and sulfate concentrations. There were no indications of any other trace metal migration via the mine seep. However, the coal ash leachate did significantly increase boron concentrations in the mine seep water.

The groundwater quality data similarly show that coal ash leachate has not resulted in the leaching and migration of most of the trace metals contained in ash. Boron, arsenic, and sulfate are leached from the coal ash in different amounts. However, because of the presence of sulfate in groundwater in the spoil material, it is not feasible to discern the migration of sulfate contained in the coal ash leachate from the sulfate contributed by the mine spoil material.

Trace metals such as barium, chromium, cadmium, copper, lead, mercury, nickel, selenium, and silver all show no signs of leaching or impacting the groundwater and surface water quality around the site. The laboratory and field leachates do show leaching of arsenic from the coal ash, but over the nearly 13-year period of monitoring it appears that arsenic has not migrated and impacted downgradient groundwater or surface water quality. The continuing monitoring and the supplemental research at the Universal site will provide a better understanding of the benefits and impacts from the use of coal ash for mine filling in the next few years.

4.7 Midwestern Abandoned Mine (IN)

This case study is an example where a state agency elected to place CCPs with a Poz-O-Tec (a mixture of FGD sludge, fly ash, and quicklime) cap, which resulted in reduced infiltration and improvement in water quality by neutralization. The Midwestern Abandoned Mine consists of approximately 550 acres of previously mined land, which in some instances intersects with abandoned deep mining of the same coal seam. The Midwestern site is located within a 6000-acre Mill Creek Abandoned Mine Land Area which was partially reclaimed in the early 1980s as part of Indiana's Abandoned Mine Land Program. Deposits of pyritic coarse- and fine-grained refuse were surrounded by high walls associated with final cut pits; ponded water and seepage from an abandoned underground mine further characterized the site.

The Indiana Department of Natural Resources Division of Reclamation (IDNR DOR) conducted extensive evaluations including review of laboratory test results and a review of state and federal regulations to decide that CCPs could be utilized for the reclamation of abandoned mined land (AML). The reclamation project consisted of placing and compacting of CCPs and then covering with Poz-O-Tec. The technical specifications required the water to be removed from the highwall pits to facilitate proper placement of the ash into these areas. In actuality, the pits would not drain because of an interconnection to flooded underground mines and the existing groundwater table. To solve this problem, ponded ash was placed along the perimeters of the pits near the areas where the opening to the underground mines were located. This placement of the coal ash through the pit water was compacted enough to seal off the underground connection to prevent further migration of groundwater into the pits. The placement of ponded ash into these pits raised the pH of the pit water and the metals precipitated out of solution. This allowed the IDNR to pump out the pits without the use of a water treatment system before discharge, and the alkalinity in the ash replaced the need for calcium hydroxide addition for pH adjustment. Since completion of the reclamation, the groundwater has returned to the normal water level and the ash fill areas are below the water table.

The Indiana Geological Survey has conducted a monitoring program covering 1 year prereclamation baseline water quality monitoring and more than 1 year of postreclamation

monitoring. The objectives of the monitoring are to 1) determine the effectiveness of acid mine drainage abatement and 2) monitor the leaching of solutes from CCPs. Water samples from wells screened at different depths and several surface water discharge points have been analyzed for a number of chemical parameters. Figure 4.8.6 shows the location of these monitoring points for both the prereclamation and the postreclamation conditions.

Olyphant et al. (1997) provided a summary of prereclamation water quality data for a selected set of monitoring locations. Branam et al. (1999) provided a summary of postreclamation water quality data for a selected set of monitoring wells and chemical parameters. Based on the monitoring results, Branam et al. (1999) concluded that using CCPs to reclaim the Midwestern Abandoned Mine has resulted in the reduction of AMD leaving the site. The authors ascribe this response to the reduction in vertical recharge of oxygenated water by the fixated scrubber sludge cap and the neutralization provided by the alkaline CCPs.

4.8 The Arnold Willis “City” Underground Coal Mine (IN)

This case study is an example in which fixated scrubber sludge (FSS) has been successfully injected into an abandoned underground coal mine for stabilization by filling voids. Groundwater-monitoring data indicated that trace metals and sulfide remained unaffected by the placement of FSS.

Coal was mined from the Arnold Willis “City” mine from the late 1940s to early 1950s. The depth to the voids in this underground mine varied from 32 to 64 ft, and the void thickness varied from 2 to 7 ft. Borings into the mine encountered both flooded and dry mine conditions. An FSS grout consisting of a mixture of FGD scrubber sludge, fly ash, lime, and water was developed for injection in the abandoned deep mine to abate surface subsidence and reduce acid mine drainage. A total of 12,502 m³ of FSS was injected over an 8-week period, resulting in filling of about 5 acres of the mine. Monitoring data show that the FSS material generally achieved unconfined compressive strengths of 100 psi or higher.

Water quality monitoring was conducted over a period of 18 months. The first 6 months of monitoring was completed prior to FSS injection, and the following 12 months of monitoring occurred after the grout was in place. Figure 32 shows the locations of the monitoring points and Tables 4-8-1 through 4-8-16 provide the groundwater quality analytical data for the various locations and chemical parameters measured.

The investigators reported that heavy metals in the water samples were relatively unaffected by the FSS injection. These metals include arsenic, cadmium, chromium, lead, selenium, aluminum, barium, and sulfide. The investigators also reported that acidity and pH did not appear to have changed as a result of the placement of FSS. However, sodium, potassium, calcium, boron, chloride, and magnesium showed some increases attributable to the FSS. Changes in sulfate, total dissolved solids, and specific conductance were also observed.

4.9 Harwick Mine Complex (PA)

4.9.1 Introduction

The Harwick Mine Complex includes the Monarch, Old Harwick, and Cornell Mines, covering approximately 7000 acres. This coal mine is a deep mine and was operated from about 1932 through June 1970. The elevation of the mine varies from about 740 to 565 ft msl, with depth to coal ranging from 20 ft and 700 ft below ground surface. After coal-mining activities ceased, the mine workings were allowed to fill with water and are currently flooded. Water level in the mine is maintained by dewatering at the Monarch Mine Dewatering Station located at the Monarch Shaft. The entire mining perimeter is surrounded by a barrier of unmined coal isolating the Harwick Mine from the neighboring mines. Approximately 8 to 12 million gallons per day of mine water is pumped out, treated, and discharged to the nearby Little Deer Creek under an National Pollutant Discharge Elimination System (NPDES) permit.

The Monarch shaft is located under the lowest portion of the mine; therefore, all Harwick Mine water flows to this shaft. Coal mining was carried out by room-and-pillar or other conventional methods with varying rates of coal extraction. In some areas, caving occurred soon after mining. In other areas, substantial voids remain within the mine.

4.9.2 History of Ash Injection in Mine

Since 1970, Duquesne Light Company sluiced fly ash from its Cheswick Power Station into the abandoned Harwick Mine, which is located adjacent to the power station. Prior to the Cheswick Power Station, in July 1967, fly ash from the Colfax Station was injected as a slurry with river water to a worked out area of the Harwick Mine as an experiment, and the fly ash particles were allowed to settle. Then the water was pumped from the mine, treated, and discharged. The water treatment method employed neutralization, aeration, and clarification. The clear water was discharged to a nearby river under a mine drainage permit issued by the Pennsylvania State Sanitary Water Board. This pilot deep mine disposal operation lasted for almost 3 years and provided the basis for the sluicing of the fly ash from the Cheswick Power Station, which is the successor plant to the Colfax Station. It was noted that the use of fly ash in preventing mine subsidence was a benefit that could be derived from such an operation.

4.9.3 Ash Sluicing and Water Pumping Operations

The mine disposal operation consists of a wet ash-handling system at the Cheswick Power Station which includes two 5000+-gpm pumps connected in series and approximately 8000 ft of slurry pipeline to two operating injection boreholes at the mine. The fly ash generated at the power station is mixed with water from ash handling operations, other intermittent low-volume wastewaters generated in the vicinity of the ash-handling operations, and additional sluicing water.

The combined slurry (approximately 10% solids) is conveyed by the pipeline to an injection borehole into the Harwick Mine Complex. Approximately 3 to 4 million gallons per day of the slurry are conveyed.

Since 1970, a number of boreholes have been used as injection points until refusal. Additional locations for future boreholes have been identified, with a program for development and testing prior to use. Approximately 150,000 tons of coal ash is injected annually into the mine along with millions of gallons of water.

4.9.4 Coal Ash Characteristics

Bulk and leachate chemical analyses of the coal ash from the Cheswick Power Station were conducted on samples collected in 1990. These results are shown in Table 4.20. These results indicate that the ash is neutral in pH and contains very low levels of leachable constituents. Even sulfate in the leachate is very low.

4.9.5 Mine Effluent Characteristics

The utility company pumps out about 8 to 10 million gallons of water daily and treats it before discharge to the nearby river. Samples of the mine water have been collected and analyzed both before and after water treatment. Table 4.21 provides the chemical composition data from the Monarch Mine for a sample collected and analyzed in 1973.

These results indicate that the mine water before treatment is slightly acidic, high in total dissolved solids, and contains over 100 mg/L of iron, over 2000 mg/L of sulfate, about 100 mg/L of chloride, and nearly 1000 mg/L of sodium. The water treatment results in an increase of pH to about 7.5 and a reduction of iron to below 2 mg/L, with no other changes in the concentrations of other dissolved constituents.

4.10 Clinton County (PA)

This case study provides an example of the placement of fluidized-bed combustor (FBC) ash in a closed-surface coal mine resulting in beneficial effects on water quality because of the favorable geochemistry that occurs. The alkaline FBC ash neutralizes the acidic AMD waters, resulting in precipitous decreases in arsenic, cadmium, and aluminum concentrations due to lower solubility and precipitation of solids.

Between 1974 and 1977, a 37-acre surface coal mine was mined and reclaimed. The upper split of coal was mined, leaving thick underclay as a pavement. Soon after the closure of the mine, acidic discharges were noted, averaging about 35 gpm and affecting water quality in the nearby streams. Schueck et al., (1996) conducted extensive studies of the site, locating piles of pyritic materials, buried refuse, and pit cleanings in the closed and reclaimed mine. A decision was made to use FBC ash, chosen for its pozzolanic properties, as a low-viscosity grout for pressure injection to encapsulate the pyritic material with a cementitious coating.

Forty-two monitoring wells were installed on and adjacent to the site. In addition, a surface discharge point was monitored for water quality. Water quality monitoring was conducted from 1990 to 1995. The grouting with the FBC ash was carried out in 1992 and 1993. The water quality data from the pregrouting period show poor water quality, characterized by pH of less than 2.5, high total dissolved solids, high sulfate, high iron, high aluminum, high manganese, and

low calcium. Cadmium, copper, and chromium were also present in the water. Table 4.10.1 contains a summary of the monitoring results as presented by Schueck et al. (1996) to show that generally the water quality at the monitoring wells improved following the grouting by the FBC ash. These results indicated limited neutralization of the AMD, but more importantly, there are no indications that the groundwater quality is adversely affected by leaching of the FBC ash under highly acidic conditions.

A monitoring well (L25) was monitored to evaluate the chemical changes that occurred as a result of the FBC ash injection into the acidic environment created by the oxidation of pyrites in the mine. The results indicated that the injection of grout caused a temporary increase in pH from about 2.3 to about 9 as the alkaline FBC ash neutralized the acidic AMD waters. The pH increase resulted in precipitous decreases in arsenic, cadmium, and aluminum concentrations, reflecting lower solubility and precipitation to solid-phase compounds. However, within a short time, the pH again became acidic, with sulfate and aluminum returning to the pregrouting concentrations, although arsenic and cadmium remained at much lower levels.

4.11 Big Gorilla Pit (PA)

4.11.1 Introduction

In eastern Pennsylvania, there are several preact strip mine pits in the middle of the anthracite coal basin where active strip and deep mining for coal was practiced since the 1800s. The Silverbrook Basin contained two abandoned strip-mined pits. They are locally known as the “Big Gorilla” and the “Little Gorilla.” The Big Gorilla had an estimated volume of approximately 120 million gallons when the water level was at 1570 feet msl but can fluctuate by about +20 feet. It was approximately 1400 feet long, 400 feet wide, and about 90 feet deep before ash placement began. Big Gorilla pit was significantly affected by AMD.

The Big Gorilla demonstration project involved dry-to-wet ash placement into standing mine water. The objectives of this demonstration project were 1) to show that large-scale placement of ash into mine water is possible with minimal turbidity generation, 2) to develop the methodology for large-scale ash placement, and 3) to demonstrate that ash in contact with mine water will not result in adverse environmental releases. Cogeneration ash, from the Northeastern Power Company (NEPCO) that produces electricity by burning culm as fuel, was first placed in contact with the surface mine pool in August 1997, which was then entirely filled by 2004. Over 3 million tons of ash was placed from two platforms to fill the Big Gorilla mine pool.

Samples were collected and analyzed for water quality parameters from the standing water in Big Gorilla pit in 1993. Table 4.23 shows the water quality-monitoring data for before and after the placement of coal ash in Big Gorilla (Loop et al., 1999).

4.11.2 Mining and Reclamation

The major coal beds in this region include Buck Mountain vein and Mammoth vein within the Llewellyn Formation. The Llewellyn also contains an abundance of pyrite and siderite minerals. The Silverbrook syncline is divided by the Centralia thrust fault, which surfaces at the

southern bank of the Big Gorilla mine pool. At this location, the Buck Mountain vein was deep mined at the turn of the 19th century.

In the Silverbrook Basin, surface mining of the Mammoth vein ceased before World War II. Currently, NEPCO burns culm from this location and other nearby sites. In late 1940s, the basins created by mining the Mammoth vein were filled with coal silt. This silt was later reclaimed, and the present-day Big Gorilla occupies the No. 1 Basin.

In 1994, clean fill was put into the western end of the Big Gorilla pit in order to build a terrace above the maximum high water level to prepare for dry land disposal of fly ash in accordance with existing permits and Department of Environmental Protection (DEP) regulations at that time. Filling the entire pit with clean fill was cost-prohibitive, whereas using fly and bottom ash from on-site avoided purchase and transportation cost in addition to having beneficial alkaline properties.

4.11.3 Regional Acid Mine Drainage Chemistry

Mine discharge quality data have been collected since the 1940s in all four fields of the anthracite basin (Felegy et al., 1948; Ash et al., 1951). The Silverbrook Basin contributes to the Little Schuylkill River, and together with the other streams in the Little Schuylkill River Basin, is one of the most significant sources of sulfate in the southern field. When the Little Schuylkill River was sampled below Tamaqua, the pH was 5.4 and the sulfate concentration was 240 mg/L (Growitz et al., 1985). The water discharged from the 13 outflow sites contributing to the river was 18 cfs, approximately 23% of the flow at the sampled site below Tamaqua. Acidity values in 1975 (Growitz et al., 1985) and 1991 (Wood, 1996) were determined in the field on an ambient-temperature sample to the end points of both pH 7.0 and 8.3, and the values for the Silverbrook mine outflow increased from 80 to 234 mg/L CaCO₃ and from 90 to 252 mg/L CaCO₃, respectively (Wood, 1996).

Thirty-eight samples of iron and sulfate concentrations between 1970 and 1993 were reported by Wood (1996). While the average concentration of iron in the Silverbrook discharge shows no apparent trend, the sulfate values have been decreasing, most notably in the 1970–1972 period and 1975, where the mean concentration decreased by over 100 mg/L.

4.11.4 Ash Placement and Characterization

Ash was first deposited into the Big Gorilla mine pool in August of 1997. Fly ash from one silo and bottom ash from another were loaded into trucks in 45-ton loads and sprayed with water to prevent dust. They were then taken to the Big Gorilla mine pool ash platform and piled on the bank. Bulldozers were used to push the ash piles into the water from the lower platform, with the exception of the winter months, when ash was only placed from the edge of the upper platform. However, toward the end of the project, no distinction was made, as both platforms were found to be safe during the winter. When ash was pushed into the water, turbidity boils could be seen approximately 300 feet from the ash face. This indicated that at least the fine portion of the ash was flowing outward from the face, in a manner similar to that of a turbidity current. When

postplacement ash samples were taken from the bottom of the Big Gorilla, only fine material was recovered.

In NEPCO's case, ash is the by-product of burning culm and limestone to produce electricity. Approximately 1700 tons of culm and 60 tons of limestone are burned daily. Ash from the NEPCO cogeneration facility is reddish brown when wet and pinkish grey when dry. A greater variety of colors are observable in the bottom ash when compared to the fly ash. Fly ash particle sizes can be described as very fine, with a diameter in the 62–88- μm range. Bottom ash particles range from approximately 100 μm to 1 cm in diameter and are angular.

Both the bottom and fly ash were subjected to testing by total digestion and analysis, in addition to leaching tests by the U.S. Environmental Protection Agency (EPA) Method 1311, TCLP, and later the synthetic precipitation leaching procedure (SPLP), EPA Method 1312. Table 4.24 provides an average of the results from 10 years of testing the fly and bottom ash from NEPCO.

4.11.5 Chemical Evolution of the Big Gorilla Mine Lake

On July 9, 1998, the entire mine pool was both horizontally and vertically homogeneous, with a pH of approximately 11.1. Four samples were collected for aqueous analysis from the Big Gorilla mine pool prior to ash placement. A surface sample was collected in June 1993, and three variable-depth samples were collected in July 1993. While the surface samples from June and July have very similar chemistry, there is very distinct stratification with depth. Conductivity, total dissolved solids, turbidity, and acidity all increased with depth. Aluminum, sulfate, iron, manganese, calcium, and magnesium concentrations all increased with depth as well. The pH values from the samples taken at depth were measured in the laboratory and showed no consistent variation. When quarterly sampling results of the Silverbrook outflow from 1993 and 1994 are compared to the preash-placement chemistry of the Big Gorilla mine pool, the Silverbrook outflow had a higher pH, in addition to higher values for acidity, iron, manganese, sodium, chloride, calcium, magnesium, and total dissolved solids.

Since October 1997, over 25,000 tons of fly and bottom ash had been placed in the Big Gorilla mine pool, and the mine pool water became alkaline. The water increased in alkalinity with increased ash input. Before the first winter, there was not a significant amount of alkalinity present, and values remained below 50 mg/L CaCO_3 until June 1998. After June, there was a consistent increase in alkalinity, averaging 80 mg/L CaCO_3 per month for the next 5 months. This trend ended when ash placement ceased for the winter. At that point, there was a decrease in alkalinity that was not reversed until ash placement resumed in July 1999. In early 1999, 2000, 2001 and 2002, alkalinity dropped dramatically, following significant suspensions in ash placement, indicating that the buffering capacity of the water was strongly tied to ash placement. Between March 2000 and May 2002 the Big Gorilla mine pool water did not fall below a pH value of 11.0.

Based on the June through November 1998 data, which were taken during ash placement, an average of 1.24% of the calcium in the ash is available. A large reservoir of excess CaO is present in the ash.

High sulfate concentrations are one of the key features of AMD impacted streams. Total sulfate concentrations in the Big Gorilla mine pool in November 1998 were approximately 147 mg/L. Initially, the Big Gorilla mine pool water had sulfate concentrations ranging from 55 to 79 mg/L. Since July 1998, sulfate concentrations have risen in response to ash input. The sulfate concentration in the Big Gorilla has been as high as 1021 mg/L, measured in August 2001. In four months, however, it had decreased to 800 mg/L, and within the year, it was less than 600 mg/L.

The pre-ash-placement Big Gorilla mine pool had an average iron concentration of 0.70 mg/L. Since ash placement began in 1997, total iron concentrations have dropped, often to below 0.2 mg/L. Iron concentrations respond to ash input in the Big Gorilla. Lower-iron aqueous concentrations are most likely due to the high pH, which causes the precipitation of iron.

Pre-ash-placement concentrations of manganese in the Big Gorilla ranged from 0.71 mg/L to 0.97 mg/L. With two apparently anomalous exceptions, manganese concentrations in the mine pool have been below 0.1 mg/L since October 1997 and have often been less than the 10- μ g/L detection limit.

The highest value given by Becher (1991) for arsenic in the Llewellyn and Pottsville Formations is 0.005 mg/L. No values for background concentrations of selenium were given by Becher (1991), and all measurements of selenium for the pre-ash-placement mine pool were less than the detection limit of 0.002 mg/L. The maximum concentration for these elements was 0.101 mg/L for Se and 0.022 mg/L for As between August 1997 and May 2002 in the Big Gorilla mine pool.

Maximum concentrations of cadmium and nickel measured by Becher (1991) in groundwater of the Llewellyn and Pottsville Formations were 0.002 mg/L and 0.440 mg/L, respectively. No values were given for mercury. There was no detectable change in cadmium, mercury, or nickel concentrations in the Big Gorilla mine pool between August 1997 and June 2000. The values during that period were comparable to those measured for the pre-ash-placement mine pool. The highest concentrations measured in the Big Gorilla mine pool for cadmium, mercury, and nickel were 0.018 mg/L, 0.003 mg/L, and 0.284 mg/L, respectively.

Chromium concentrations measured in the Llewellyn and Pottsville Formations by Becher (1991) were no higher than 0.02 mg/L; samples from the pre-ash-placement Big Gorilla had chromium concentrations no higher than 0.05 mg/L. Chromium concentrations in the post-ash-placement Big Gorilla mine pool were never above 0.260 mg/L.

Of the four analyses taken in 1993, the highest lead concentration was 0.002 mg/L. The highest post-ash-placement concentration of lead of 0.0274 mg/L in the Big Gorilla was measured in August 1999.

The highest concentration of copper measured in groundwater of the Llewellyn and Pottsville Formations was 0.047 mg/L (Becher, 1991). The highest value for copper in the pre-ash-placement Big Gorilla mine pool was 0.030 mg/L. The maximum concentration of copper detected in the Big Gorilla after ash placement is 0.132 mg/L.

4.11.6 Groundwater Monitoring Well Chemistry

Five monitoring wells and three test borings on and near the NEPCO property have been sampled for water quality. Wells 1 and 2 are to the east of the Big Gorilla mine pool. Well 3 is within 100 feet of the southern boundary of the mine lake. Well 9 is to the west of the Big Gorilla and is associated with a Superfund site that is between Route 309 and the NEPCO property. Well 4, installed in July 2001, was to intersect the Buck Mountain vein. Table 4.25 presents average concentrations of a number of chemicals measured in the five monitoring wells. These results show that each well has a distinct chemical signature.

Only eight groundwater samples were collected from Well 1, but the iron concentration is significantly higher than that found in Well 2. Well 2 has higher concentrations of sodium and chloride than does Well 1. Both Wells 1 and 2 have much lower concentrations of sulfate and aluminum compared to Wells 4 and 9. Wells 4 and 9 are more heavily impacted by AMD than Wells 1 and 2. Well 4 intersects the Buck Mountain seam, and Well 9 is downgradient of surface culm piles and coal refuse.

The largest change in water quality in Well 3 occurred between mid-1999 and mid-2000. The pH value increased from 4 to 4.5–5. Sulfate and calcium concentrations dropped from about 125 to 50 mg/L and from 30 mg/L to 10 mg/L, respectively. With a change in pH, iron and aluminum concentrations dropped significantly.

From the beginning of ash input to the Big Gorilla surface mine pool, the concentrations of acidity were being reduced. In January 2000, acidity was less than 10, while the alkalinity concentration in Well 3 was 4.8 mg/L CaCO₃. For the next 14 months, acidity was below 15 mg/L CaCO₃, and alkalinity was 2.0 mg/L CaCO₃ or above. During this time, iron and aluminum concentrations remained consistently low. After May 2001, despite continuing ash placement in the Big Gorilla mine pool, acidity began to rise, and alkalinity remained low. However, in April 2003 acidity again decreased and alkalinity increased slightly.

