

HYDROCHEMICAL EFFECTS OF USING COAL COMBUSTION BYPRODUCTS AS STRUCTURAL FILL AND CAPPING MATERIAL AT AN ABANDONED MINE LANDS RECLAMATION SITE, SOUTHWESTERN INDIANA¹

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Abstract. The use of coal combustion by-products (CCBs) in mine reclamation has been advocated by some because of their low permeability and potential to generate alkalinity. However, others have argued that these benefits are outweighed by the potential for leaching of trace elements that can enter ground and surface waters. In 1996, an abandoned mine land (AML) site in southwestern Indiana was reclaimed using ponded ash as structural fill in highwall cuts, and fixated scrubber sludge (FSS) as capping material over pyritic refuse. Pre-reclamation and post-reclamation monitoring of surface water discharge from the site, groundwater elevations and chemistry, as well as soil moisture fluctuations in the unsaturated zone have provided a basis for evaluating the effects of CCBs on the hydrochemistry of the site and potential for off-site impacts. Limited recharge through the FSS is supported by barometric efficiency changes in the refuse aquifer, the presence of perched water measured in monitoring wells installed on the cap, and minimal fluctuations in soil moisture within and immediately below the cap. Reductions in oxygenated rainwater reaching the refuse are indicated by groundwater chemistry data, collected from the refuse aquifer between 1995 and 2007, which show an increase in pH along with decreasing trends in total acidity, specific conductivity (SpC), and arsenic. Concentrations of boron, a trace element commonly associated with CCBs, have declined to near pre-reclamation levels at most sites (~1 mg/L) after an increase immediately following reclamation. Although arsenic concentrations at 14 µg/L (EPA maximum contaminant level, or MCL, is 10 µg/L) along with boron (14 mg/L) remain slightly elevated in groundwater associated with ash-filled lakes, improvements in surface water quality leaving the site include significant reductions in total mineral acidity and total iron concentrations, while trace metal concentrations remain below EPA MCLs.

Additional Key Words: groundwater recharge, fixated scrubber sludge (FSS)

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Introduction

The use of coal combustion by-products (CCBs) as structural fill in mine reclamation continues to meet opposition from those concerned that harmful concentrations of trace elements may be leached back into the environment (Vories, 2002). Laboratory and mesocosm experiments examining the chemical and leaching characteristics of CCBs have shown soil/water increases in pH in addition to elevated B and Cu in leachate (Laperche and Bigham, 2002) as well as As, B, Se, and Mo in plant biomass (Punshon et al., 2001). Physical laboratory tests on CCBs have yielded permeability values for dewatered fly ash as well as dewatered and/or stabilized flue gas desulfurization (FGD) material that were less than 1×10^{-6} cm/s (Butalia and Wolfe, 1999; Smith, 1992; Soliman, 1990). However, with the exception of Lamminen et al. (2001) who examined FGD grout used in an underground mine, there have been surprisingly few studies conducted on the effects of CCBs in field settings where ambient conditions can be very different from those imposed in a laboratory.

In an effort to determine the long-term physical and chemical impacts of CCBs on an AML reclaimed mine site, baseline conditions (including both physical and geochemical parameters) were established for groundwater and surface water at a site where reclamation efforts included using coal ash as structural fill and fixated scrubber sludge (FSS) as capping material. The baseline data are compared with intensive post-reclamation monitoring of surface water discharge leaving the site, multiple groundwater settings where CCBs were emplaced and temporal soil moisture fluctuations within each setting. The results of our investigation show that direct recharge of oxygenated water into the coarse refuse was successfully reduced through capping with CCBs and concentrations of trace elements commonly leached from these materials generally remain below EPA maximum contaminant levels (MCLs).

Study area

The Patoka River basin in southwestern Indiana (USA) has been severely impacted by acid mine drainage (AMD) resulting from extensive surface and underground coal mining that commenced over a century ago. The upper watershed is a rugged highland characterized by high rates of rainfall runoff (Corbett, 1969). Surface mining that took place before the enactment of the Surface Mining Control and Reclamation Act (SMCRA) has altered the natural hydrologic regime of headwater catchments through the generation of lakes in unreclaimed highwall pits,

abandoned piles of pyritic coal and shale, and ridges of overburden soil and bedrock, all of which act as reservoirs for meteoric water. Recharge to unconsolidated aquifers within coarse-grained refuse piles generates abundant AMD through the transport of pyrite oxidation products and, in some settings; highwall lakes store acidity as well. AMD is then issued to streams via groundwater seeps or outflow from lakes.