The obvious source of alkalinity in groundwater in Well 3 is the Big Gorilla surface mine pool. At times, the alkalinity levels in the Big Gorilla have been over 600 mg/L CaCO₃. However, iron concentrations in groundwater in Well 3 were initially below 0.5 mg/L, but rose to a maximum of 6.6 mg/L with the additional input of Big Gorilla water. After November 1999, iron concentrations generally remained low (<0.5 mg/L). Just as iron and alkalinity values were increasing in Well 3, sulfate and calcium concentrations began decreasing.

Two test borings were drilled into the lower ash platform, and one boring was drilled into the culm. PVC pipe was placed in each hole so that water samples could be collected. The water samples from the borings in the ash platform are very similar in chemistry to the Big Gorilla. All of the borings in the ash platform have consistently high pH and calcium values. It appears that the CaO component of the ash has raised the pH values.

4.11.7 Conclusions

The Big Gorilla mine pool project involved placement of ash to fill the mine pit containing the large amounts of AMD water. Over 3 million tons of fly ash has been beneficially used on the former acidic surface mine pool, which will no longer present a potential environmental hazard to nearby residents. In addition, abandoned culm piles are removed from local communities, resulting in a further decrease in acid mine drainage production.

While the Silverbrook outflow chemistry has changed since 1989, most likely in response to land reclamation in the basin. To date, the change in the chemistry of the Silverbrook has not been in the form of increasing heavy metal content or other toxic chemicals. The Big Gorilla water has maintained a consistently high pH value (11 to 12) in response to the placement of ash. Alkalinity varied with ash deposition. A small amount of the calcium oxide (<2%) derived from the limestone added in the fluidized-bed reactor appears to have been controlling the pH in the Big Gorilla mine pool.

Iron, manganese, magnesium, aluminum, and zinc have all decreased significantly since ash placement began. Based on the chemistry of Well 3, it appears that there were only low levels of drainage from and/or flow to the Big Gorilla. Once the area above the mine pool is returned to the approximate premining grade, very little water will flow through the ash plug in the former Big Gorilla mine pool. Thus, one long-term effect of ash placement in the former Big Gorilla mine pool will be the prevention of acidic water production through the surface mine pool and interconnected passages.

4.12 Red Oak Mine (OK)

This case study illustrates the use of CCPs for in situ chemical treatment to ameliorate the adverse impacts associated with AMD. The introduced alkalinity from the CCPs neutralized the acidity in mine pool waters, leading to an increase in pH, which in turn caused the precipitation of metals as hydroxides and carbonates within the mine, thereby improving water quality.

The Red Oak Mine was operated from 1907 to 1925 utilizing the room-and-pillar extraction method. The mine, which covers approximately 46.5 acres, contains water pools or reservoirs of AMD. The University of Oklahoma and the Oklahoma Conservation Commission injected 418 tons of FBC ash in 15 hours into this acidic (pH 4.3) flooded mine to chemically alter the mine water. Table 4.26 provides a summary of the chemical characteristics of the seep water from the mine pool.

Several chemical parameters were monitored for a 1-year postinjection period as well as during injection. Canty and Everett (1999) reported that there were three phases in the treatment of AMD water by the alkaline material. In Phase I, neutralization of existing acidity resulted in an increase in pH from 4.4 to 12.2 in 15 hours. Alkalinity (as CaCO₃) increased from 0 to 950 mg/L, and precipitation of metals occurred. In Phase 2, carbonate alkalinity formed and reached a minimum of 50 mg/L for about 3 weeks and then increased to about 100 mg/L for the following several months. The pH declined (Figure 4.35) to below 7, and concentrations of Fe,

Mn, and Al increased (Figure 4.36). Phase 3 represents a near-equilibrium condition where pH and alkalinity have leveled off at approximately 6.3 and 150 mg/L as CaCO₃, respectively.

A comparison of the AMD water preinjection and about 12 months after CCP injection shows that Fe decreased from 200 to 120 mg/L, Mn decreased from 7 to 4.8 mg/L, Al decreased from 6 to 2.3 mg/L, and pH increased from 4.4 to 6.3. The injection of 418 tons of CCPs has introduced a finite amount of alkalinity to the mine pool, which has resulted in an increase in pH and a decrease in the concentration of metals. However, the finite amount of added alkalinity will be depleted as the neutralization of newly produced AMD continues, and the mine water will most likely return to preinjection conditions. This is expected to occur slowly over several years. Further studies will quantify the longevity of the treatment.

Trace metals (arsenic, barium, cadmium, chromium, copper, lead, molybdenum, nickel, selenium, and zinc) were measured in seep water before and after CCP injection. These data indicate that trace metal concentrations were either less than preinjection levels or remained the same after CCP injection.

4.13 Winding Ridge Demonstration Project at Frazee Mine (MD)

4.13.1 Introduction

The Maryland Department of Natural Resources Power Plant Research Program and the Maryland Department of Environment Bureau of Mines have launched a joint effort with private industry to demonstrate large-volume beneficial uses of CCPs to create flowable grouts for placement of abandoned, underground coal mines to reduce acid formation. In April 1995, this multiyear project initiative started with the Winding Ridge Demonstration project involving injection of a 100% CCP grout into the Frazee Mine, located near Friendsville, Garrett County, Maryland.

4.13.2 Characteristics of Frazee Mine

The Frazee Mine is located in the Appalachian Plateau region and is overlain by about 100 ft of shale and sandstone overburden. The mine was a small mine that was used to extract coal from the Upper Freeport coal seam and was operated from 1930s through 1960. Frazee Mine is an abandoned 10-acre underground coal mine consisting of two main tunnels connected by various crosscuts (Figure 4.37). AMD occurs through one mine opening at a steady rate of about 5 gpm. Total mine voids were initially estimated to about 3900 cubic yards.

4.13.3 Geology and Hydrogeology

The bedrock above the mine is sandstone and shale with thickness of about 100 feet. The bedrock below the mine consists of shale with occasional coal seams. The mine floor consists of a dense, dry, weathered shale and clay of low permeability.

Water-level data obtained from the boreholes show that the mine contains submerged and dry conditions. The upper tunnel appears to be essentially dry and the mine pool is present in the lower tunnel, generally at an elevation of about 2160 ft msl.

4.13.4 CCP Grout Mix

The grout mixture used for the Frazee Mine project was about 60% FBC material from Morgantown Energy Associates Power Plant mixed with 20% FGD material and 20% Class F fly ash both from Mt. Storm Power Plant of Virginia Power Company. The median particle size of the FBC material was 42.3 μm and 15 μm for the Class F fly ash. The FGD showed some variation with the average particle size as 7.7 μm . The FBC material contained about 5% free lime. The Atterberg tests were used to determine that the liquid limit and plastic limit for the fly ash were 29.3% and 27.1%, respectively. The FBC and FGD by-products were nonplastic. The chemical composition data of the grout admixtures based on x-ray fluorescence (XRF) analysis are presented in Table 4.27. The x-ray diffraction (XRD) analysis showed that calcium minerals were present in the FBC dominated by hannebachite, gypsum, anhydrite, and calcium oxide, with some quartz. The FGD material did not contain any free lime (CaO) and the calcium was primarily calcite, anhydrite, and hannebachite.

The grout mix was also tested by TCLP to obtain potential leaching characteristics of the mixture. The TCLP test results showed that the grout mixture was not a characteristic hazardous material. Arsenic and barium were found to leach at levels of 0.13 and 0.11 mg/L, respectively. The remaining six TCLP metals were not detected in the TCLP leachate.

4.13.5 Grout Injection

CCP grout was injected in Frazee Mine from October 3, 1996, through November 7, 1996. Approximately 5600 cubic yards of CCPs were injected. Altogether 3800 tons of FBC material and 1200 tons of FGD material and Class Fly ash were utilized for the grout mixed. Mine water was pumped and used in mixing the grout.

A portable mixing plant was utilized to mix the grout and was pumped to the injection boreholes through steel pipes. Grout mixing required about 6 to 12 minutes for each 5 cubic yard batch and average injection rate was about 220 cubic yards for a 10-hour work day.

4.13.6 Pre- and Post-Injection Monitoring

The preinjection water quality was typical of AMD quality water. The pH values ranged from 2.5 to 3.45 sulfate concentrations ranged from 80 to 1800 mg/L, TDS ranged from 160 to 2900 mg/L, total acidity ranged from 50 to 2400 g/L, and total iron concentrations ranged upwards of 200 to 300 mg/L.

Postinjection monitoring has been performed for water quality since 1996 to 2004. Robin Guynn of ERM, Inc., has kindly provided the summary tables and time-series plots for the L-2 seep location showing both the pre- and the post-injection measurements for calcium (Figure 4.38), iron (Figure 4.39), magnesium (Figure 4.40), potassium (Figure 4.41), sodium

(Figure 4.42), bromide (Figure 4.43), chloride (Figure 4.44), sulfate (Figure 4.45), aluminum (Figure 4.46), manganese (Figure 4.47), zinc (Figure 4.48), cobalt (Figure 4.49), copper (Figure 4.50), cadmium (Figure 4.51), nickel (Figure 4.52), total acidity (Figure 4.53), pH (Figure 4.54), flow rate (4.55), and TDS (Figure 4.56). Table 4.27 shows the average concentrations of the chemical constituents measured in L-2 seep during the preinjection period from January 1995 through September 1996 as well as for the post injection period from October 1998 through December 2004.

In addition, grout cores have been collected on a periodic basis from the Frazee Mine. Visual inspection and physical testing results indicate that the grout is in very good shape without evidence of in situ weathering. The unconfirmed compressive strength ranged from 560 to 1417 psi. The hydraulic conductivity of the cores ranged from 10^{-8} cm/sec to 10^{-6} cm/sec.

Overall, the water quality-monitoring results, presented in Table 4.27 and the time-series plots presented in Figures 4.38 through 4.56, show no significant increases in AMD-related parameters since the CCP grout was placed in the Frazee Mine.

4.13.7 Conclusion

The Winding Ridge demonstration project at the Frazee Mine has shown that CCP grout mixture can be beneficially used for an abandoned underground coal mine to reduce acid formation as well to fill mine voids with a high-strength, low-permeability material that would control mine subsidence. The placement of the CCP grout appears to have not caused an unacceptable water quality impact either.

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APPENDIX A

TABLES

Table 2.1 Summary of Total Concentrations of Selected Elements in Fly Ash, Bottom Ash, and FGD Sludge (from EPRI 1987)

Element	Concentration (mg/kg solid)								
	Fly Ash (39 samples)			Bottom Ash (40 samples)			FGD Sludge (11 samples)		
	ND ^a	Mean	Range	ND ^a	Mean	Range	ND ^a	Mean	Range
Al	39	113,000	46,000–152,000	40	101,000	30,500–145,000	10	30,300	8400–73,000
As	39	156.2	7.7–1,385	34	7.6	<5–36.5	8	25.2	<5–53.1
Ba	39	1880	251–10,850	40	1565	150–9360	10	370	<25–2280
Ca	39	62,000	7400–223,000	40	52,000	2200–241,000	11	274,000	110,000–345,000
Cd	2	11.7	6.4–16.9	0	< 5	<5	0	< 5	<5
Cl	7	477	180–1190	8	712	<150–2630	9	2514	<150–8970
Cr	29	247.3	37–651	38	585	<40–4710	8	72.7	<40–168
Cu	39	185	44.6–1452	40	78.2	27.3–146	11	31.8	8.0–121
Fe	39	76,000	25,000–177,000	40	105,000	20,200–201,000	11	64,000	1300–138,000
K	39	14,300	3000–25,300	40	12,000	2600–24,000	11	1900	3700–17,000
Mg	39	11,800	1600–41,800	40	9700	2500–46,000	11	7200	600–17,800
Mn	39	357	44–1332	40	426	56–1940	11	138	37.3–312
Mo	36	43.6	7.1–236	36	14.4	2.8–443	8	12.2	<4.0–52.6
Na	39	9087	1300–62,500	40	6188	814–41,300	11	3817	200–16,500
Ni	39	141	22.8–353	39	216	<10–1,067	11	40.4	<5–145
P	24	3070	1100–10,340	19	1470	<500–4630	10	4271	<1000–5900
Pb	39	170.6	21.1–2120	38	46.7	4.6–843	10	29.2	<4–181
S	39	12,643	1300–64,400	40	7422	460–74,000	11	152,000	39,400–228,000
Sb	7	42.5	11–131	0	<10	<10	1	15.5	15.5
Se	30	14.0	5.5–46.9	10	4.1	<1.5–9.96	8	35.8	<2–162
Si	39	209,000	89,500–275,000	40	222,000	51,000–312,000	5	66,000	<500–170,000
Sn	18	43.6	7.9–56.4	24	28.2	<9–90.2	6	29.9	<10–50.5
Sr	39	1331	204–6,820	38	992	182 – 6,440	11	608	70.8–2990
Ti	39	6644	1310–10,100	40	5936	1,540 – 11,300	10	1,599	120–5050
U	13	18.7	11.1–30.4	16	11.9	<5 – 25.9	1	16.2	16.2
V	35	271.5	<95–652	30	176	<50 – 275	3	156	<50–261
Zn	39	449.2	27–2880	40	127	3.8 - 515	11	99.7	7.7–612

* ND represents the number of measured values above the detection limit for a given element.

Table 2.2. Ranges of Hydraulic Conductivity in Surface Mine Spoils (Hawkins, 2002)

Geographic Area	Hydraulic Conductivity (M/A)
Western Pennsylvania	4.2×10^{-9} to 7.6×10^{-2}
Northern West Virginia	9.5×10^{-8} to 2.7×10^{-4}
Eastern Ohio	5.4×10^{-8} to 1.9×10^{-5}
Western Kentucky and Southern Illinois	2.4×10^{-5} to 4.1×10^{-5}
Western North Dakota	2.9×10^{-5} to 4.6×10^{-5}
Northern Great Plains	8.0×10^{-7}
Wyoming	1.9×10^{-6} to 2.1×10^{-4}
Edmonton, Alberta	1.5×10^{-6}

Table 2.3 Chemical Analyses of Groundwater from Pre-mined Areas in the Denver Basin (from Kirkham and O'Leary, 1980)

Sample No.	Location	pH	Dissolved Concentration (mg/L)													
			Alkalinity (as CaCO ₃)	Ca	Mg	K	Na	Cl	SO ₄	F	NO ₂ + NO ₃	P	Si	Fe	Mn	TDS
DB-1	Watkins	7.7	230	230	95	4.4	380	140	1300	1.1	8.3	0.03	19	150	10	2440
DB-2		7.9	260	27	2.7	3.7	130	93	7.2	0.9	0.14	0.00	8.8	80	30	436
DB-3		7.8	160	92	15	5.3	39	50	140	0.6	6.3	0.07	29	20	3	495
DB-4		7.8	200	120	22	3.1	20	3.5	200	1.8	15	0.00	20	200	20	592
DB-5		8.0	180	86	16	3.7	43	33	140	0.5	7.2	0.03	24	10	3	486
AVG		7.84	206	111	30	4.0	122	64	357	0.98	7.4	0.03	20	92	13	890
DB-6	West Bijou Creek	7.6	88	97	8.2	6.9	87	14	430	0.8	0.14	0.00	11	1000	370	709
DB-7		8.4	240	5.6	0.6	1.6	160	8.6	130	1.7	0.59	0.07	9.7	80	9	465
DB-8		8.3	230	4.4	0.6	1.4	140	7.7	68	1.8	0.80	0.10	9.9	70	10	376
DB-9		7.7	170	99	13	5.6	110	4.3	370	0.5	2.6	0.00	19	670	120	736
DB-10		8.0	190	62	4.6	4.4	100	5.9	200	2.0	0.61	0.02	8.5	20	30	504
DB-11		8.2	230	13	1.2	2.4	180	7.1	200	1.1	0.49	0.03	9.9	60	20	555
AVG	8.03	191	47	4.7	3.7	130	7.9	233	1.3	0.87	0.04	11	317	93	558	
DB-12	Keenesburg	7.7	430	280	140	35	600	180	1700	4.7	27	0.01	17	10	300	3340
DB-13		8.1	540	3.0	0.8	2.1	300	110	5.7	4.1	0.39	0.05	11	60	10	781
DB-14		7.7	250	160	56	21	180	63	580	2.3	35	0.01	20	60	100	1390
DB-15		7.7	250	160	66	9.6	240	200	830	2.2	3.9	0.05	25	0.0	80	1800
DB-16		8.2	540	2.9	0.7	2.2	290	110	11	4.1	0.15	0.01	11	40	6	769
AVG		7.88	402	121	53	14	322	133	625	3.5	13	0.03	17	34	99	1616
DB-17	Matheson	6.1	35	67	15	3.2	16	4.8	210	0.7	3.4	0.03	41	2300	620	397
DB-18		7.2	99	27	5.5	3.4	18	2.1	29	0.5	0.13	0.01	29	70	30	175
DB-19		7.6	84	24	4.3	2.8	11	1.8	11	0.4	1.7	0.07	28	20	2	141
DB-20		7.6	100	24	5.0	3.0	18	1.4	14	0.4	0.69	0.06	25	10	1	154
AVG		7.12	80	36	7.5	3.1	16	2.5	66	0.5	1.5	0.04	31	600	163	217
State of Colorado ¹ Groundwater Standard		6.5–8.5	–	–	–	–	–	250	250	2.0	10	–	–	0.3	0.05	–

¹ Human health/secondary drinking water standard or agricultural use standard, whichever is more stringent.

Table 2.4 List of CCP Mine Placement Sites

State		Mine Name	Type of Mine	CCP Types Placed
Arizona	1	Navajo Mine	Active and inactive, surface coal mine	Fly ash, bottom ash, fgd sludge
Colorado	2	Keenesburg Mine	Surface coal mine	Fly ash, bottom ash
	3	Trapper Mine	Active surface coal mine	FA, BA, SS
Illinois	4	Black Beauty, Riola Mine		
	5	Burning Star Mine		
	6	Cottonwood Mine No. 1		
	7	Forsythe Energy No. 5	Abandoned surface mine	FGD sludge, FA
	8	Freeman United Crown III		
	9	Freeman United, Fidelity No. 11		
	10	Freeman United, Orient No. 4		
	11	Harco Mine		
	12	Razorback No. 1		
	13	Creek Paum		
	14	No. 10 Randolph Prep		
	15	Razorback No. 11		
	16	Sahara, Mine No. 6		
	17	Silvercreek, Norris		
	18	Surefire, Majestic No. 14		
	19	Thunderbird Mine	Surface mine	FBC
	20	U.S.A. Coal, Mine No. 10		
Indiana	21	Arnold Willis City Mine	Abandoned underground coal mine	Scrubber sludge, fly ash
	22	Keywest Mine	Surface mine	FBC
	23	Midwestern Mine	Abandoned surface mine	Fixated scrubber sludge
	24	Center Pit	Surface mine	
	25	Little Sandy Mine	Sand and gravel mine	Fly Ash, FGD Sludge
	26	Prides Creek Mine	Active surface coal mine	Fly Ash/Bottom Ash
	27	Black Beauty, Viking	Active surface coal mine	
	28	Black Beauty, Miller Creek	Surface mine	FA,BA
	29	United Minerals, Deer Ridge	Surface mine	FA
	30	Black Beauty, Farmersberg		
Indiana	31	Black Beauty, Francisco Mine	Active Surface Coal Mine	
	32	Vigo, Cypruss Creek		
	33	Black Beauty, Somerville		
	34	Universal Mine	Surface pit	FA

Continued . . .

Table 2.4 List of CCP Mine Placement Sites (continued)

		Mine Name	Type of Mine	CCP Types Placed
Kentucky	35	Starfire Mining		
	36	Bluediamond		
	37	Buckhorn		
	38	Floyed County	Coal	FBC ash
Maryland	39	Mettiki Coal		
	40	Patriot Mining		
	41	Fraze Mine, Winding Ridge	Abandoned underground coal	Fly ash, FBC ash, FGD sludge
	42	Kempton Mine	Underground coal	
	43	BBSS Mine	Surface Sand and gravel quarry	Fly ash
	44	Rossville Mine	Surface clay	Fly ash
Ohio	45	Miller Mining, Bucks		
	46	Miller Mining,, Auburn		
	47	Painesville		
	48	Mead		
	49	Clayton		
	50	Milton		
	51	Washington		
	52	Roberts-Dawson		
	53	Broken Aro	Underground	FGD sludge
	54	Fleming Mine	Surface	FBC
Oklahoma	55	Georges Colliers		
	56	MIMHF, Inc.		
	57	Red Oak Mine	Abandoned Underground Coal Mine	FBC ash
Pennsylvania	58	Big Gorilla Pit	Surface	FBC
	59	Alden Mine	Surface	FBC
	60	Knickerbocker Pit	Surface	FBC
	61	McCloskey Site	Surface	FBC
	62	Abel Dreshman	Surface	FBC
	63	Wildwood	Surface	FBC
	64	Clinton County	Surface	FBC
	65	Centralia	Surface Coal Mine	Fly ash
	66	Morris Ridge	Surface Coal Mine	Stabilized ash
	67	Pine Hill	Inactive Coal Mine	FA, BA, mill rejects
	68	Rock Run	Inactive Coal Mine	FA, BA, mill rejects
	69	Zullinger Quarry	Abandoned Limestone quarry	Fly ash
	70	Harwick Mine	Underground	Fly ash
	71	Bognanni Mine	Surface	FBC, FA
72	Wheelabrator Mine	Surface	FBC	
73	Mckay Coal Mine	Surface	FBC	
74	Panther Creek Partners	Surface	FBC	
Texas	75	Monticello Winfield Mine	Lignite surface	Fly ash
	76	San Miguel Mine		FA, BA

Continued . . .

Table 2.4 List of CCP Mine Placement Sites (continued)

		Mine Name	Type of Mine	CCP Types Placed
	77	Big Brown Mine	Active Lignite Surface	Bottom ash
Washington	78	Centralia	Strip	BA, FA
West Virginia	79	Fairfax	Underground	FBC
	80	Omega Mine	Surface	FBC, FA
	81	Pierce	Surface	FA
	82	Mt. Storm	Active coal	FA, BA, mill rejects
	83	Long Ridge	Underground	FBC
Wyoming	84	Wyodak Mine	Active surface coal	FA, BA, SS
	85	Jim Bridger		
	86	Kemmerer Mine		
	87	Black Thunder		
	88	Arch Minerals		
Michigan	89	Sibley Quarry	Surface limestone	Fly ash
Missouri	90	Lafarge Sugar Creek		Fly ash
	91	Foster Mine		
	92	Tiger Mine		
Montana	93	Savage Mine	Surface lignite	FA, BA
New Mexico	94	San Juan Mine		
	95	Navaho		
North Dakota	96	Abby		
	97	Beulah		
	98	Buechler		
	99	Burlington		
	100	Center	Surface	FGD sludge, FA
	101	Falkirk		
	102	Freedom		
	103	Lehigh Road		
	104	Parshall		
	105	Scenic East		
	106	Wilton	Underground	Fly ash

Table 3.1. Post-mining Water Quality Data for Several Pennsylvania Mines

Mine Site	pH	Net				
		Alkalinity, mg/L	Fe, mg/L	Mn, mg/L	SO ₄ , mg/L	Al, mg/L
A	3.2	-185	25	23	1434	N/A
B	5.6	6	0	0	<43	↓
C	7.6	382	5	5	744	
D	2.9	-814	51	29	981	
E	3.1	-361	52	31	432	
F	7.4	237	0	1	32	
Well 6	3.4	-1,140	120	NA*	4600	
Discharge	3.3	-380	48	N/A	2300	↓
Clinton	2.3	N/A	876	39.2	3477	256
County	2.5	N/A	747	48.1	2958	236

* Not available.

Table 4.1. Average Annual Ash Quantities Generated by Wyodak, Neil Simpson I, and Neil Simpson II Power Plants

CCP Type	Generated Quantity of CCPs, tons/yr		
	Neil Simpson I	Neil Simpson II	Wyodak
Fly Ash	7665	–	–
Bottom Ash	2555	7280	38,325
Scrubber Ash	–	26,880	148,190

Table 4.2. Summary of Ash-Filled Pits at Wyodak Mine

Area No.	Year(s) of Ash		Compacted Liner
	Placement	Ash Type	
1	1978–1979	Fly ash	Yes
2	1980	Fly ash	No
2	1981	Fly ash	No
2	1982	Fly ash	No
3	1983	Bottom ash	Yes
3	1983	Fly ash	Yes
3	1984	Fly ash	Yes
3	1985–1987	Fly/scrubber ash	Yes
3	1988–1989	Scrubber ash	Yes
3	1990–1991	Scrubber ash	Yes
3	1992–1995	Scrubber/bottom ash	Yes
3	1995–1998	Scrubber/bottom ash	Yes
4	1998–2009	Scrubber/bottom ash	No

Table 4.3. Results of 1983 Column Leaching Study for Wyodak (WY) Bottom Ash

Analyte in Effluent	Effluent Concentration, mg/L unless otherwise noted		Difference, %	Wyoming Groundwater Standard ¹
	Overburden	Bottom Ash		
pH	8	8.2	2.5	6.5–8.5 (6.5-9.0)
TDS	3840	4790	24.7	5000 (500)
Al	<0.1	1	–	5.0 (–)
As	<0.005	< 0.005	0.0	0.2 (0.05)
B	0.07	6.97	9857	5.0 (0.75)
Ba	< 0.5	< 0.5	0.0	– (1.0)
Ca	473	634	34.0	–
Cd	<0.002	< 0.002	0.0	0.05 (0.01)
Cl	250	301	20.4	2000 (250)
Cr	0.02	0.03	50.0	0.05 (0.05)
Cu	0.01	0.02	100	0.5 (1.0)
F	0.96	0.39	–59.4	– (2.4)
Fe	<0.05	< 0.05	0.0	– (0.3)
Hg	<0.001	< 0.001	0.0	0.00005 (0.002)
K	46	45	–2.2	–
Mg	326	277	–15.0	–
Mn	0.13	< 0.02	–100	– (0.05)
Mo	<0.02	0.02	0.0	–
Na	248	413	66.5	–
NH ₃ -N	1.53	0.46	–69.9	– (0.58)
Ni	0.06	0.04	–33.3	–
NO ₂ + NO ₃	12.4	15.4	24.2	100 (–)
Pb	<0.02	< 0.02	0.0	0.1 (0.05)
Se	<0.005	< 0.005	0.0	0.05 (0.01)
SO ₄	2250	3060	36.0	3000 (250)
Zn	0.02	< 0.01	–100	25 (5.0)

¹ Class III standards for livestock watering and Class I standards for domestic use (in parentheses).

Table 4.4. Batch Test Results from 1997–1998 on Wyodak Bottom and Scrubber Ash

Analyte	Groundwater Concentration ¹ , mg/L unless otherwise noted				Wyoming Groundwater Standard ²
	Bottom Ash Test		Scrubber Ash Test		
	Preextraction	Postextraction	Preextraction	Postextraction	
pH (s.u.)	7.3	8.0	7.25	10.48	6.5–8.5 (6.5–9.0)
TDS	3220	3280	3285	3497	5000 (500)
Al	0.79	1.93	0	3.88	5.0 (–)
As	0	0.006	0	0.008	0.2 (0.05)
B	0.37	1.7	0.31	2.69	5.0 (0.75)
Ba	0	0	0	0.25	– (1.0)
Ca	321	279	347	483	–
Cd	0	0	0	0	0.05 (0.01)
Cl	127	119	160	208	2000 (250)
CO ₃	0	0	0	54.5	
Cr	0	0	0	0.25	0.05 (0.05)
Cu	0	0	0	0	0.5 (1.0)
F	0.37	0.22	0.15	3.15	– (2.4)
Fe	0	0	1.74	0.42	– (0.3)
HCO ₃	1120	905	1098	873	
Hg	0	0	0	0	0.00005 (0.002)
K	14.4	16.9	17.5	34.7	–
Mg	234	257	221	270	–
Mn	1.29	0.07	1.21	0.24	– (0.05)
Mo	0	0	0	0.12	–
Na	413	399	463	493	–
NH ₃ -N	1.42	1.41	2.03	2.75	– (0.58)
Ni	0.01	0.02	0.01	0.03	–
NO ₂ +NO ₃	0	0.09	0	1.48	100 (–)
Pb	0	0	0	0	0.1 (0.05)
Se	0	0.005	0	0.027	0.05 (0.01)
SO ₄	1560	1760	1538	2082	3000 (250)
Zn	0	0	0	0	25 (5.0)

¹ Post-extraction value represents the maximum concentrations in 3 to 10 sequential extractions with fresh groundwater.

² Class III standards for livestock watering and Class I standards for domestic use (in parentheses).

Table 4.5. Groundwater Quality Data for Regulated Analytes in Selected Wells

Analyte	Wyoming Groundwater Standard ¹	Groundwater Concentration, mg/L unless otherwise noted ²											
		M-1A (1987–1999)		LF-5 (1990–1999)		M-5A (1984–1999)		M-12 (1977–1981)		M-17 (1977–1999)		M-22 (1981–1999)	
		Max	Mean	Max	Mean	Max	Mean	Max	Mean	Max	Mean	Max	Mean
pH (s.u.)	6.5–8.5	8.1	7.11	7.8	6.7	7.8	7.22	7.8	7.51	8.2	7.51	8.1	7.57
TDS	5000	2954	2403	5468	4831	4220	2558	2520	2520	2200	1781	842	759
Al	5.0	0.1	0.08	0.3	0.07	2.1	0.152	0.5	0.25	0.3	0.1	0.4	0.098
As	0.2	<0.005	<0.005	0.009	<0.005	<0.005	<0.005	<0.005	<0.005	<0.01	<0.005	<0.005	<0.005
B	5.0	0.21	0.09	0.42	0.29	0.31	0.098	6.90	3.49	0.5	0.212	0.22	0.092
Ba	–	0.5	0.36	0.25	0.27	0.5	0.40	0.5	0.5	0.5	0.43	0.9	0.513
Cd	0.05	0.003	0.002	0.003	0.0016	0.006	0.0017	0.03	0.01	0.03	0.0039	0.003	0.0017
Cl	2000	57.28	14.64	224.04	149.8	16.81	10.29	12	4.98	69.16	9.34	29.87	16.83
Cr	0.05	0.02	0.014	0.1	0.014	0.04	0.016	0.1	0.073	1.0	0.05	0.03	0.02
Cu	0.5	0.01	0.03	0.06	0.008	0.02	0.0082	0.01	0.027	0.06	0.012	0.05	0.0095
F	–	0.81	0.454	0.57	0.336	3.77	0.508	0.37	0.37	0.5	0.33	1.6	0.93
Fe	–	2.47	0.228	0.83	0.2	2.06	0.233	22.6	5.865	2.75	0.412	1.11	0.171
Hg ³	0.00005	0.002	<0.001	<0.001	<0.001	0.002	<0.001	<0.001	<0.001	0.002	<0.001	<0.001	<0.001
Mn	–	1.57	0.56	2.41	1.52	0.21	0.111	0.37	0.257	0.19	0.107	0.04	0.018
NH ₃	–	3.07	0.81	1.86	0.82	10.42	3.12	2.54	2.54	11.86	3.86	3.47	1.67
NO ₂ +NO ₃	100	2.42	0.347	0.89	0.121	4.35	0.26	2.8	1.01	4.1	0.68	0.35	0.0475
Pb	0.1	0.02	0.014	0.01	0.011	0.02	0.016	0.02	0.03	0.02	0.017	0.02	0.0165
Se	0.05	0.005	0.0035	0.0025	0.0027	0.005	0.0039	0.005	0.005	0.01	0.0044	0.005	0.004
SO ₄	3000	1427	1255	2930	2501	2543	1318	1510	1279	912	794	15	1.51
Zn	25	0.41	0.066	0.47	0.05	0.34	0.039	0.2	0.143	0.55	0.141	0.07	0.012

¹ Values represent Wyoming Class III livestock use standards.

² Values in boldface exceed Class III standards.

³ The reported detection limit for Hg (0.001 mg/L) is higher than the current Class III standard (0.00005 mg/L).

Table 4.6. Paste Extract Analysis of Fly Ash and Bottom Ash Samples Prior to Disposal at Keenesburg Mine (CO) (Hazen Research Inc., 1989)¹

Analyte	Units	Bottom Ash	Fly Ash
pH	s.u.	11	9.42
Electrical Conductivity	mS/cm	2650	11,900
Arsenic	mg/L	<0.01	0.13
Calcium	mg/L	579	494
Cadmium	mg/L	<0.001	0.008
Chromium	mg/L	0.02	0.17
Copper	mg/L	0.01	0.09
Iron	mg/L	0.04	0.11
Mercury	mg/L	<0.0001	<0.0001
Potassium	mg/L	44.1	89.3
Magnesium	mg/L	0.6	73.2
Manganese	mg/L	<0.01	0.02
Molybdenum	mg/L	0.99	13.3
Sodium	mg/L	260	4410
Nitrate	mg/L	1.39	1.31
Lead	mg/L	0.17	0.12
Selenium	mg/L	0.031	2.42
Sulfate	mg/L	1950	10,500
Zinc	mg/L	0.03	0.05

¹ Pastes prepared using 0.42 g H₂O/1 g ash.

Table 4.7. Results of Two Toxicity Characteristic Leaching Procedure Tests on Keenesburg Fly Ash and Bottom Ash Samples (Hazen Research Inc., 1991)

Analyte	Units	January 1991		December 1991		EPA Limit
		Fly Ash	Bottom Ash	Fly Ash	Bottom Ash	
Silver	mg/L	<0.05	<0.05	<0.05	<0.05	5.0
Arsenic	mg/L	<0.01	<0.01	<0.05	<0.01	5.0
Barium	mg/L	1.6	<0.1	2.4	<0.1	100
Cadmium	mg/L	0.02	<0.01	<0.01	<0.01	1.0
Chromium	mg/L	<0.05	<0.05	<0.05	<0.05	5.0
Mercury	mg/L	<0.0001	<0.0001	<0.0001	<0.0001	0.2
Lead	mg/L	<0.1	<0.1	<0.3	<0.1	5.0
Selenium	mg/L	0.29	0.03	0.08	0.16	1.0
Final pH	s.u.	5.02	4.74	5.70	5.64	—

Table 4.8. Historic Groundwater Concentrations (mg/L) of Selected Analytes at Keenesburg Mine (CO)

Analyte	Standard, ¹ mg/L	Well	Pre-Ash-Placement (1978–1986)		Post-Ash-Placement (1988–2000)	
			Max	Mean	Max	Mean
As	0.05 (0.1)	DH96	NA	NA	0.11	0.11
		DH122	NA	NA	0.11	0.11
Ba	2.0	AMW-1	NA	NA	0.03	0.03
		DH96	NA	NA	0.14	0.14
		FPW	NA	NA	0.05	0.05
Cl	250	SMW-2	950	815	920	766
Fe	0.3 (5.0)	DH96	0.464	0.464	NA	NA
		DH122	NA	NA	21	12.0
		FPW	1.15	0.69	NA	NA
Mn	0.05 (0.2)	SMW-2	2.7	2.25	1.1	1.015
		DH96	0.53	0.46	1.5	0.52
		DH122	0.96	0.74	1.2	0.71
		DH172	1.3	0.94	NA	NA
		FPW	0.14	0.10	0.33	0.16
		SMW-2	0.31	0.27	0.82	0.38
NO ₃	10.0	AMW-1	NA	NA	4.6	4.4
		DH96	9.7	6.4	NA	NA
		DH122	11.2	3.86	NA	NA
		FPW	9.8	4.7	NA	NA
Se	0.05	AMW-1	NA	NA	0.06	0.028
		DH122	NA	NA	0.05	0.05
		DH133	NA	NA	0.075	0.075
SO ₄	250	AMW-1	NA	NA	990	395
		AMW-2	NA	NA	4100	3567
		DH96	897	716	860	736
		DH122	591	515	690	578
		DH133	3300	3005	NA	NA
		FPW	514	364	280	265
		SMW-2	3500	3300	3800	3047
TDS	400 mg/L or 1.25× background level, whichever is least restrictive	AMW-1	NA	NA	1200	887
		AMW-2	NA	NA	7700	5833
		DH96	1750	1464	1400	1363
		DH122	2934	1818	1300	1175
		DH133	4710	4252	NA	NA
		FPW	720	601	610	610
		SMW-2	7300	6840	7300	6481

¹ State of Colorado basic standards for groundwater (values in parentheses represent standards for agricultural use).

**Table 4.9. Chemical analysis of Craig fly ash and bottom ash
(Radian Corporation, 1991)**

Analyte	Concentration, mg/kg	
	Fly Ash	Bottom Ash
Aluminum	123,000	110,000
Arsenic	4.0	4.2
Barium	6700	5800
Boron	30	12
Cadmium	19	17
Calcium	65,000	63,000
Chlorine	0.0017	–
Chromium	81	57
Cobalt	30	22
Copper	69	40
Iron	37,000	44,000
Lead	42	16
Magnesium	15,700	13,000
Manganese	300	360
Mercury	0.02	0.02
Molybdenum	47	36
Nickel	54	36
Phosphorous	1900	2052
Potassium	9400	7800
Selenium	0.56	0.56
Silicon	250,000	285,000
Silver	6.9	4.7
Sodium	2400	2200
Strontium	2500	2400
Sulfur	1400	1800
Titanium	5500	5200
Vanadium	380	290
Zinc	170	96

Table 4.10. Results of Batch Leaching Tests on Trapper Mine (CO) Fly Ash and Bottom Ash (Radian Corporation, 1991)

Analyte	Units	Fly Ash RCRA Extraction ¹	Fly Ash DI Water Extraction ²	Bottom Ash DI Water Extraction ²	Groundwater Quality Standards ³
pH	s.u.	–	11.4	8.6	6.5-8.5
Ag	mg/L	0.014	0.014	0.008	0.05
Al	mg/L	0.1	92	1.4	(5.0)
As	mg/L	<0.005	<0.005	<0.005	0.05 (0.1)
B	mg/L	12	42	0.55	(0.75)
Ba	mg/L	1.6	12	1.6	2.0
Cd	mg/L	<0.005	< 0.005	<0.005	0.005 (0.01)
Cl	mg/L	<1	<1	12	250
Cr	mg/L	0.025	0.18	0.015	0.1 (0.1)
Cu	mg/L	0.027	0.005	<0.005	1.0 (0.2)
Fe	mg/L	0.017	0.032	0.01	5.0 (0.3)
Hg	mg/L	<0.002	0.002	<0.002	0.002 (0.01)
Mn	mg/L	0.2	0.036	0.036	0.05 (0.2)
Mo	mg/L	0.12	0.48	0.04	–
Ni	mg/L	0.02	0.06	0.02	0.1 (0.2)
Pb	mg/L	<0.002	<0.002	<0.002	0.05 (0.1)
Se	mg/L	<0.002	<0.002	<0.002	0.05 (0.02)
SO ₄	mg/L	80	<3	190	250
V	mg/L	0.41	0.038	0.056	(0.1)
Zn	mg/L	0.09	0.01	0.01	5.0 (2.0)

¹ RCRA extraction performed with acetic acid buffer solution (pH ~ 5.0).

² DI water extraction performed with distilled water.

³ Drinking water standards from Colorado “Basic Standards for Groundwater” (Regulation No. 41, effective March 2, 1999); values in parentheses represent standards for agricultural use.

Table 4.11. Historic groundwater quality summary for Trapper Mine

Analyte	Standard ²	Analyte Concentrations, mg/L unless otherwise noted															
		Well GF6 (1982–2000)				Well GF7 (1987–2000)				Well GF11 (1988–2000)				Well GMP1 (1990–2000)			
		Max	Mean	NS	ND	Max	Mean	NS	ND	Max	Mean	NS	ND	Max	Mean	NS	ND
Al	(5.0)	0.37	0.026	42	34	0.13	0.01	45	39	<0.05	<0.05	22	22	0.25	0.02	34	27
As	0.05 (0.1)	0.034	0.0065	19	3	0.004	<0.001	15	13	0.002	<0.001	13	10	0.004	<0.001	11	8
B	(0.75)	0.24	0.12	45	3	0.3	0.22	45	0	0.19	0.13	22	0	0.36	0.24	34	0
Ba	2.0	0.42	0.11	45	7	0.3	0.046	45	0	0.2	0.067	22	0	0.23	0.12	34	0
Cd	0.005	<0.003	<0.003	20	20	<0.003	<0.003	16	16	0.02	<0.003	14	13	<0.003	<0.003	12	12
Cl	250	81	17.5	19	0	67	40.3	15	0	25	118.5	13	0	35	23.8	11	0
Cr	0.1	0.01	<0.001	42	40	<0.01	<0.01	44	44	0.01	<0.01	21	18	<0.01	<0.01	33	33
Fe	0.3 (5.0)	0.1	0.009	27	21	0.31	0.076	17	6	0.95	0.24	14	4	0.25	0.05	12	4
Mn	0.05	0.06	0.016	44	17	0.76	0.41	45	4	1.35	0.58	22	0	0.17	0.027	34	13
Mo	–	0.03	<0.01	44	42	0.09	<0.01	45	44	<0.05	<0.05	22	22	<0.05	<0.05	33	33
NH ₃	–	1.83	0.9	19	1	6.6	3.82	15	0	2.12	0.79	13	0	0.85	0.63	11	0
NO ₃ +NO ₂	10.0 (100)	0.34	0.063	8	4	2.13	0.31	7	5	1.23	0.19	8	5	0.61	0.17	7	3
Pb	0.05 (0.1)	0.02	<0.02	19	18	0.06	<0.02	15	12	0.04	<0.02	14	12	<0.05	<0.05	11	11
pH (s.u.)	6.5 - 8.5	8.6	7.67	32	0	8.5	7.58	31	0	8.44	7.01	31	0	8.7	7.94	27	0
SO ₄	250	284	120	45	0	2840	1898	45	0	1090	569	22	0	873	194	34	0
TDS	1.25 × Background	980	740	42	0	4758	3390	25	0	2320	1581	39	0	1650	945	17	0

Continued . . .

¹ NS = total number of sampling events; ND = number of nondetect measurements.

² Colorado drinking water standards (agricultural use standards in parentheses).

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Table 4.11. Historic Groundwater Quality Summary for Trapper Mine (data from Trapper Mining, Inc.) (continued)

		Analyte Concentration, mg/L unless otherwise noted															
		Well GD2 (1982–2000)				Well GD3 (1988–2000)				Well GF4 (1982–2000)				Well GF5 (1984–2000)			
Analyte	Standard ²	Max	Mean	NS	ND	Max	Mean	NS	ND	Max	Mean	NS	ND	Max	Mean	NS	ND
Al	(5.0)	0.07	<0.05	42	40	2.1	0.10	22	20	2.1	0.10	60	39	1.01	0.04	53	44
As	0.05 (0.1)	0.002	<0.001	19	15	0.003	<0.001	13	10	0.003	<0.001	19	14	0.002	<0.001	17	13
B	(0.75)	0.74	0.19	44	0	0.23	0.17	22	0	0.23	0.17	62	1	0.47	0.35	54	0
Ba	2.0	0.3	0.09	45	7	0.22	0.07	22	0	0.22	0.07	64	7	0.68	0.27	53	2
Cd	0.005	<0.003	<0.003	20	20	0.005	<0.003	14	13	0.005	<0.003	21	20	0.005	<0.003	18	17
Cl	250	15	11.7	19	0	26	22.6	13	0	26	22.6	19	0	21	11.2	17	0
Cr	0.1	0.01	<0.004	44	43	0.04	<0.004	21	18	0.04	<0.004	60	57	0.01	<0.004	52	49
Fe	0.3 (5.0)	4.7	0.24	30	8	0.94	0.25	14	3	0.94	0.25	29	7	0.46	0.053	25	8
Mn	0.05	0.14	0.082	45	0	0.6	0.25	22	0	0.6	0.25	64	57	0.08	0.017	53	8
Mo	–	0.2	<0.05	43	40	<0.05	<0.05	21	21	<0.05	<0.05	62	58	<0.01	<0.01	52	52
NH ₃	–	4.1	1.7	19	0	7	1.35	13	0	7	1.35	19	0	3.59	2.85	17	1
NO ₃ +NO ₂	10.0 (100)	0.03	0.004	7	6	0.9	0.2	7	4	0.48	0.11	7	3	3.69	0.53	7	4
Pb	0.05 (0.1)	0.06	<0.02	19	17	<0.05	<0.05	13	13	0.02	<0.02	19	18	<0.05	<0.05	17	17
pH (s.u.)	6.5 - 8.5	8.74	7.25	22	0	8.4	7.02	22	0	9.3	8.86	32	0	9.15	8.06	32	0
SO ₄	250	210	99.7	45	0	564	435	22	0	250	122	64	0	251	186	53	0
TDS	1.25 x background	900	793	31	0	1640	1325	28	0	1030	848	31	0	1300	1156	28	0

¹ NS = total number of sampling events; ND = number of nondetect measurements.

² Colorado drinking water standards (agricultural use standards in parentheses).

Table 4.12. Summary of Annual Laboratory Leaching Studies from 1987 to 2000 for coal ash at Savage Mine (Holly Sugar) (Savage, MT)

Analyte	Analyte Concentration, mg/L unless otherwise noted			Federal DWS ¹	Montana GQS
	Av.	Max.	Std. Dev.		
pH (s.u.)	8.37	9.49	0.68	6.5 – 8.5	–
As	0.005	0.018	0.0052	0.05	0.02
B	9.75	17.80	5.06	–	–
Ba	NM	NM	NM	2.0	2.0
Cd	0.002	0.02	0.0059	0.005	0.005
Cl	23.6	79	25.7	250	4.0
Cr	0.005	0.02	0.0064	0.1	0.1
Cu	0.007	0.02	0.0084	1.0	1.3
F	0.497	1.22	0.417	2.0	4.0
Fe	0.071	0.26	0.091	0.3	–
Hg	0.001	0.005	0.0016	0.002	0.002
Ni	0.01	0.02	0.0083	–	0.1
NO ₂ + NO ₃	8.924	88.2	26.3	1.0	1.0
Pb	0.001	0.006	0.0018	0.015	0.015
Se	0.002	0.008	0.0025	0.05	0.05
SO ₄	2249	3824	751	250	–
Zn	0.039	0.238	0.0678	5.0	2.1

¹ Primary or secondary drinking water maximum contaminant level (MCL).

NM = not measured; DWS = drinking water standard; GQS = groundwater quality standard

Table 4.13. Historic Maximum and Mean Concentrations of Trace Metals in Selected Wells at Savage Mine (MT)¹

Analyte	Montana Detection		Well 001		Well 106		Well 261		Well 263		Well 284		Well 286		Well 289		Well 290		Well 291	
	MCL (mg/L)	Limit (mg/L)	Max	Mean	Max	Mean	Max	Mean	Max	Mean	Max	Mean	Max	Mean	Max	Mean	Max	Mean	Max	Mean
As	0.02	0.002	0.002	0.0001	0.004	0.002	0.004	0.001	0.008	0.0013	ND	ND	0.006	0.0004	0.006	0.0024	0.035	0.016	0.29	0.20
B	–	0.1	0.64	0.090	0.72	0.532	0.54	0.36	0.54	0.36	0.58	0.38	0.76	0.59	0.25	0.12	0.59	0.43	0.87	0.68
Cd	0.005	0.01 ¹	0.01	0.0013	0.01	0.0014	0.013	0.0017	0.01	0.0010	0.01	0.0007	0.01	0.0007	0.01	0.001	0.01	0.0014	0.01	0.0017
Cr	0.1	0.05	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Pb	0.015	0.002	0.027	0.0032	0.017	0.0046	0.017	0.0019	0.017	0.0028	0.004	0.001	0.007	0.001	0.016	0.0019	0.03	0.0032	0.027	0.0038
Ni	0.1	0.04	ND	ND	0.21	0.021	ND	ND	ND	ND	0.4	0.050	ND	ND	ND	ND	0.1	0.0071	0.09	0.012

¹ Nondetect readings were assigned a value of zero for evaluation of historic mean concentrations.