The Midwestern Abandoned Mine Site lies in a headwater catchment of the Patoka River basin where both surface and underground mining operations took place between 1895 and 1983 (Fig. 1). Pyrite and associated weathering products were distributed among a large coarse-grained refuse pile in the central lowland, a network of underground mine workings, and several highwall lakes in an area encompassing approximately 50 hectares. Acidic outflow at the site was primarily generated by a centrally located spring issuing from the underground mine workings (Fig. 1) and baseflow from the coarse refuse deposit. Additional contributions of AMD resulted from surface runoff that dissolved acid-generating salts brought to the surface of refuse-laden regolith during extended periods of evaporation, and outflows from the pit lakes during wet periods.

In 1996, an effort was made to limit direct recharge of oxygenated water to the refuse aquifer and to introduce alkalinity into the site's hydrochemical system, approximately 450,000 m³ of CCBs were utilized. A passive anoxic limestone drain (PALD) was installed to treat water flowing from the central spring. The CCBs were capped with soil and a network of riprap channels was constructed to direct surface runoff and treated groundwater into two settling ponds before discharging from the site.

CCBs used at the site included pond ash (3:2 ratio, fly ash: bottom ash) from a nearby power station that was used as structural fill in the central lowland and adjacent mined depressions. The ash was removed from collection ponds where it was originally deposited, subsequently drained, and then stockpiled prior to reclamation such that indeterminate leaching occurred prior to emplacement. Fixated scrubber sludge (FSS), a mixture of fly ash, desulfurization sludge, and quick-lime (~1:1 ratio, fly ash: FGD sludge with 1.5 to 2.0 percent quicklime) was used as a cap material over the coarse-grained refuse deposit and ash-filled south pond. The FSS and ash were then overlain by a 1-m-thick layer of soil that consisted of reworked spoil and animal waste