² Reported detection limit for Cd exceeds current Montana MCL.

Table 4.14. Savage Mine

	Federal Secondary MCLs	Max.	Min.	Downgraders	Upgraders
263	TDS (500 mg/L)	1156*	1156*	1430	1093
	SO ₄ (250 mg/L)	320*	320*	438	380
284	TDS (500 mg/L)	578*	578*	1130	747
	SO ₄ (250 mg/L)	226*	226*	320	192
286	TDS (500 mg/L)	650*	650*	1330	897
	SO ₄ (250 mg/L)	321.1*	321.1*	541	290
289	TDS (500 mg/L)	1140	890	NA	NA
	SO ₄ (250 mg/L)	430	304.6	NA	NA
290	TDS (500 mg/L)	1085*	1085*	1770	1235
	SO ₄ (250 mg/L)	468*	468*	745	398
291	TDS (500 mg/L)	1611*	1611*	2550	2155
	SO ₄ (250 mg/L)	798.9*	798.9*	1340	1093

* Asterisk indicates that the reported value is based on analysis of a single groundwater sample prior to ash placement.

Table 4.15. Annual Tonnage Deposited in the Universal Mine Pit

Year	Coal Ash	Coal Gasification Slag
1989	270,364	0
1990	254,806	0
1991	0	0
1992	320,000	0
1993	0	0
1994	0	0
1995	75,194	0
1996	114,740	0
1997	95,387	0
1998	117,742	23,301
1999	54,368	14,113
2000	152,571	0
2001	151,335	0
Total Amount	\$1,607,507	37,414

Table 4.16. Statistical Summary of Bulk Composition Data (mg/Kg) of Coal Ash Samples Placed in the Universal Mine Pit (Wabash River Station)

Parameter	No. Of Samples	Median	Average	Maximum	Minimum
Aluminum	23	7150	7373	13,200	42
Iron	25	15,700	15,935	27,500	86
Potassium	23	1200	1271	2700	ND***
Magnesium	23	752	714	1280	ND
Sulfate	25	455	526	2000	55
Sodium	25	230	276	570	ND
Boron	25	223	241	455	ND
Calcium	4	5720	5755	6680	4900
Zinc	25	123	129	227	2.1
Barium	25	76	71	220	ND
Manganese	25	80	86	200	1.2
Arsenic	25	57	64	143	ND
Vanadium	23	42	45	71	ND
Nickel	25	40	40	77	ND
Chloride	25	24	41	270	ND
Lead	25	35	36	59	ND
Chromium	25	22	24	42	ND
Copper	25	21	21	41	ND
Fluoride	25	8	18	177	ND
Molybdenum	25	3.7	4.6	12	1.9
Selenium	25	4.1	4.6	9.8	ND
Cadmium	25	1.0	1.1	3.2	ND
Silver	25	ND	0.18	1.6	ND
Mercury	25	0.130	0.113	0.30	ND
Total Organic Carbon	25	52,300	58,357	153,000	5
pH*	22	9.0	9.1	10.6	7.5
Potential Acidity**	25	0.3	2.6	21.6	ND
Neutralization Potential**	25	17.1	19.8	48.6	7.5
Net Neutralization Potential**	25	15.5	17.6	48	ND

* Unit for this parameter is s.u.

** Unit for these parameters is T/1000T.

*** Nondetect.

Table 4.17. Statistical Summary of Concentrations (mg/L) Measured in Coal Ash Leachates (Universal Mine Study – Wabash River Station Coal Ash)

Parameters	18-hr Leachate			30-day Leachate	
	No. of Samples	Average	Median	Average	Median
Aluminum	23	3.43	2.9	2.73	2.6
Boron	25	2.65	2.3	3.39	3.1
Sulfate	25	135	31	56.8	47
Chloride	25	0.7	ND	1.34	1.1
Iron	25	0.67	ND	0.34	ND*
Magnesium	23	0.5	ND	0.44	ND
Potassium	23	0.84	ND	0.78	ND
Sodium	25	1.45	ND	1.59	ND
Arsenic	25	0.285	0.26	0.301	0.31
Fluoride	25	0.225	0.23	0.285	0.29
Vanadium	23	0.18	0.21	0.23	0.27
Molybdenum	25	0.12	0.09	0.13	0.11
pH (standard units)	24	10.05	10.05	9.63	9.66
Barium	25	2	0.04	0.078	ND
Cadmium	25	0.0009	ND	0.0005	ND
Chromium	25	0.01	ND	0.01	0.011
Copper	25	0.008	ND	0.002	ND
Lead	25	0.002	ND	0.006	ND
Manganese	25	0.022	ND	0.004	ND
Nickel	25	0.002	ND	0.003	ND
Selenium	25	0.067	0.058	0.102	0.085
Zinc	25	0.043	0.02	0.018	ND
Silver	25	ND	ND	0.001	ND
Mercury	25	ND	ND	ND	ND
Sulfide	25	0.133	ND	0.077	ND

* Nondetect.

Table 4.18. Summary Statistics for Measured Concentrations (mg/L) in MW-8 Leachate Water

Parameter	No. Of Samples	Mean	Median	Minimum	Maximum
Alkalinity	17	276.2	220	120	530
Boron	17	44.2	46	ND*	56
Chloride	17	284.4	91	21	700
Sodium	17	190	120	ND	430
Sulfate	17	1847	1700	1400	3800
Total Organic Carbon	17	15.4	1.9	ND	110
Magnesium	17	6.6	6.9	ND	8.1
Molybdenum	17	1.7	1.6	0.96	2.3
Aluminum	15	0.38	0.44	ND	0.49
Arsenic	17	0.205	0.208	0.13	0.26
Sulfide	17	0.21	ND	3.4	ND
Barium	17	0.053	0.036	ND	0.35
Fluoride	17	0.007	ND	ND	0.12
Manganese	17	0.012	0.012	ND	0.023
Acidity	17	ND	ND	ND	0.0004
Cadmium	17	ND	ND	ND	ND
Chromium	17	ND	ND	ND	ND
Iron	17	ND	ND	ND	ND
Lead	17	ND	ND	ND	ND
Mercury	17	ND	ND	ND	ND
Selenium	17	ND	ND	ND	0.006
Silver	17	ND	ND	ND	ND
Zinc	17	ND	ND	ND	ND
pH (standard units)	17	8.96	9.2	9.5	6.9

* Nondetect.

Table 4.19. List of Water Quality Parameters Monitored at the Universal Site

Acidity	Mercury
Alkalinity	Molybdenum
Aluminum	Nickel
Arsenic	ORP
Boron	Potassium
Cadmium	Selenium
Calcium	Silver
Chloride	Sodium
Chromium	Specific conductivity
Copper	Sulfate
Fluoride	Sulfide
Hardness	Temperature
Iron	TDS
Lead	TSS
Magnesium	TOC
Manganese	Zinc
	pH

Table 4.20: Cheswick Power Station (Harwick Mine Complex, PA) Chemical and Leachate Analysis of Fly Ash

Parameter	Total Constituent Analyses, mg/kg	EP Toxicity Leachate, mg/L	ASTM Water Leachate, mg/L
Aluminum	109,000	0.03	0.06
Antimony	<30	0.041	<0.005
Arsenic	<10	<0.005	0.033
Barium	158	0.07	0.08
Cadmium	<10	0.01	0.01
Chloride	–	0.8	8.0
Chromium (total)	148	<0.03	<0.03
Chromium (+6)	247	<0.03	<0.03
Copper	325	1.43	0.03
Iron	60,100	0.04	0.02
Lead	<10	0.01	0.05
Magnesium	5680	3.60	1.99
Mercury	<0.3	<0.0004	<0.0004
Molybdenum	19.7	0.006	0.038
Nickel	493	7.35	0.01
Selenium	2	0.023	0.020
Silver	19.7	0.01	0.01
Sulfate	2070	51	41
Sulfide by N ₂ generations	<1	3	<3
Tin	10	<0.004	<0.004
Zinc	2436	0.11	0.01
pH	7.05	4.77*	7.21
Conductivity (µmhos/cm)	–	309	160
Total Suspended Solids	–	7.2	8.0
Chemical Oxygen Demand	–	110	<3
Corrosivity	No	–	–
Corrosive	7.0	–	–
pH	<1	<2	<2
Cyanide	>200	–	–
Flashpoint (°F)	0.6	–	–
Nitrate (as M)	–	<1.0	7.3
Oil & Grease`	<5	<0.005	<0.005
Phenols	–	237	134
Total Residue	2.54	–	–
% Volatile Residue	–	230	128
Total Dissolved Solids	6200	48.0	<1.0
Total Organic Carbon	<20	–	–
Total Organic Halogens	280	–	–
Total 1 Nitrogen	–	–	–

* 12 mls acetic acid added.

**Table 4.21. Chemical Analysis Results for the Monarch Mine
(Harwick Mine Complex, PA) Effluent, November 1973**

pH	6.0
Specific Conductance	4000 $\mu\text{mhos}/\text{cm}^2$
Total Acidity	<1 mg/L
Alkalinity	300 mg/L
Turbidity	200 JTU
Chloride	170 mg/L
Sulfate	1560 mg/L
Hardness	1400 mg/L
Calcium	184 mg/L
Magnesium	235 mg/L
Total Dissolved Solids	3854 mg/L
Total Iron	200 mg/L
Sodium	810 mg/L

Table 4.22. Pre-and Post-Grouting Mean Concentrations of Mine Drainage Constituents for Wells Located in Discharge Plumes at the Clinton County, PA Site (Schuck et. al. 1996)

Monitoring Well FF62

Condition	Lab, pH	TDS, mg/L	SO ₄ , mg/L	Acid, mg/L	Fe		Al, mg/L	Mn, mg/L	Cd, µg/L	Cu, µg/L	Cr, µg/L	Ca, µg/L
					Tot, mg/L	Fe,+3, mg/L						
Pregrout	2.3	7970	3477	4088	876	737	256	39.2	83.6	806	221	58.1
Postgrout	2.5	5780	3110	2879	527	373	173	24.7	29.3	813	168	61.4

Monitoring Well S80D

Condition	Lab, pH	TDS, mg/L	SO ₄ , mg/L	Acid, mg/L	Fe,	Fe,	Al, mg/L	Mn, mg/L	Cd, µg/L	Cu, µg/L	Cr, µg/L	Ca, µg/L
					Tot, mg/L	+3, mg/L						
Pregrout	2.4	9951	3500	5096	937	749	394	45.5	108.5	154.2	394	66.4
Postgrout	2.7	7222	3483	3230	530	254	282	32.4	24.9	771	232	58.7

Monitoring Well W70

Condition	Lab, pH	TDS, mg/L	SO ₄ , mg/L	Acid, mg/L	Fe,	Fe,	Al, mg/L	Mn, mg/L	Cd, µg/L	Cu, µg/L	Cr, µg/L	Ca, µg/L
					Tot, mg/L	+3, mg/L						
Pregrout	2.6	9689	3695	4611	735	606	397	49.4	60.2	985	221	81.9
Postgrout	3.0	4795	3327	2348	268	185	180	21.8	17.3	635	156	56.3

Monitoring Well V36

Condition	Lab, pH	TDS, mg/L	SO ₄ , mg/L	Acid, mg/L	Fe,	Fe,	Al, mg/L	Mn, mg/L	Cd, µg/L	Cu, µg/L	Cr, µg/L	Ca, µg/L
					Tot, mg/L	+3, mg/L						
Pregrout	2.7	6777	3568	3624	570	396	316	63.7	57.3	526	104	83.1
Postgrout	3.1	5330	3111	2351	380	263	212	33.8	21.6	689	149	71.9

Table 4.23. Water Quality Data for Before Placement of Ash in the Big Gorilla Pit and after Placement of Ash in the Big Gorilla. Silverbrook Outflow Water Quality Data are also shown.

Analyte	Big Gorilla 1993	Big Gorilla 1998	Silverbrook 1998
pH	3.2	11.4	4.3
Conductivity, $\mu\text{mhos/cm}$	210	1273	443
Alkalinity, mg/L CaCO_3	<1	172	6.2
Acidity mg/L	37.8	<1	78
Turbidity, NTU	0.24	1.6	<1
Dissolved Solids, mg/L	98	311	268
Chloride, mg/L	6	3	9
Sulfate, mg/L	59	369	193
Aluminum, mg/L	4.2	1.32	6.81
Arsenic, mg/L	<0.001	<0.004	<0.004
Barium, mg/L	<0.1	0.0013	0.028
Cadmium, mg/L	0.0007	<0.01	<0.01
Calcium, mg/L	3.95	195	27
Chromium, mg/L	<0.001	0.051	<0.05
Copper, mg/L	0.02	0.012	0.023
Iron, mg/L	0.4	<0.02	17.7
Lead, mg/L	0.001	<0.001	<0.001
Magnesium, mg/L	4.5	0.193	9.6
Manganese, mg/L	0.72	<0.01	1.64
Mercury, mg/L	<0.0002	0.0014	<0.001
Nickel, mg/L	<0.04	<0.05	0.091
Nitrogen, mg/L	<0.5	0.09	<0.04
Potassium, mg/L)	1	10.4	1.84
Selenium, mg/L	<0.002	<0.007	<0.007
Silica, mg/L	N/A	19.11	15.96
Sodium, mg/L	1.15	9.87	9.85
Zinc, mg/L	0.2	<0.01	0.295

NA Not available.

Table 4.24. Average of Values Above Detection from Analyses Performed on NEPCO Fly and Bottom Ash for Module 25 Regulations from 1992–2002 (maximum allowable leachate concentration values provided for comparison)

Parameter Name	Bulk Analysis, mg/Kg		Leachate Analysis, mg/L*		Max. Leachate Conc., mg/L
	Fly Ash	Bottom Ash	Fly Ash	Bottom Ash	
pH, s.u.	10.67	11.47			
Sulfate	4901	3073	440	169	2500
Aluminum	26,440	13,993	2.53	2.77	5
Antimony	2.24	LF	0.05	0.01	0.15
Arsenic	17.79	3.55	LF	LF	1.25
Barium	211	116	0.16	0.21	50
Boron	44.07	8.49	0.30	0.37	31.5
Cadmium	1.22	0.68	LD	LD	0.13
Chromium	27.25	11.49	0.09	LF	2.5
Cobalt	3.62	1.11	LF	LF	
Copper	28.13	8.18	0.09	0.07	32.5
Iron	8713	4399	0.14	0.24	7.5
Lead	29.99	12.17	LF	LF	1.25
Manganese	161.04	2672.65	0.20	0.21	1.25
Mercury	0.64	LF	LF	LF	0.05
Molybdenum	9.28	3.29	0.13	0.02	4.38
Nickel	11.75	5.08	LF	LF	2.5
Potassium	6737	4400	9.88	7.75	
Selenium	14.38	5.47	0.17	0.03	1
Silver	3.56	LF	LF	LF	
Zinc	18.96	9.78	0.18	0.26	125
Nitrate-N			1.27	2.92	
Chloride			15.75	54.67	2500
Sodium			10.73	6.14	
Total Organic Carbon			LF	2.11	
Acid Neutralizing Potential	847,400	1,258,625			

LD = always less than detection.

LF = less than 5 measurements above detection (of 22).

*Leachate analyses were performed by TCLP before January 1995, and SPLP thereafter.

Table 4.25. Average of Concentrations of Groundwater Samples from Wells Sampled Four times a Year on or Near the NEPCO Property

Location	Well 1 5/89–2/91	Well 2 12/90–11/02	Well3 9/96–11/02	Well 4 12/01–11/02	Well 9 9/96–11/02
No. of samples	8	58	40	4	39
pH	4.30	5.04	4.30	3.75	3.7
Conductivity	89.13	68.88	249.28	528.25	455
Alkalinity	1.20	8.16	3.30		
Hot Acidity	37.56	47.79	47.49	114.95	147.1
Iron, T	3.77	0.60	1.74	18.22	0.6
Manganese, T	0.33	0.43	0.85	8.69	1.7
Sulfate, T	33.50	30.81	82.82	191.75	182.7
Chloride	0.95	1.89	2.26		6.6
Sodium	0.25	4.66	9.38	4.47	16
Dis. Solids	75.25	69.40	161.54	438	368.6
Sus. Solids	2.86	7.15	9.48	5.33	29.2
Nitrate N		1.49	0.07		0.2
Silica, T		7.38	10.26	28.61	57.7
Aluminum, T	0.66	0.45	2.65	10.09	18
Barium, T		0.03	0.03		
Calcium, T		6.26	20.65	32.03	18.8
Magnesium, T	2.70	3.79	5.84	14.23	8.9
Zinc, T	0.12	0.33	0.31	0.68	0.4
Potassium, T		1.68	1.84	2.34	3.9

All concentrations reported in mg/L except for conductivity ($\mu\text{S}/\text{cm}$), turbidity (ntu), and pH (s.u.).
T indicates total concentrations in the water samples.

Table 4.26. Average Values for Chemical and Physical Characteristics of the Acid Mine Drainage from the Red Oak Mine Site

Parameter	Concentration, mg/L
Aluminum	6
Calcium	63
Iron	200
Magnesium	42
Manganese	7
pH	4.3
Acidity (as CaCO ₃)	475

Table 4.27. XRF Data for the Materials Used in Grout Admixtures

Constituents, %	FBC By-Product	Class F Flyash	FGD Material	Quick Lime
K ₂ O	1.6	2.4	0.2	None
MgO	2.3	1.1	0.67	1.5
Fe ₂ O ₃	6.2	5.6	0.56	0.35
Al ₂ O ₃	11.5	28.5	2.6	0.5
SiO ₃	12.3	0.59	47.1	0.04
SiO ₂	24.7	52.4	3.4	1.5
CaO	24.8	1.6	35.4	97.5

APPENDIX B

FIGURES

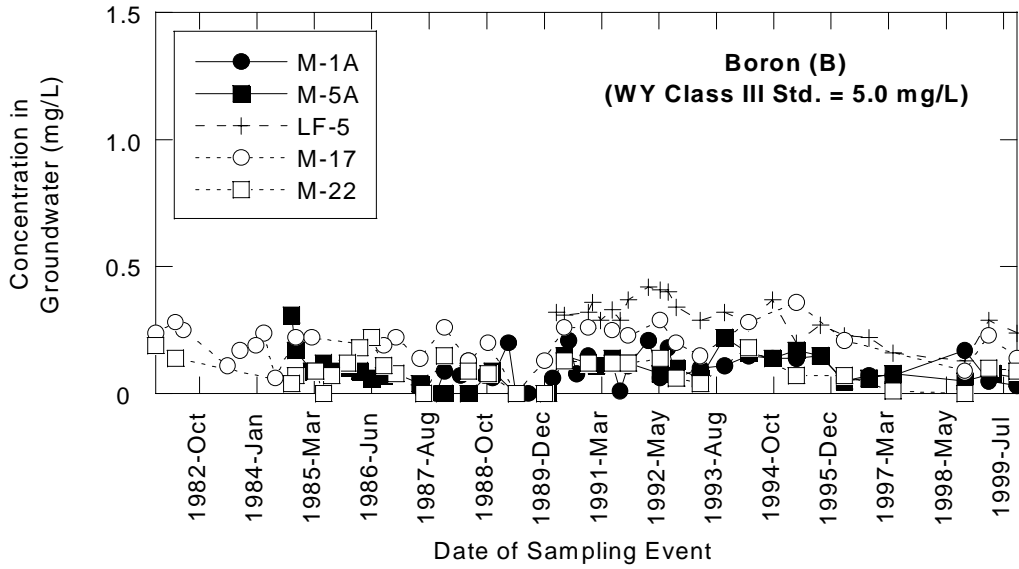


Figure 4.1. Historical trends of boron (B) in groundwater from selected wells at the Wyodak Mine.

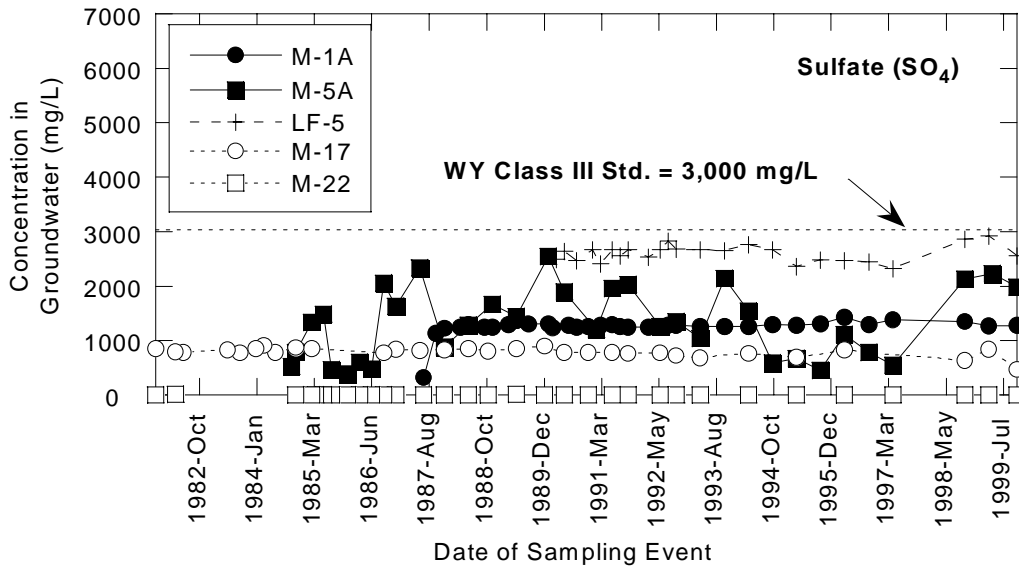


Figure 4.2. Historical trends of sulfate (SO₄) in groundwater from selected wells at Wyodak Mine.

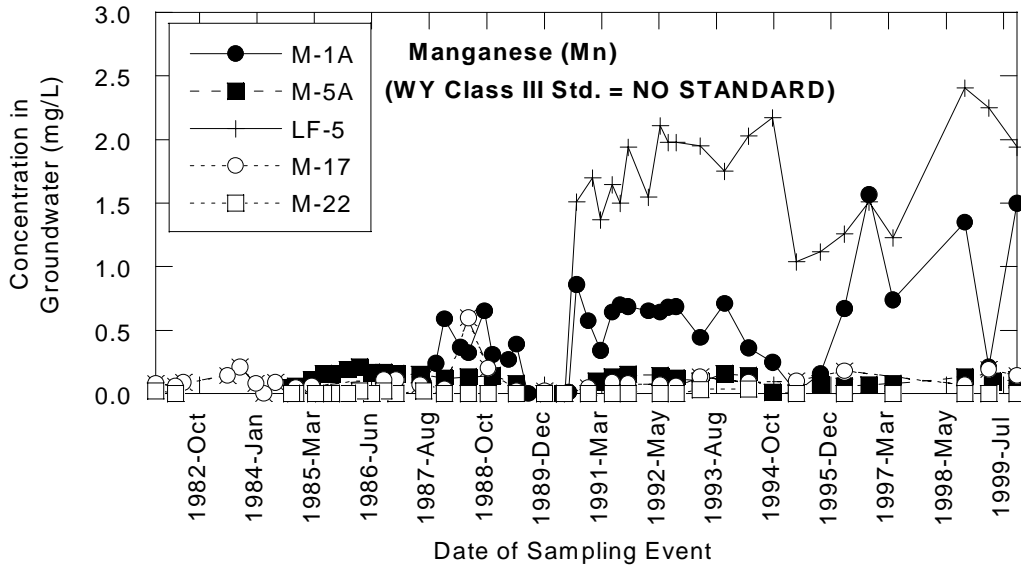


Figure 4.3. Historical trends of manganese (Mn) in groundwater from selected wells at Wyodak Mine.

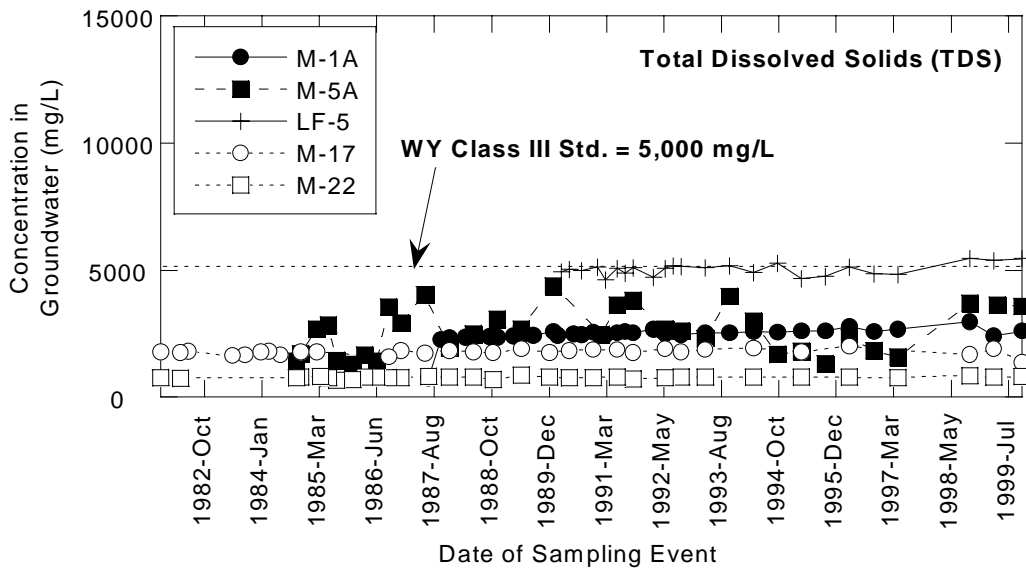


Figure 4.4. Historical trends of total dissolved solids (TDS) in groundwater from selected wells at Wyodak Mine.

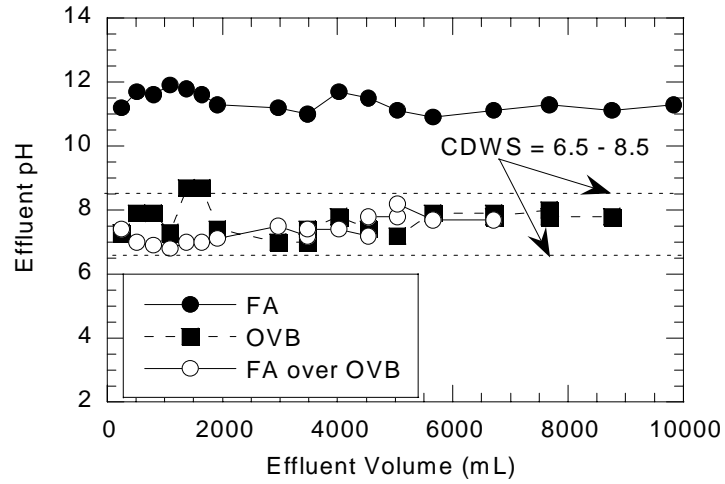


Figure 4.5

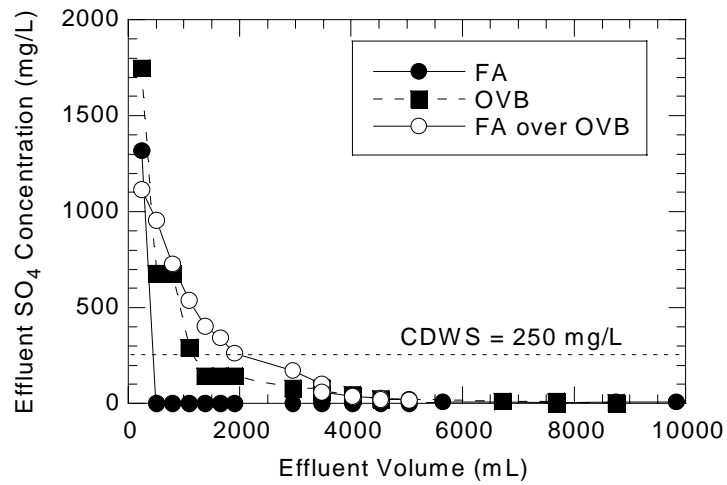


Figure 4.6. Effluent levels of pH SO₄, Al, and Ba measured in Trapper Mine (CO) column leaching study (FA = fly ash, OVB = overburden, CDWS = Colorado drinking water standard).

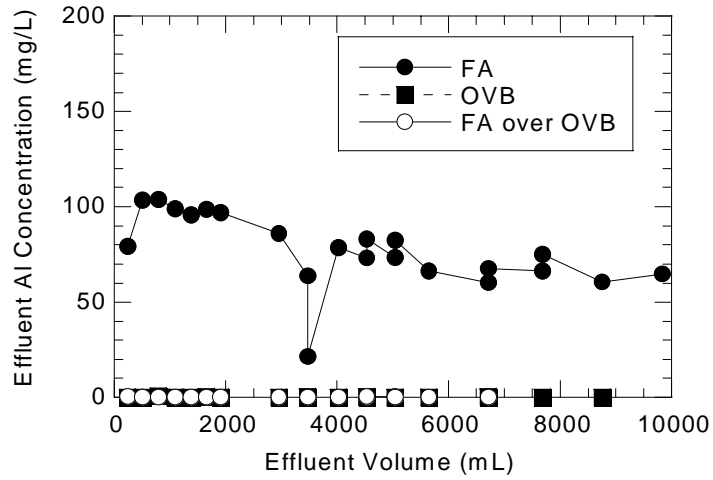


Figure 4.7. Effluent levels of B, Cr, Mn, and V measured in Trapper Mine (CO) column leaching study (FA = fly ash, OVB = overburden, GWS = Colorado groundwater standard).

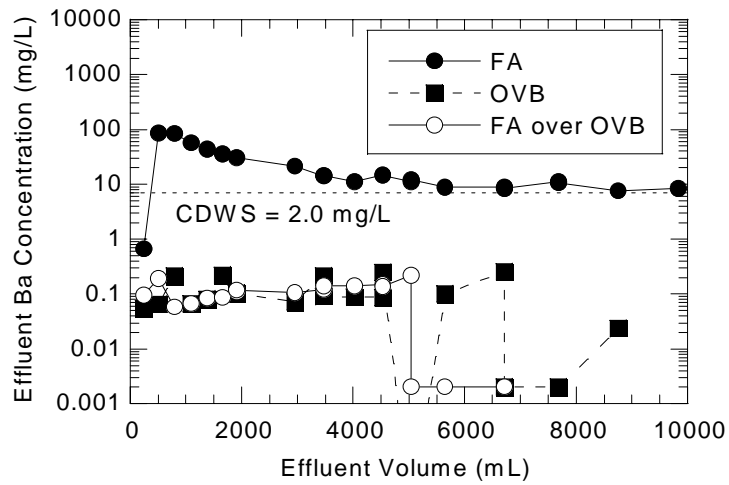


Figure 4.8

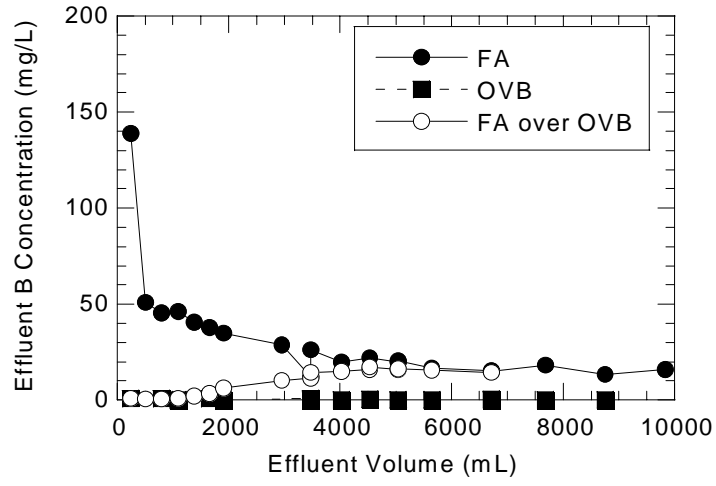


Figure 4.9

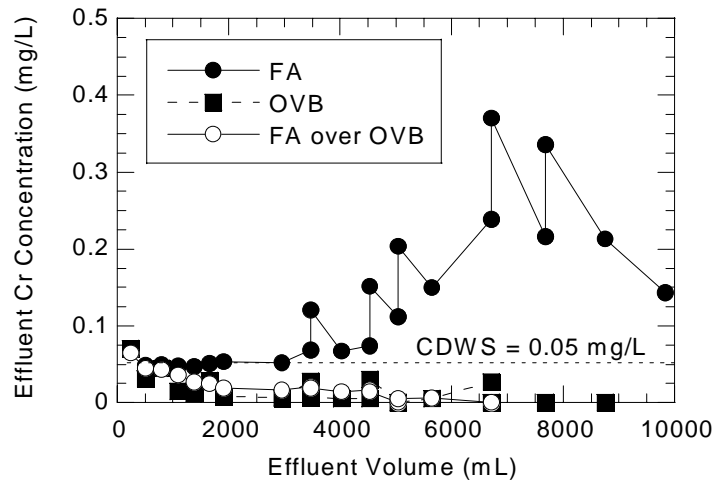


Figure 4.10

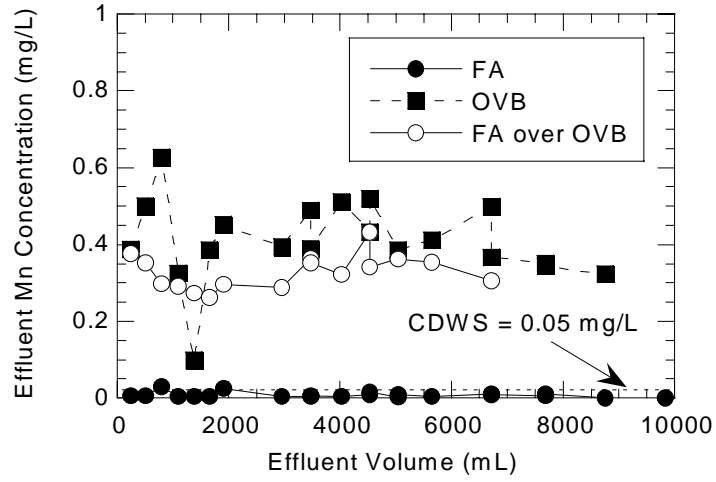


Figure 4.11

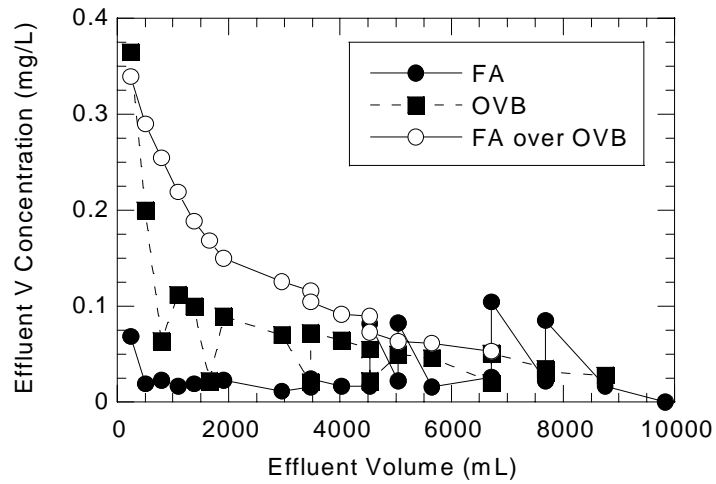


Figure 4.12

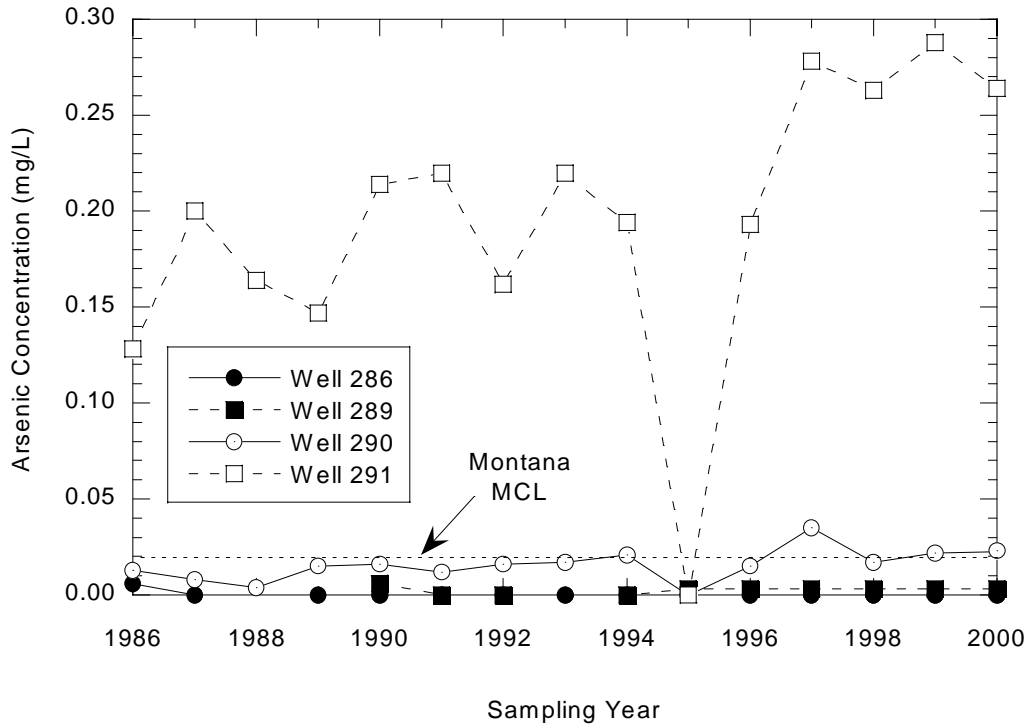


Figure 4.13 Historic arsenic (As) concentrations in groundwater monitoring wells at Savage Mine (MT).

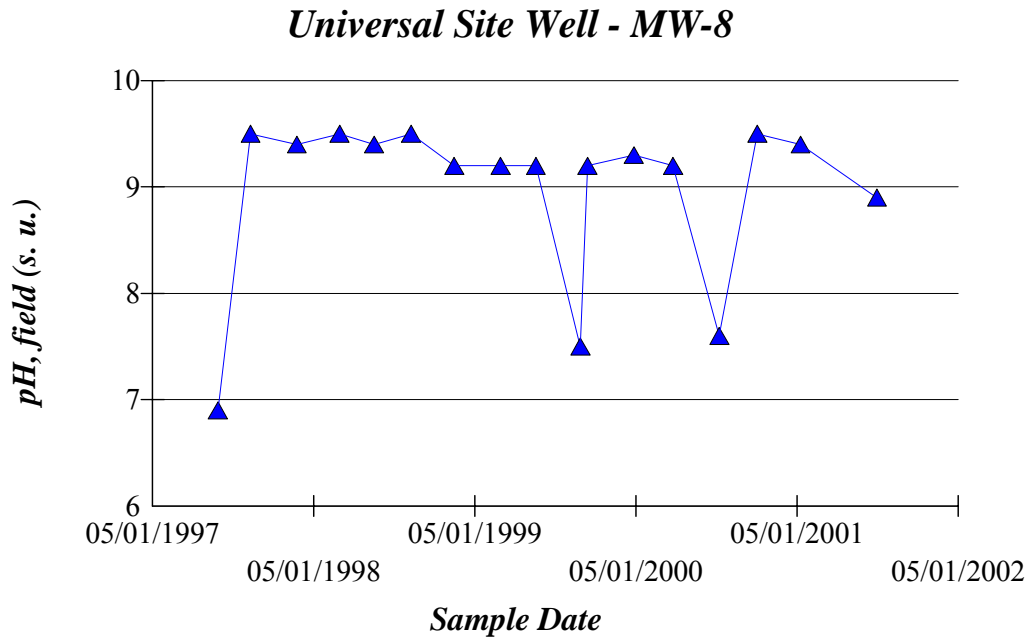


Figure 4.14 Time-series plot of measured pH in field leachate samples.

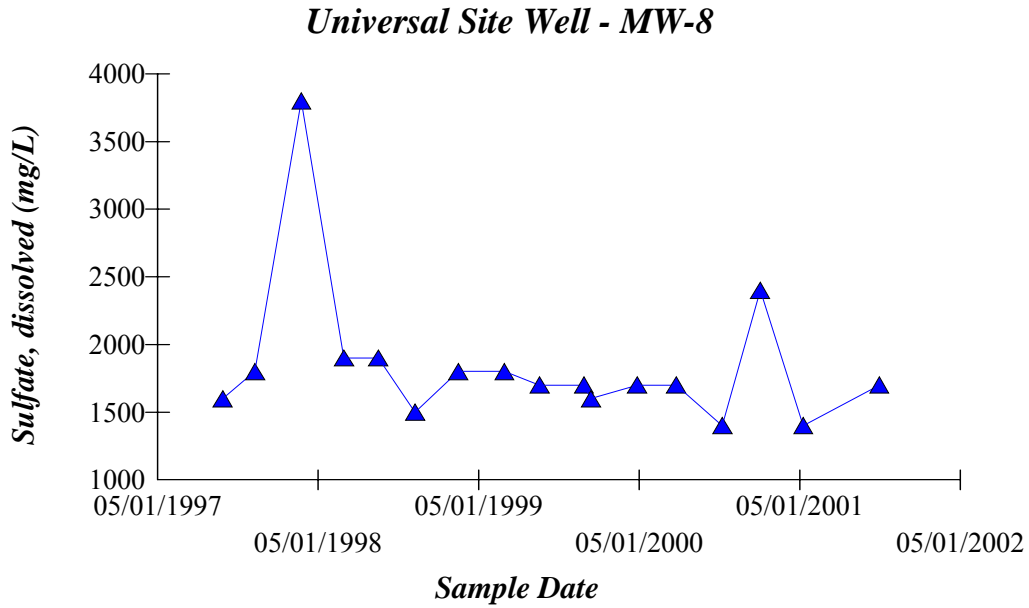


Figure 4.15. Time-series plot of measured sulfate concentrations in field leachate samples.

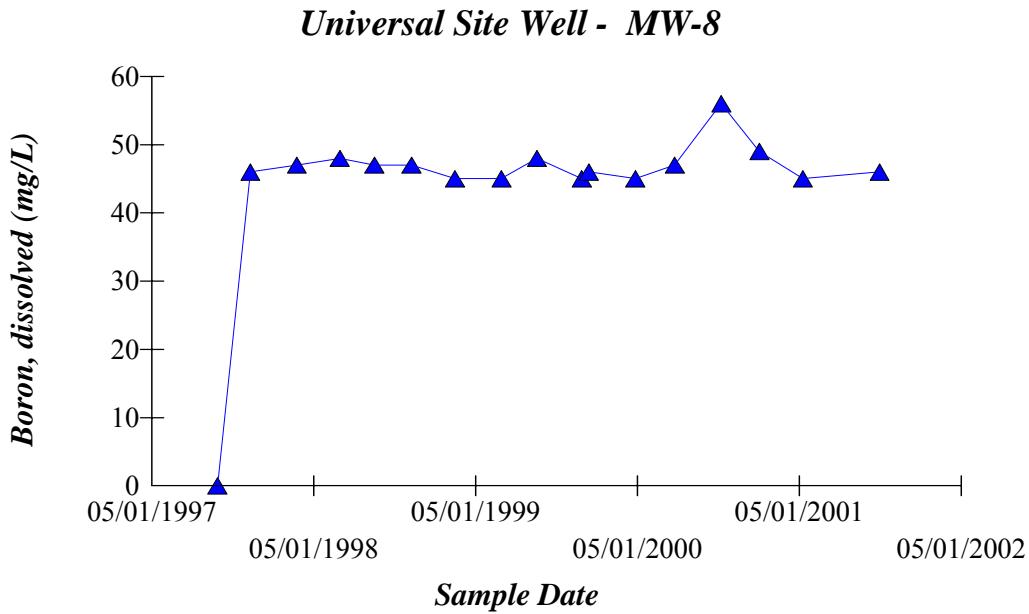


Figure 4.16. Time-series plot of measured boron concentrations in field leachate samples.

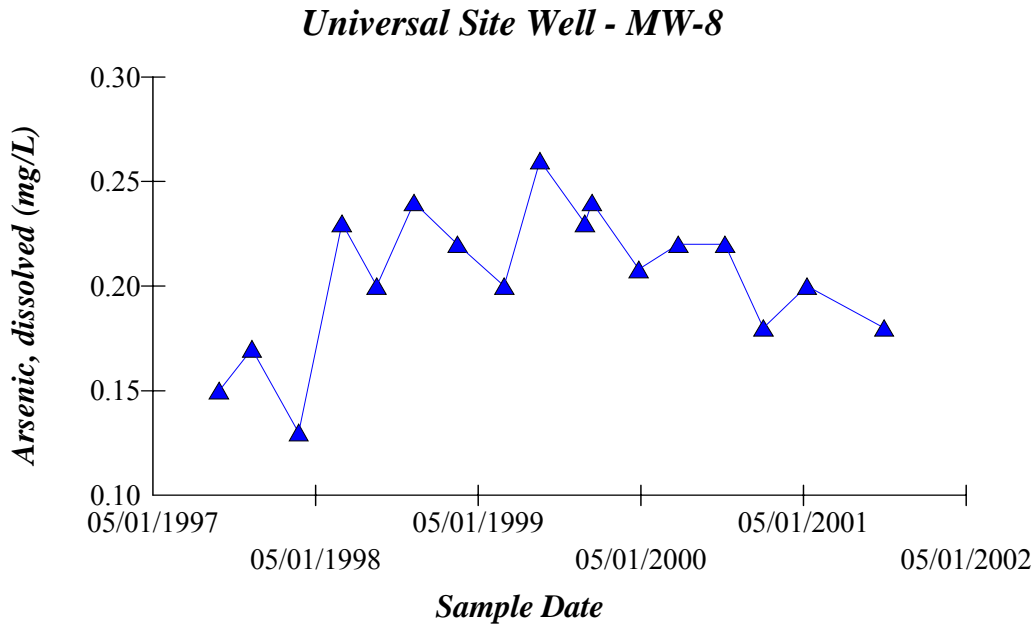


Figure 4.17. Time-series plot of arsenic concentrations in field leachate samples.

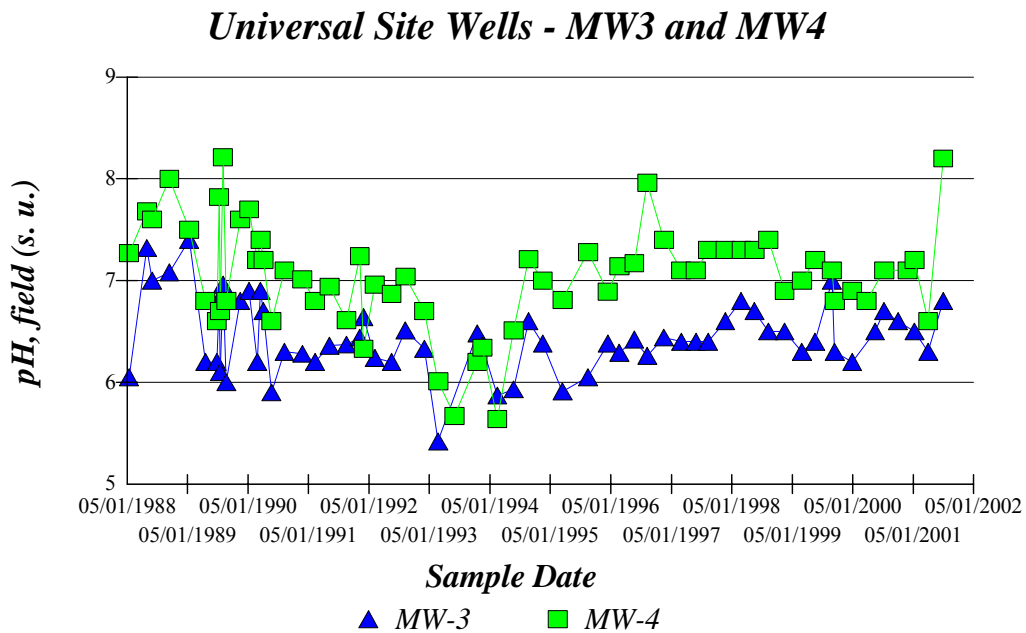


Figure 4.18. Time-series plot showing pH measured in groundwater samples.