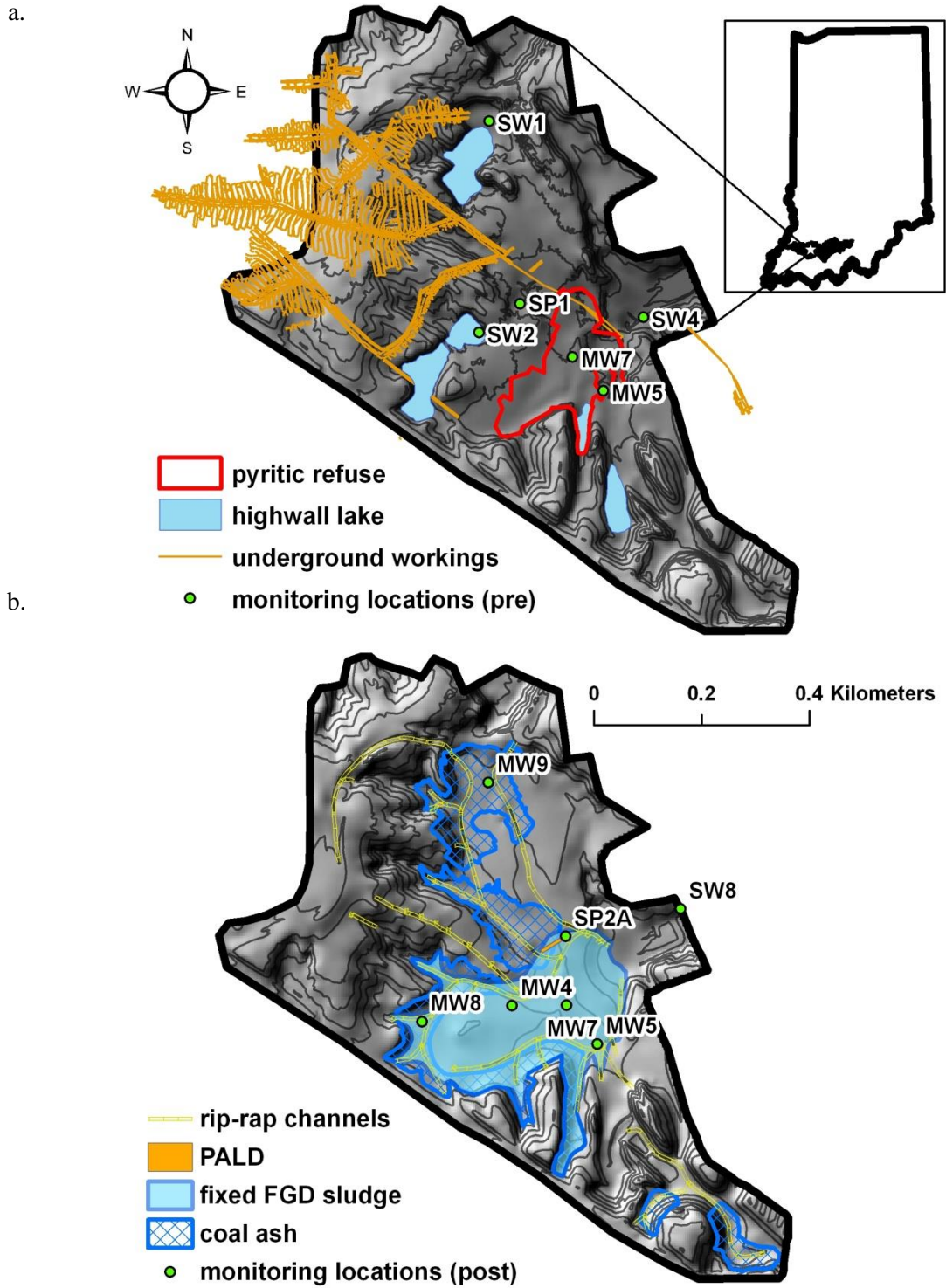


Figure 1. (a) Site figure showing pre-reclamation topography, mine features, and monitoring locations (inset map shows location of site within Patoka River Watershed in Indiana). (b) Post-reclamation topography, distribution of CCBs used in reclamation, and monitoring locations.

fertilizer that was used to promote revegetation. Additional details regarding CCB emplacement can be found in Bryenton and Gasper (1996).

Branam et al. (1999) presented bulk chemical analysis results for trace elements in CCBs from the source facility where ponded ash and FSS were produced for the Midwestern reclamation effort (data were provided by project engineering consultants). They noted high B and Pb concentrations for both FSS and ponded ash while Fig. 2 indicates that fly ash contains the greatest amount of B as well as measurable quantities of As, indicating that elevated concentrations of these two elements in ponded ash and FSS may be primarily attributable to the fly ash component. Laboratory leaching studies conducted on ponded ash showed that B was the most concentrated element in leachate following 30-day leaching experiments (~1.5 mg/l) while Mo leached from FSS following 30 days displayed the highest percentage leached for all trace elements at ~3% (Branam et al., 1999). The relative mobility of B has been studied by other authors such as Pagenkopf and Connolly (1982) who noted that B concentration in leachate is dependent upon contact time, pH, and ash to leachate ratio. Both B and As concentrations at post-reclamation groundwater monitoring locations are discussed later herein.