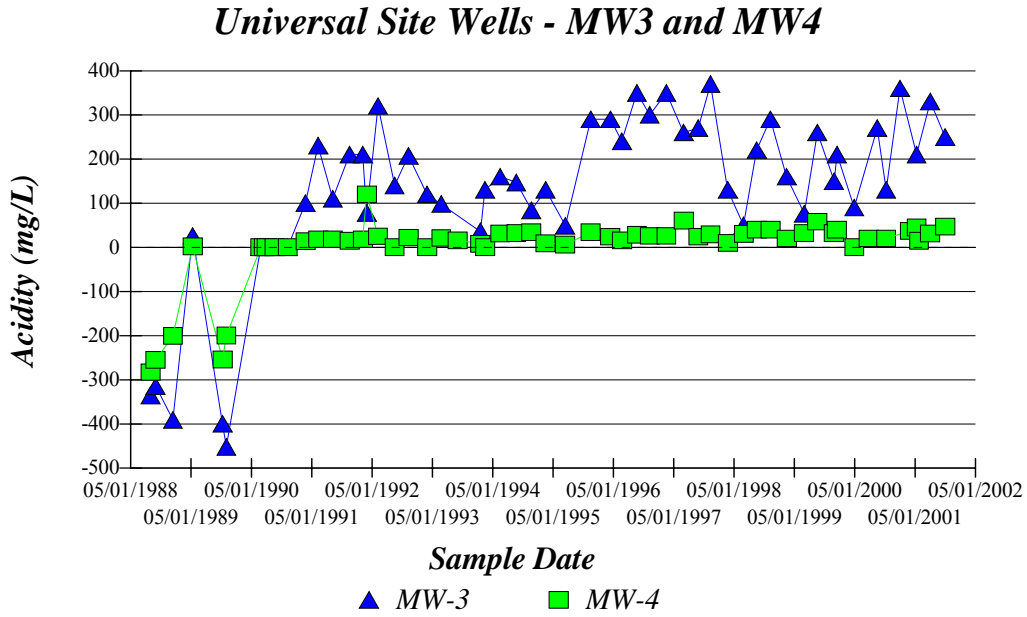


Figure 4.19. Time-series plot of acidity measured in groundwater samples.

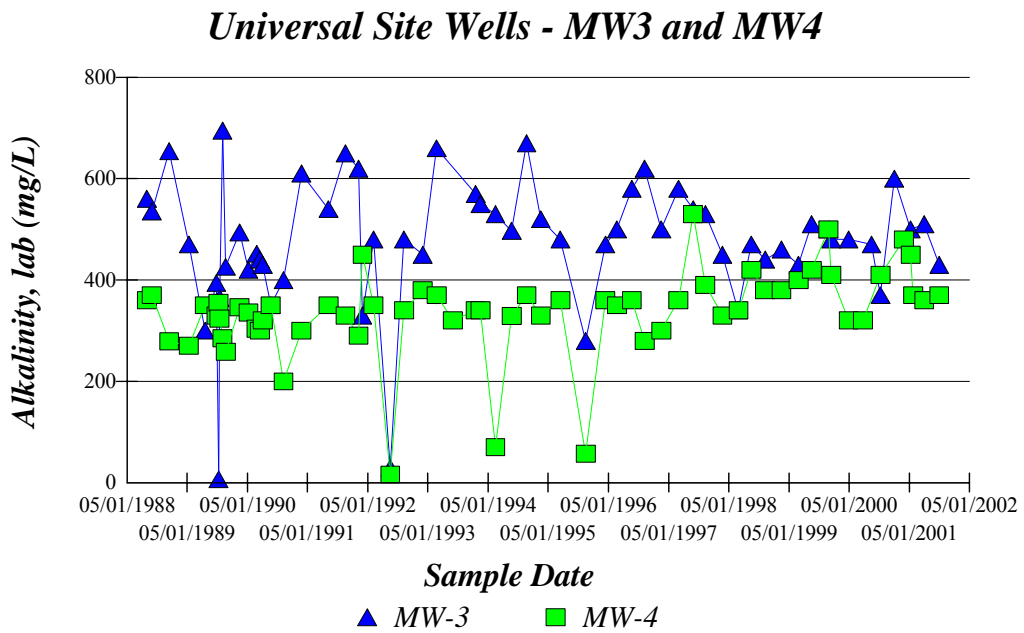


Figure 4.20. Time-series plot of alkalinity measured in groundwater samples.

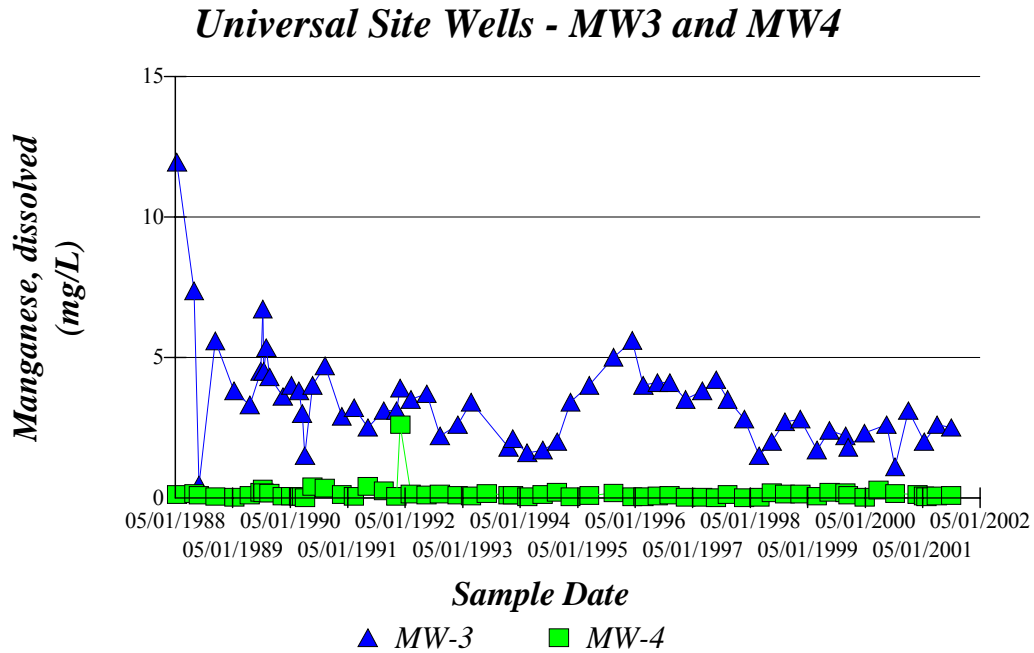


Figure 4.21. Time-series plot of manganese measured in groundwater Samples

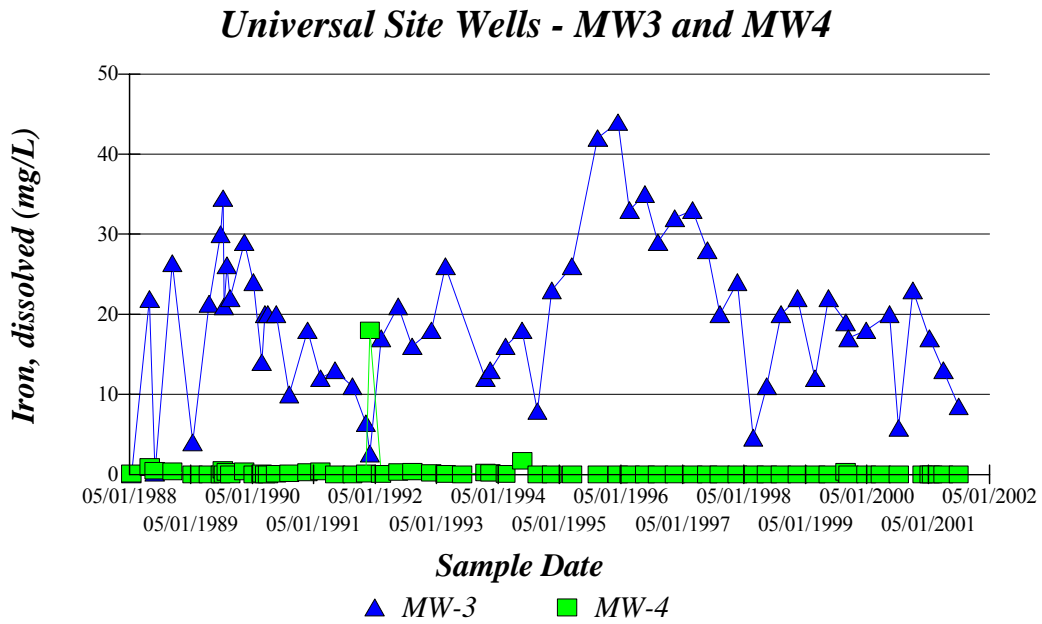


Figure 4.22. Time-series plot of iron measured in groundwater samples.

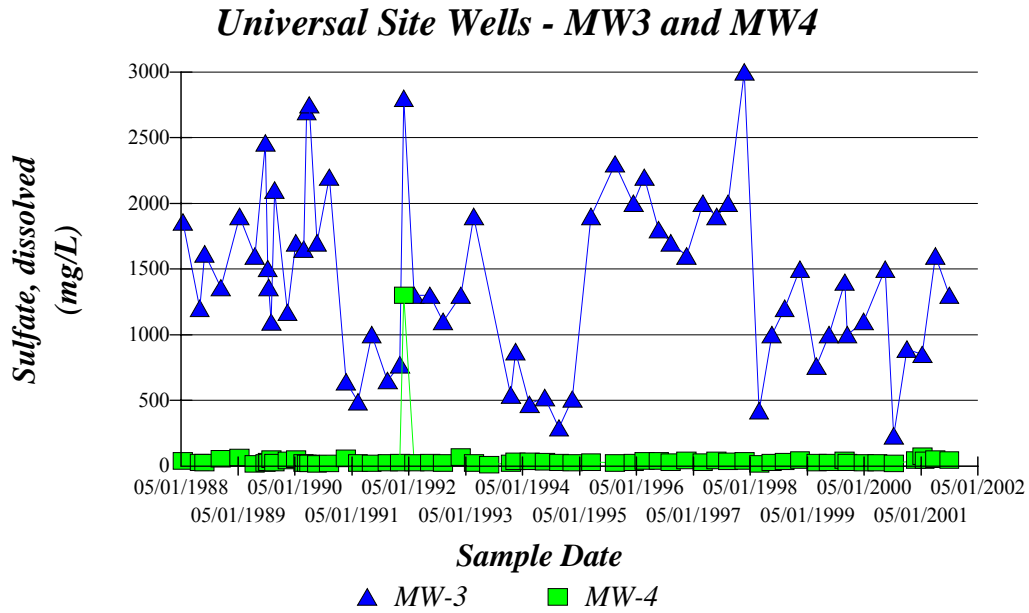


Figure 4.23. Time-series plot of sulfate measured in groundwater samples.

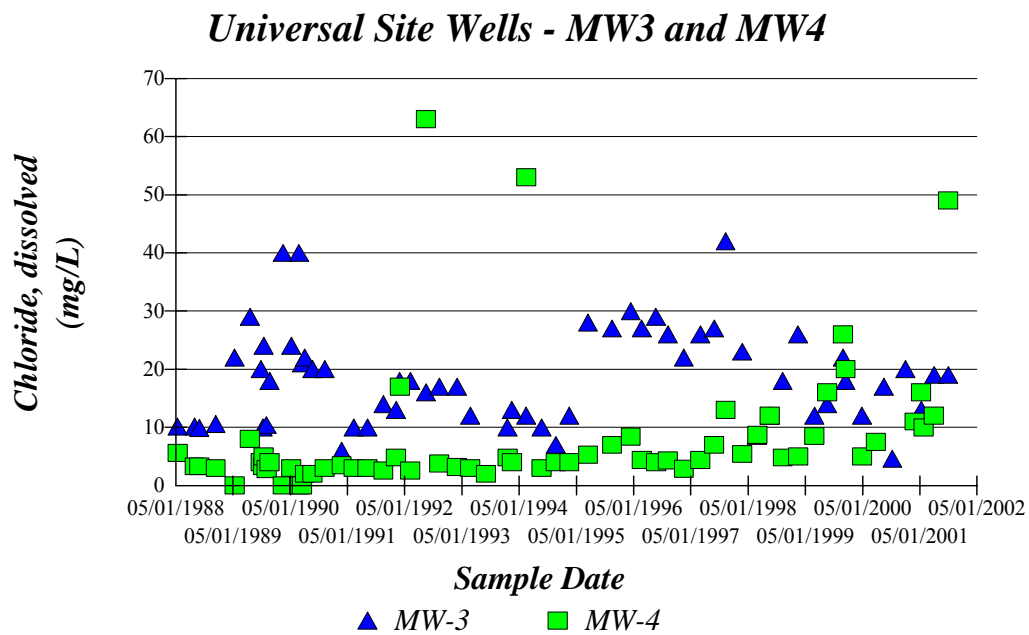


Figure 4.24. Time-series plot of chloride measured in groundwater samples.

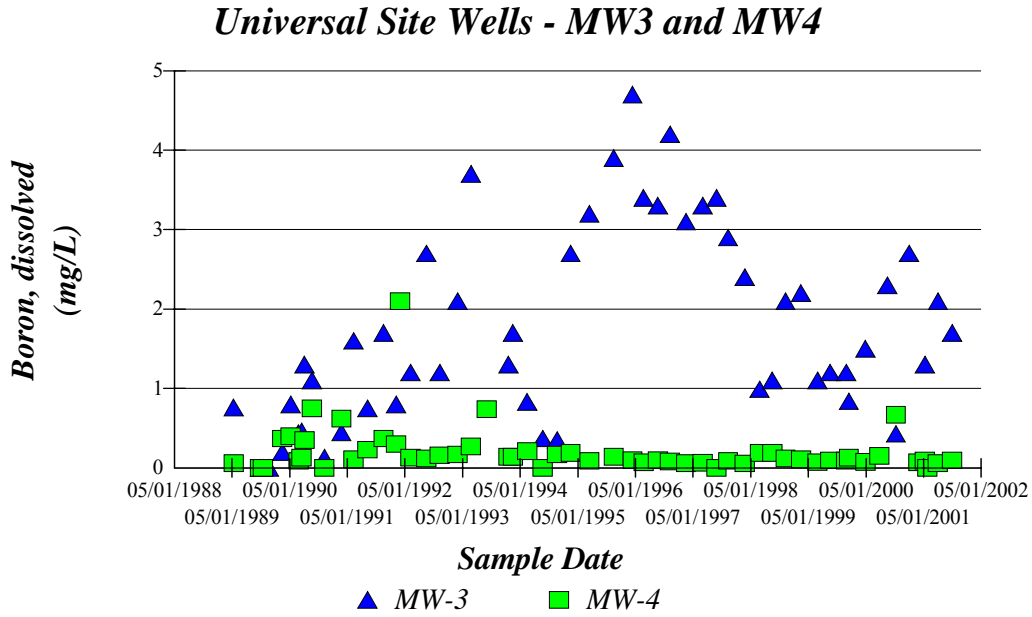


Figure 4.25. Time-series plot of boron measured in groundwater samples.

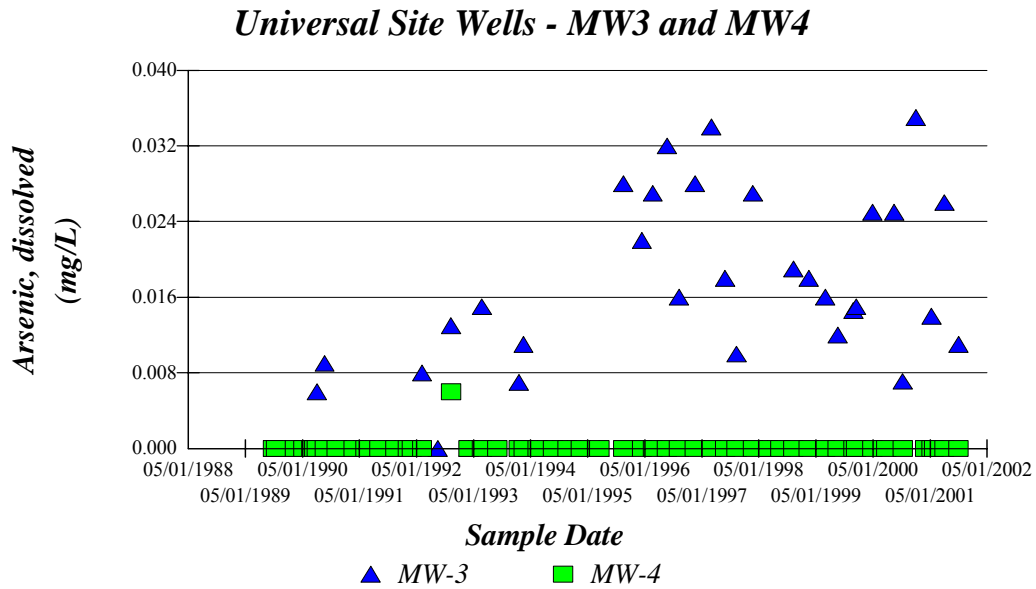


Figure 4.26. Time-series plot of arsenic measured in groundwater samples.

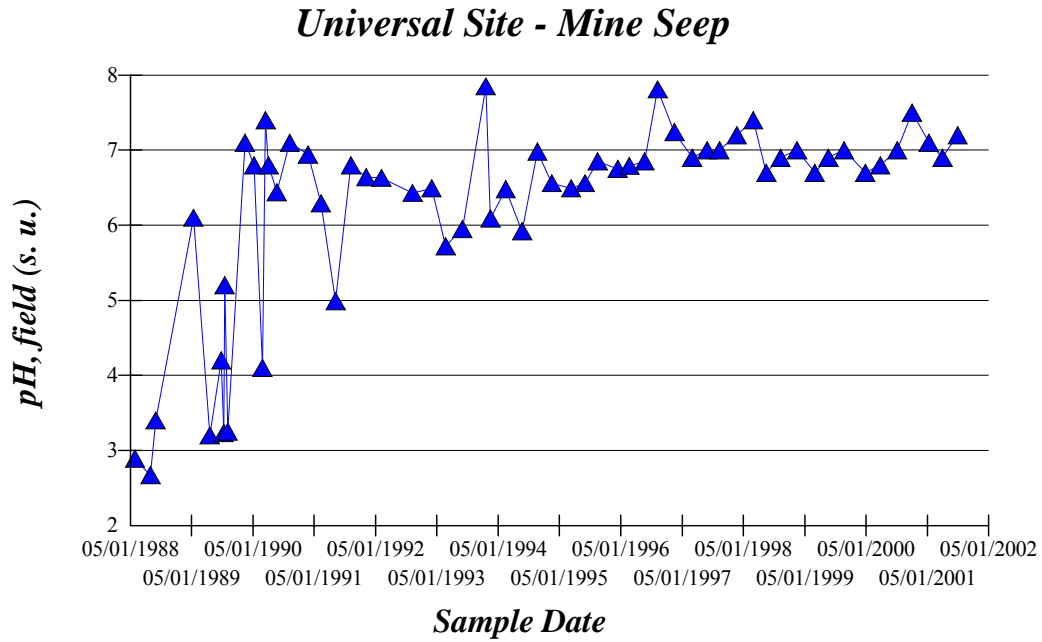


Figure 4.27. Time-series plot of pH measured in mine-seep water samples.

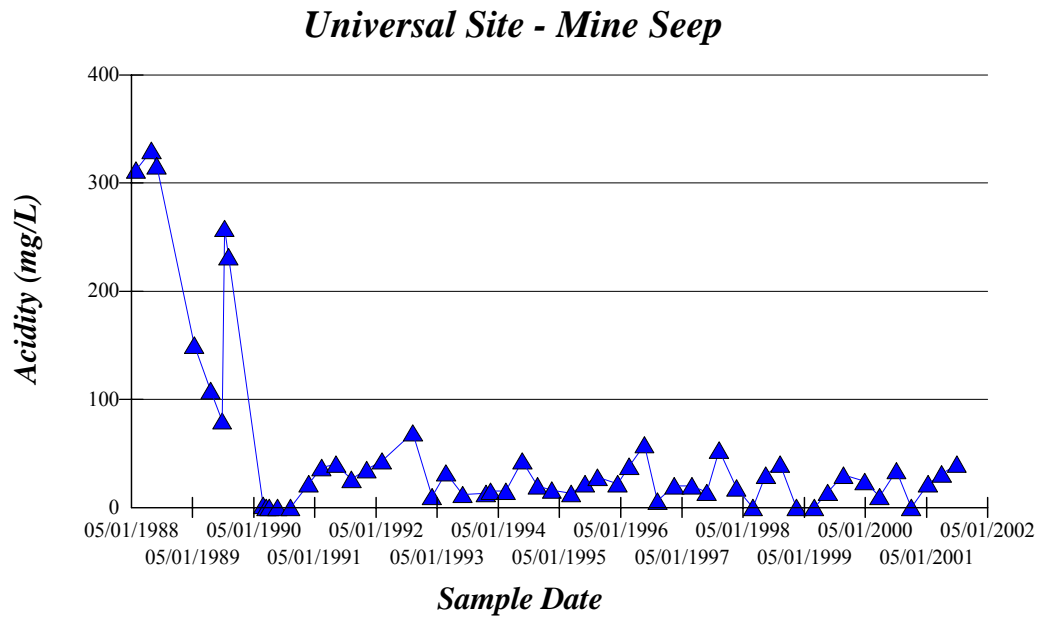


Figure 4.28. Time-series plot of acidity measured in mine-seep water samples.

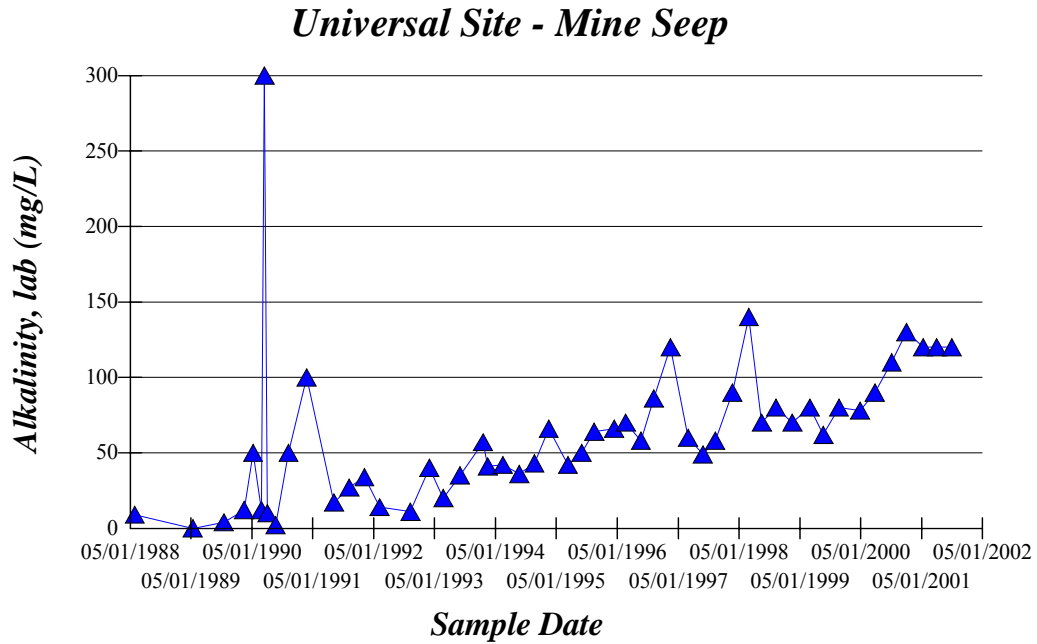


Figure 4.29. Time-series plot of alkalinity measured in mine-seep water samples.

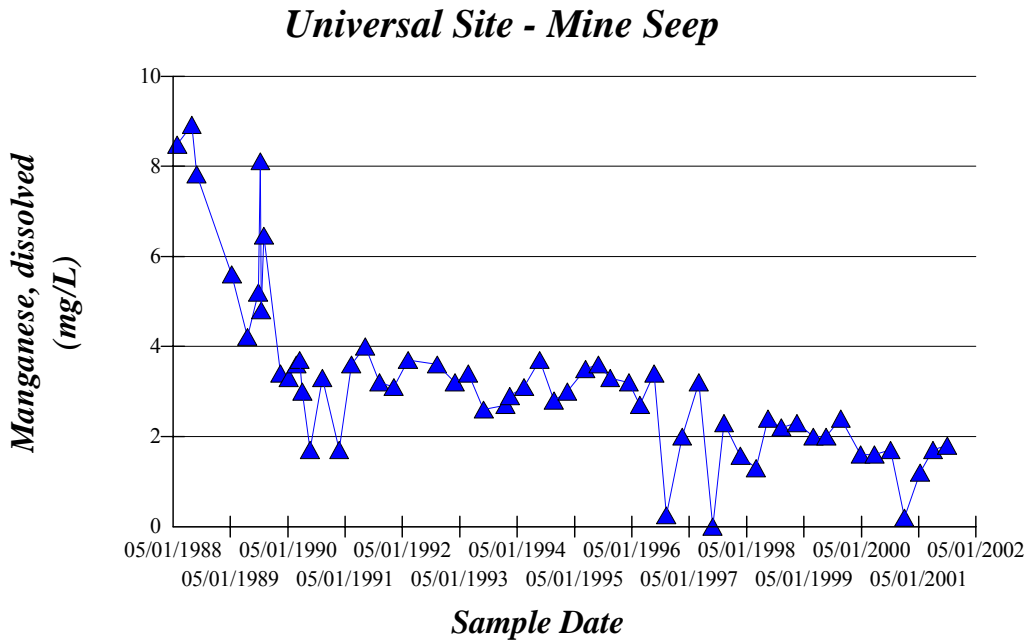


Figure 4.30. Time-series plot of manganese measured in mine-seep water samples.

Universal Site - Mine Seep

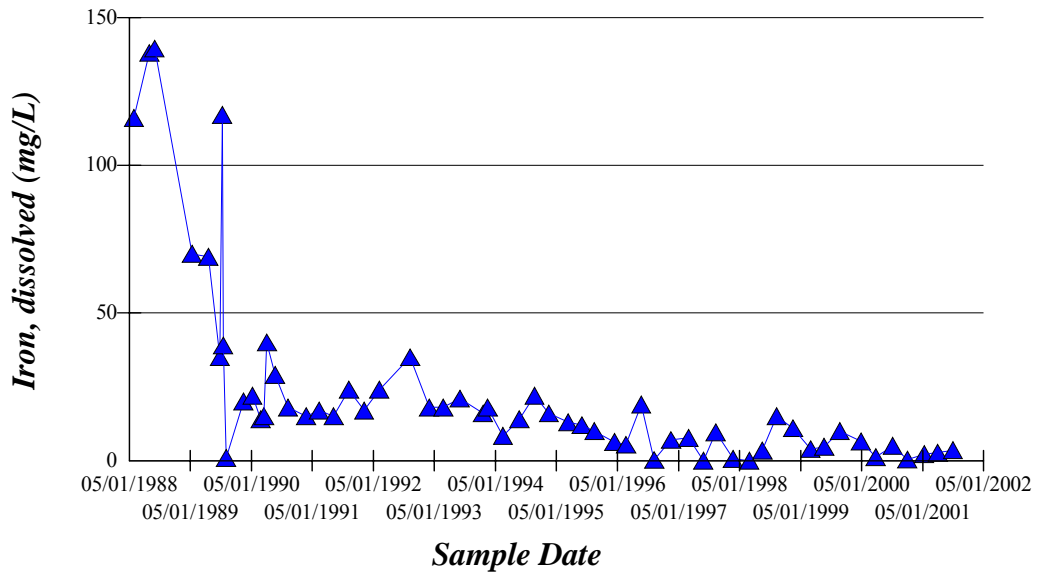


Figure 4.31. Time-series plot of iron measured in mine-seep water samples.

Universal Site - Mine Seep

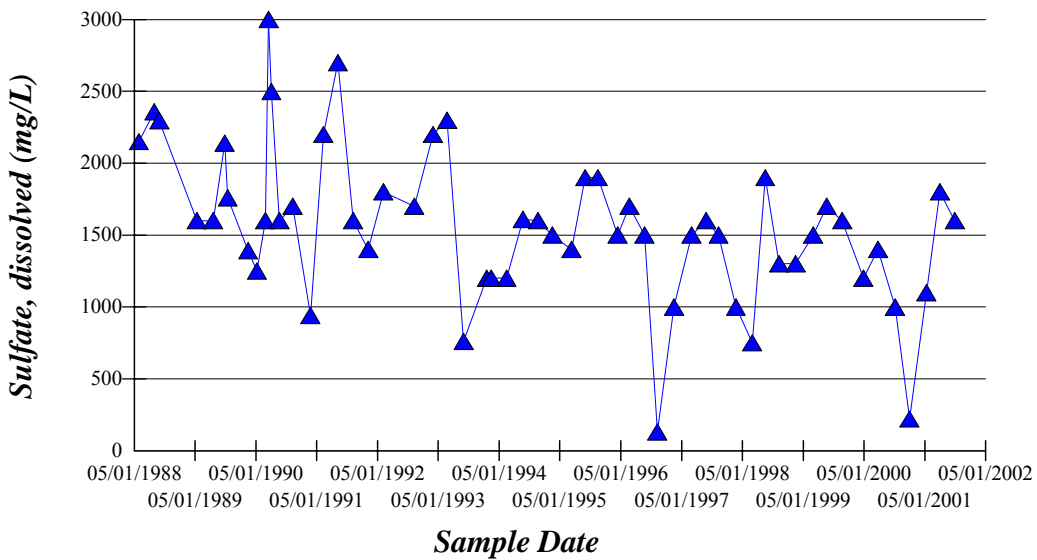


Figure 4.32. Time-series plot of sulfate measured in mine-seep water samples.

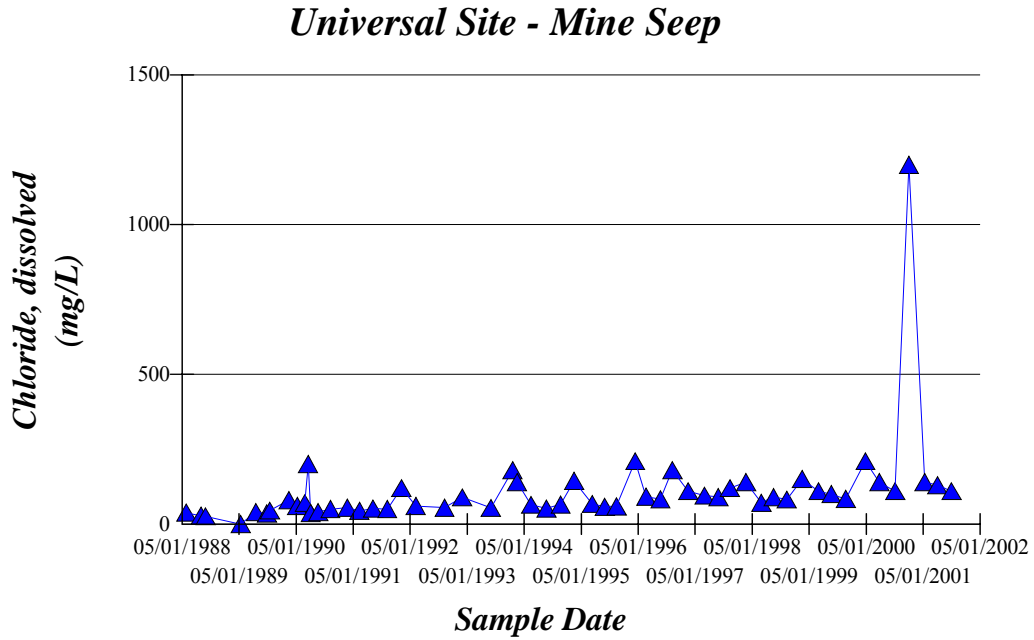


Figure 4.33. Time-series plot of chloride measured in mine-seep water samples.

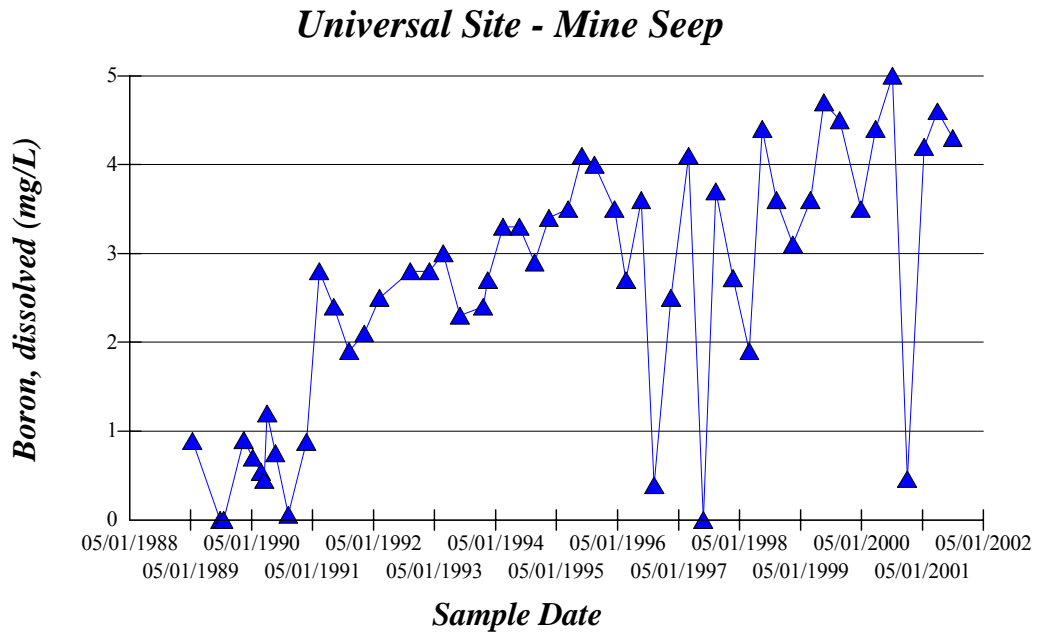


Figure 4.34. Time-series plot of boron measured in mine-seep water samples.

Figure 4.35. Alkalinity and pH values versus time for the mine discharge for 1-year postinjection.

Figure 4.36. Concentration of iron, aluminum, manganese, and pH versus time for the mine discharge for 1-year postinjection.

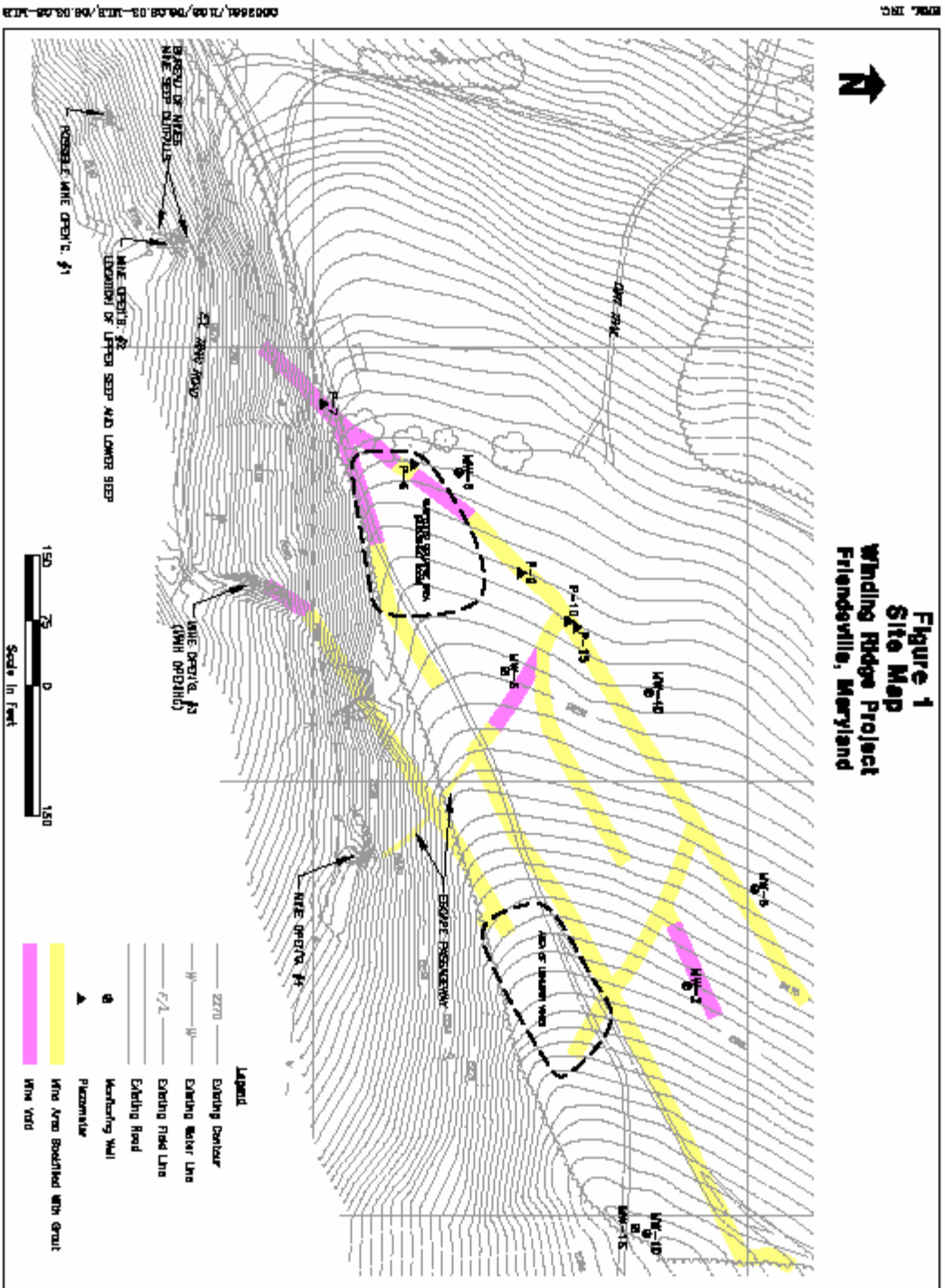


Figure 4.37

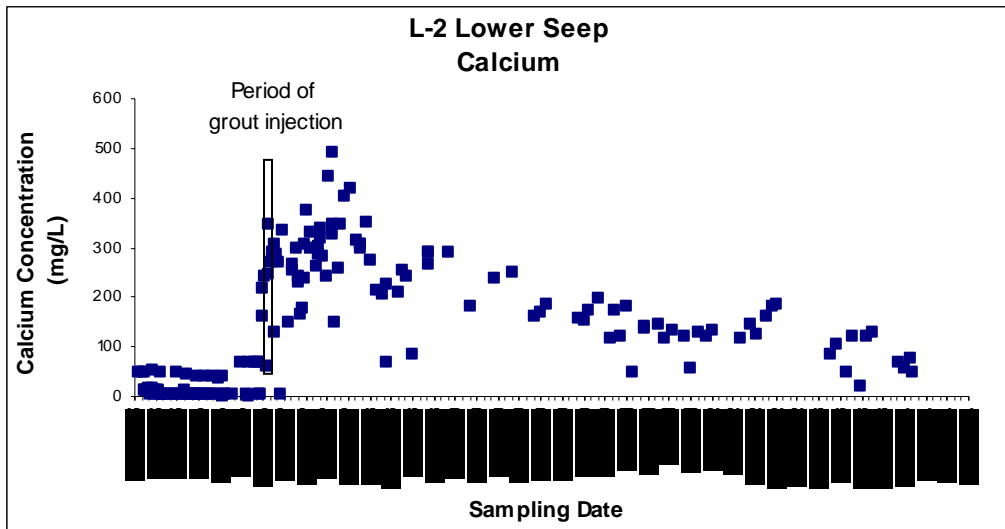


Figure 4.38. Time-series plot for measured calcium concentrations.

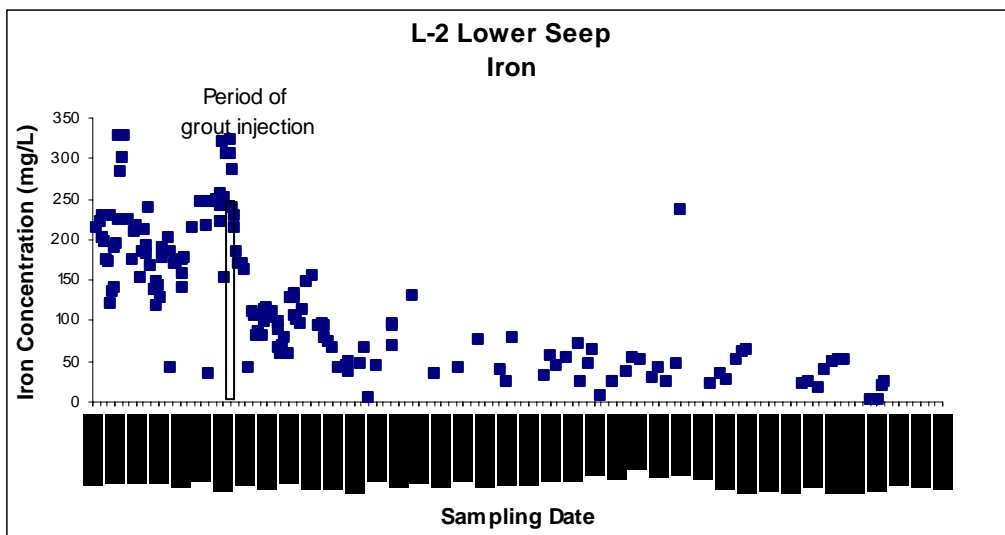


Figure 4.39

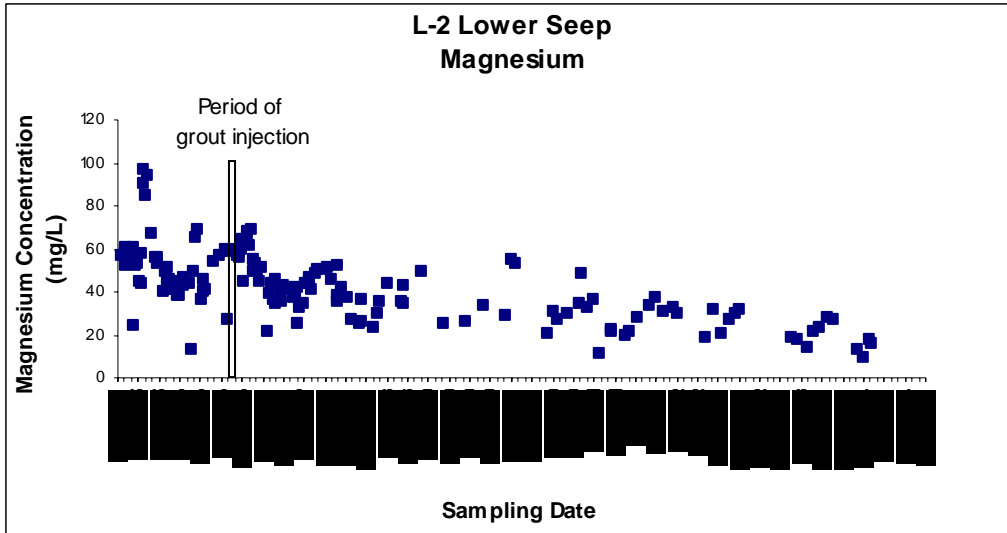


Figure 4.40

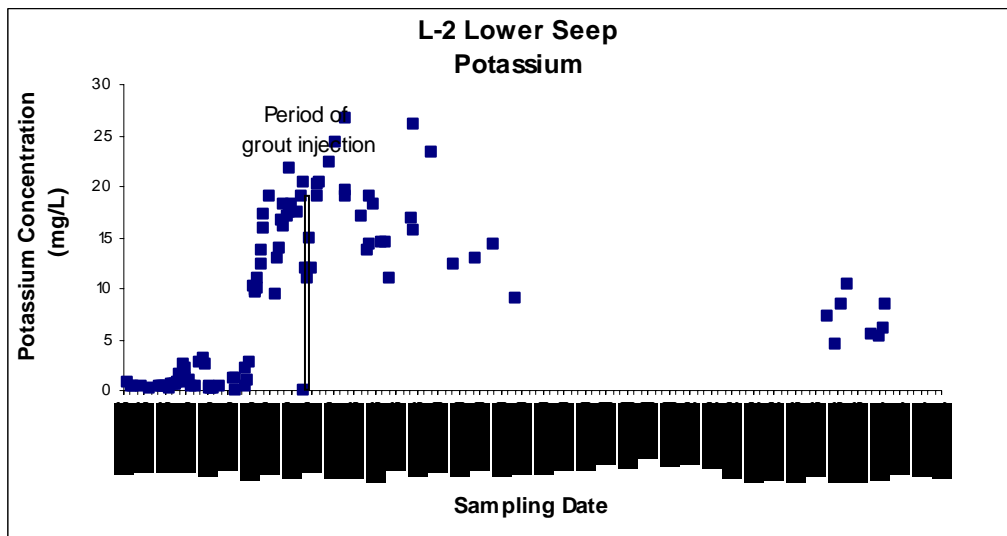


Figure 4.41

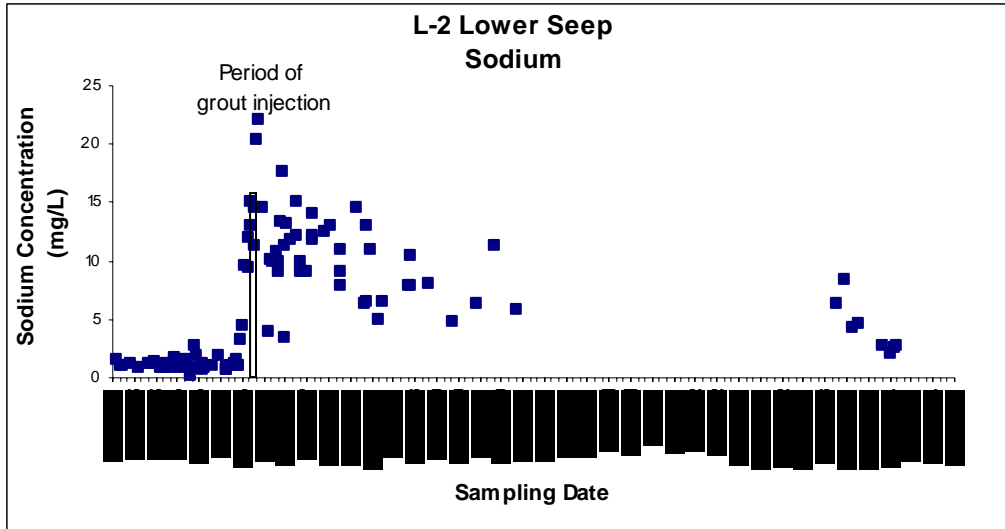


Figure 4.42

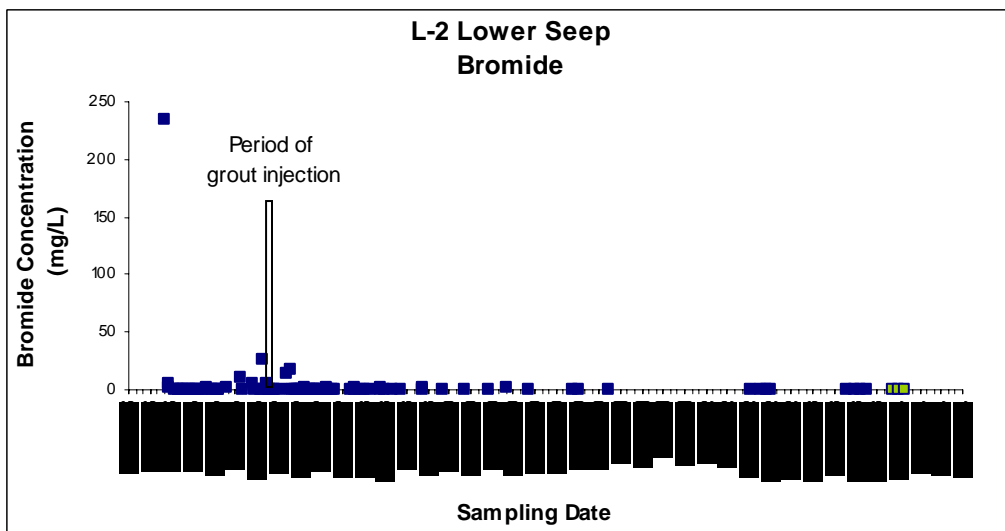


Figure 4.43

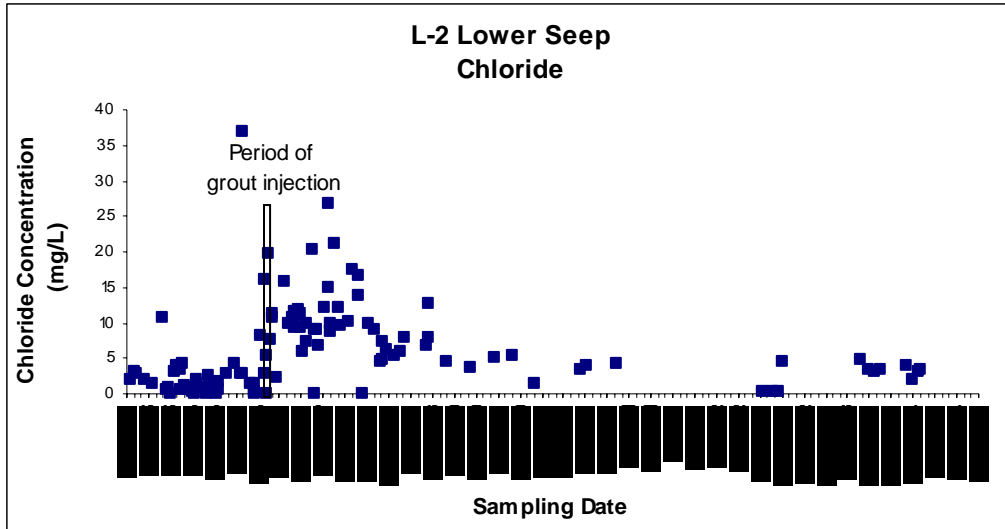


Figure 4.44

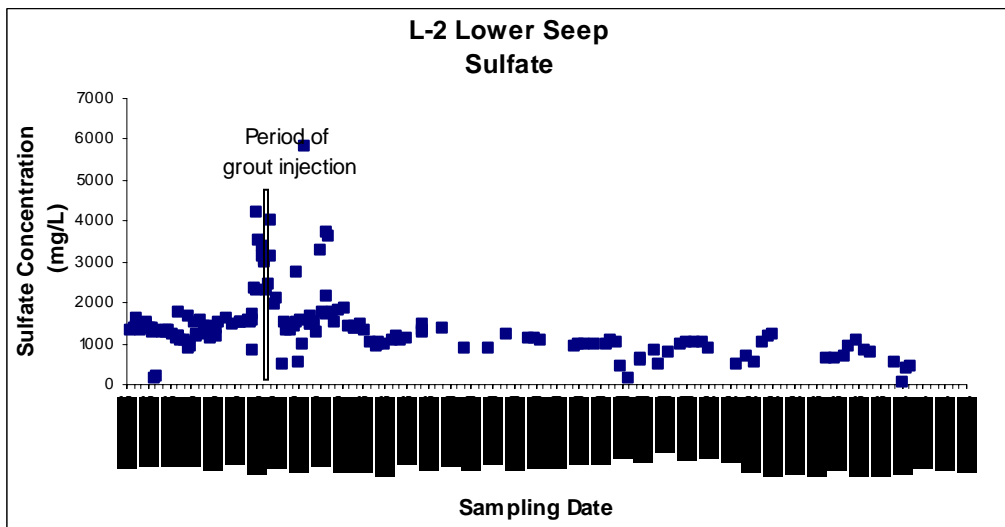


Figure 4.45

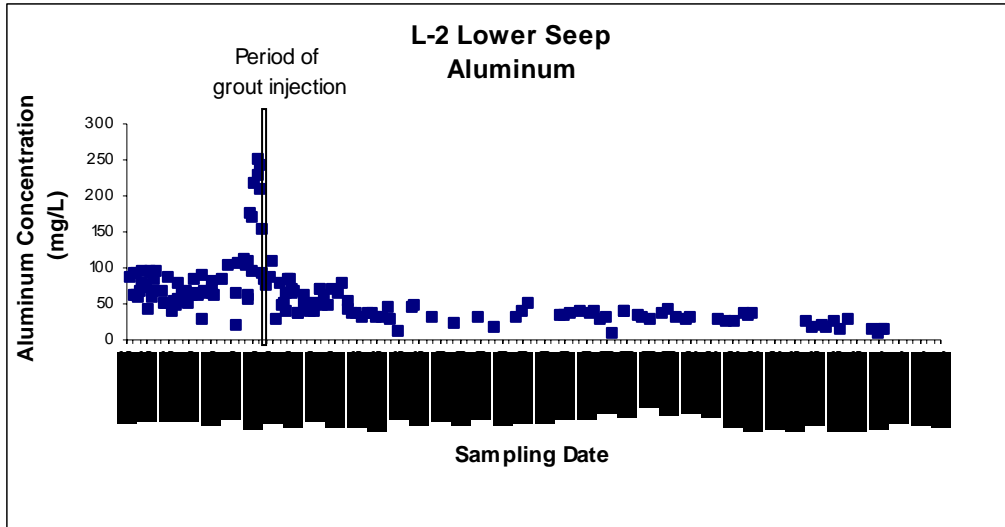


Figure 4.46

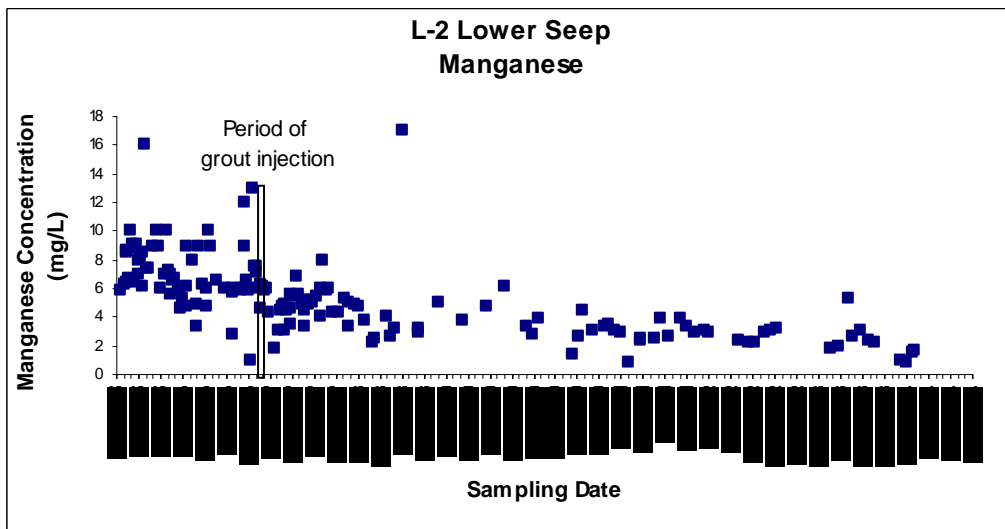


Figure 4.47

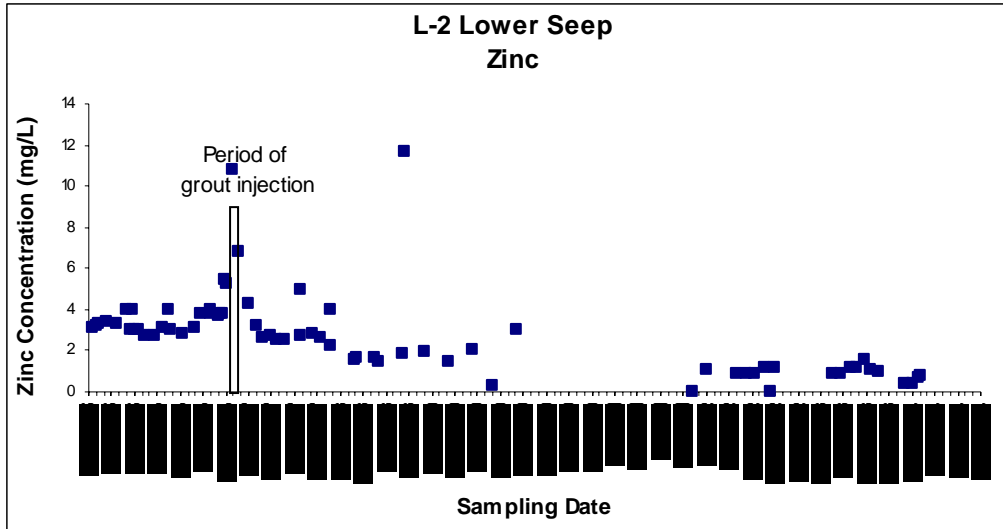


Figure 4.48

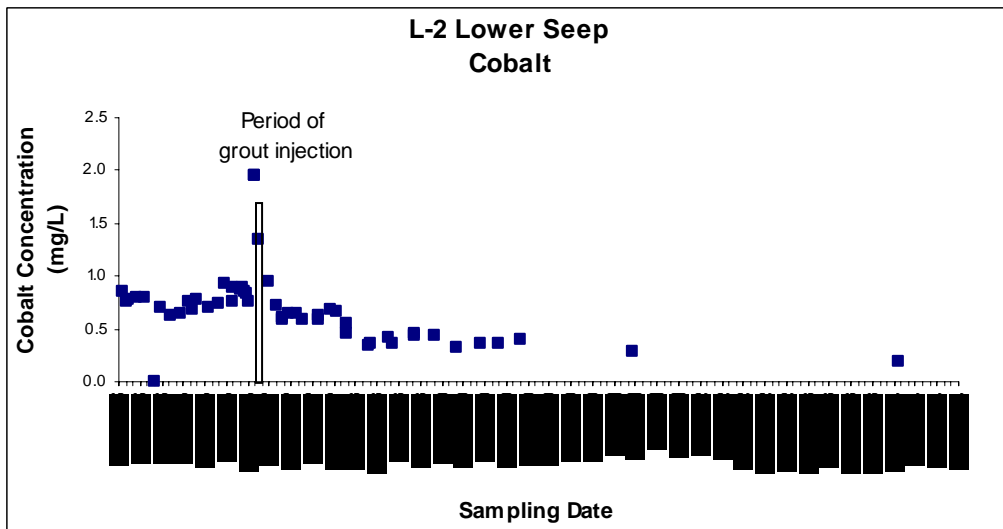


Figure 4.49

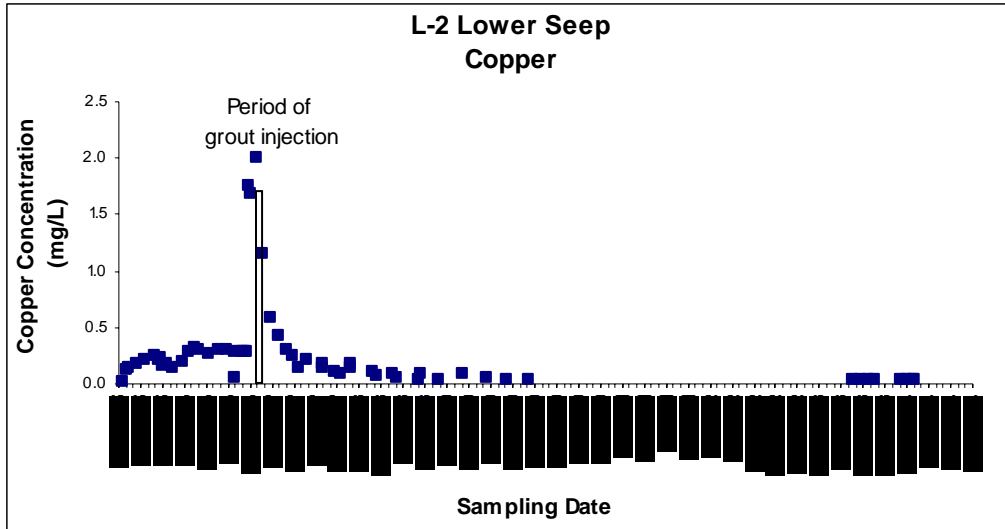


Figure 4.50

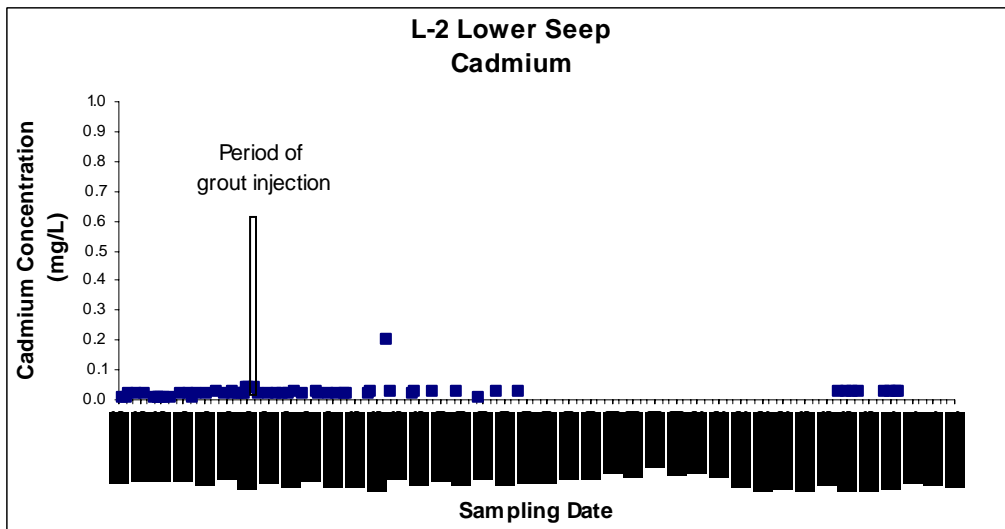


Figure 4.51

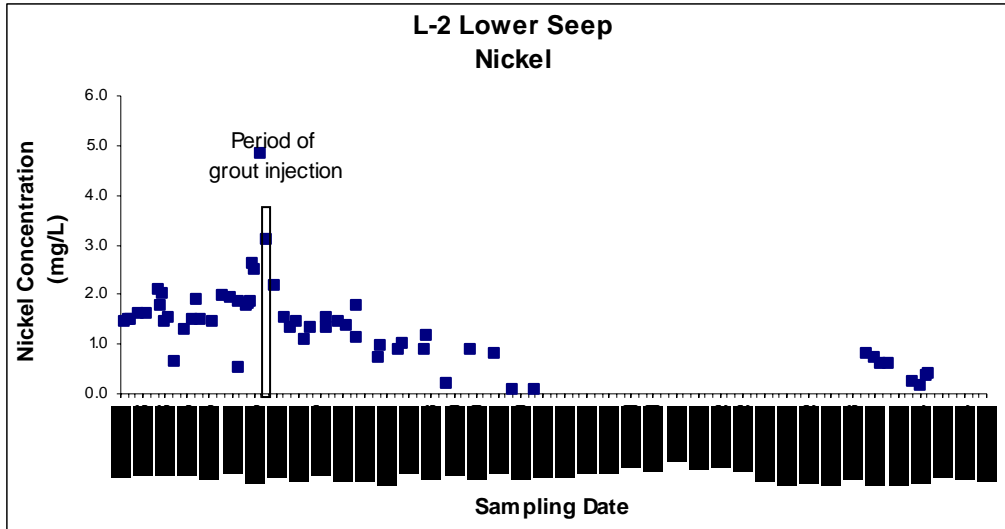


Figure 4.52

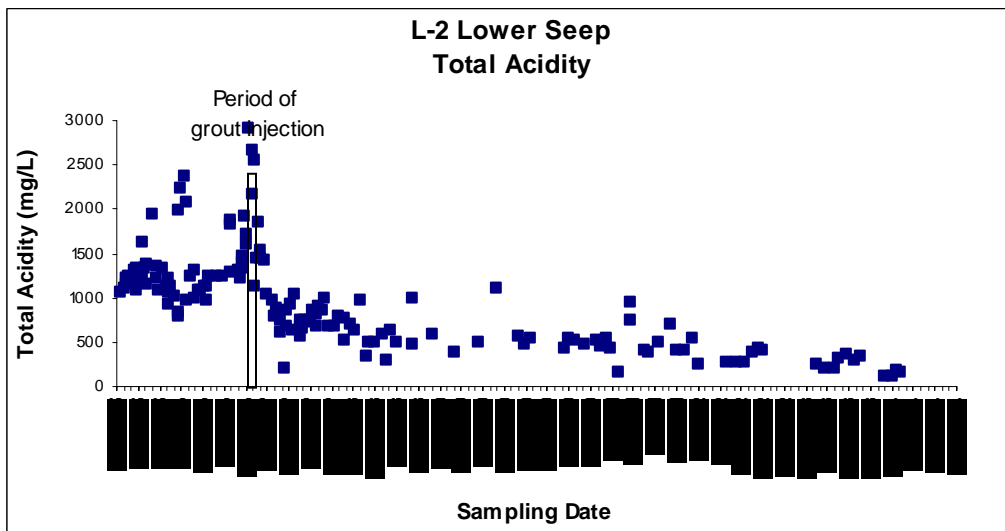


Figure 4.53

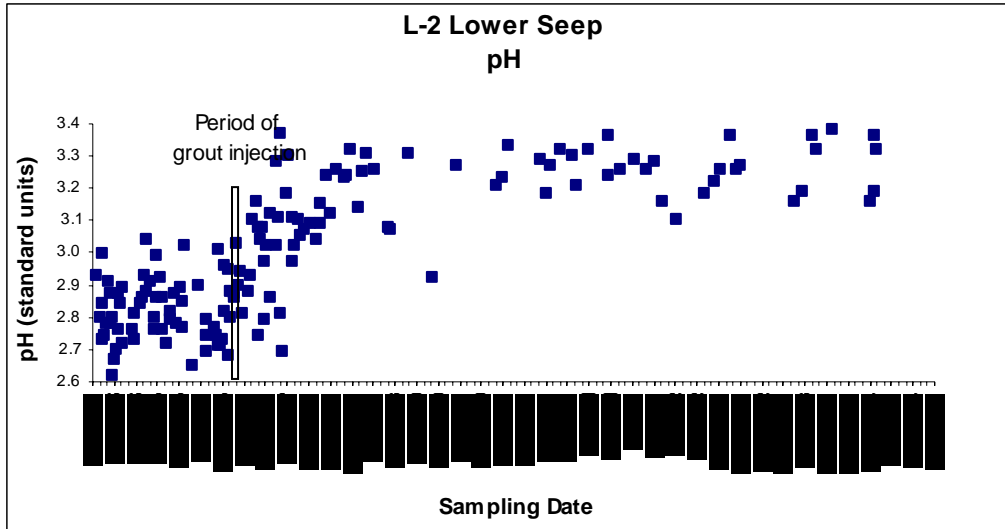


Figure 4.54

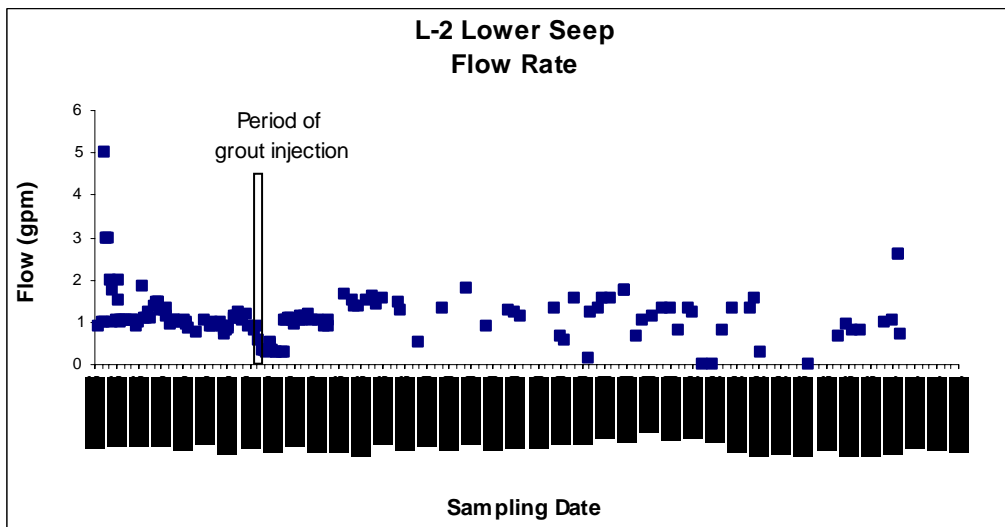


Figure 4.55

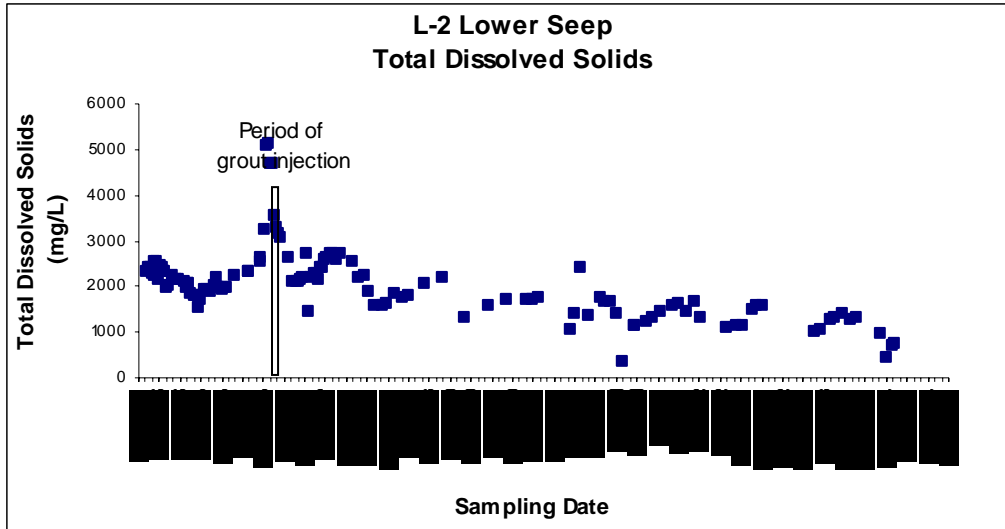


Figure 4.56