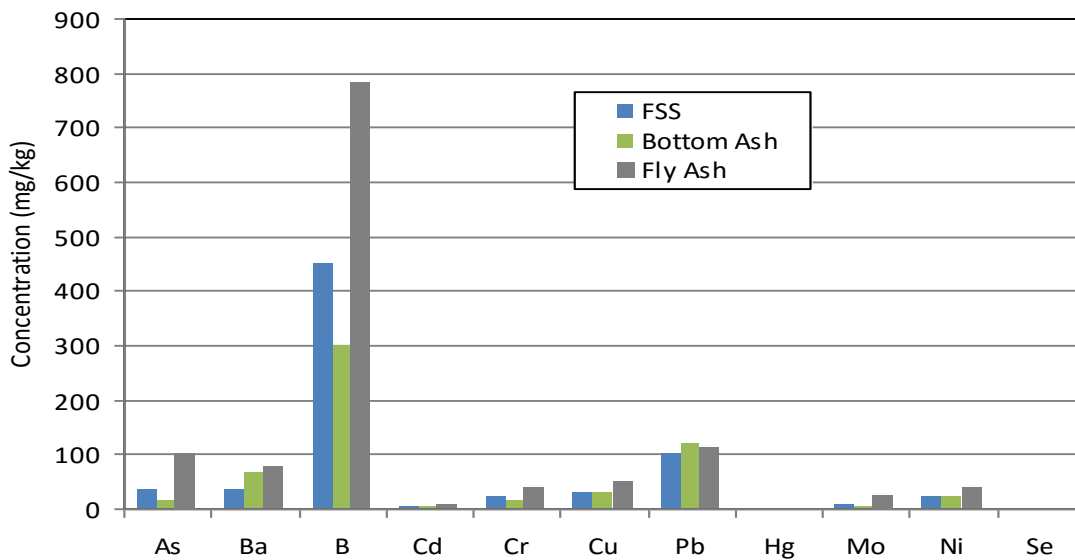


Figure 2. Figure modified from Branam et al. (1999) showing trace element concentrations for FSS and component CCBs used to produce FSS and ponded ash (the figure has been modified to include trace element concentrations for bottom ash and fly ash). Fly ash and bottom ash samples were analyzed prior to being wet sluiced at the production facility.

Methods

Monitoring network

Pre-reclamation monitoring stations identified in Fig. 1 were emplaced in an effort to establish hydrochemical conditions prior to reclamation. These consisted of two highwall lake sampling sites (SW1 and SW2), the main discharge location of the flooded mine workings (SP1), and three groundwater monitoring stations (MW1, MW5, and MW7). In addition, a v-notch weir was installed at the outlet of the site (SW4) where samples representative of the final effluent were collected and runoff determinations could be made.

Monitoring wells, installed during and after reclamation, were positioned such that reasonable comparisons could be made with pre-reclamation conditions (Table 1). All monitoring wells were constructed using 51-mm PVC pipe. Pressure transducers were installed to monitor continuous groundwater level fluctuations at MW5, MW7, MW8, and MW9 (Fig. 1). Nested wells were installed at MW7 to compare water levels above the FSS cap and in the underlying refuse aquifer and at MW8 to measure levels above the cap and in the underlying pond ash deposits. Aluminum access tubes were installed adjacent to MW4, MW7, MW8, and MW9 to allow measurements of soil moisture profiles that were used to determine soil moisture storage above the water table in the coarse refuse deposit, and in the areas of ash fill. Pre-reclamation soil moisture data were also collected adjacent to the MW7 monitoring well.

Table 1. Descriptions of pre- and post-reclamation monitoring sites. The pre-reclamation period was from April 10 to October 25, 1995. The reclamation period was from January 11 to October 7, 1996. The post-reclamation monitoring period was from November 18, 1996, to June 13, 2007.

Site ID	Description	Pre-reclamation	Post-reclamation	Screen depth interval (meters)
SW1,2	Ponds in final-cut pits	X		
SW4	Stream at outlet to reclamation site	X		
SW8	Stream at outlet to reclamation site		X	
SP1	Spring issuing from underground mine	X		
SP2A	Outlet from passive anoxic limestone drain (PALD)		X	
MW4	Well in spoil deposit below FSS cap		X	3.1-4.7
MW5	Well at east edge of pyritic refuse	X		5.2-5.5
MW5S	Well in spoil deposit at edge of FSS cap		X	2.0-3.6
MW7	Well in pyritic refuse	X		1.8-3.4
MW7S	Well above pyritic refuse and FSS cap		X	0.8-1.8
MW7D	Well in pyritic refuse below FSS cap		X	3.3-4.8
MW8	Well in pond ash that filled a final-cut pit		X	4.4-5.9
MW9	Well in pond ash that filled a final-cut pit		X	7.7-10.8

Water balance

The water balance approach was used to estimate groundwater recharge for the reclaimed area using a simplified water balance equation:

$$R = P - ET - \Delta S - R_o \quad (1)$$

where R is groundwater recharge, P is precipitation, ET is evapotranspiration, ΔS is change in soil moisture storage, and R_o is site runoff (all computed in cm of water).

A tipping bucket rain gauge was used to measure precipitation and evapotranspiration was measured using a small weighing lysimeter (27 cm in diameter and 18 cm in depth) that utilized a collection device for surface runoff and a free-draining base to collect vertical drainage. Soil and vegetation from the surrounding area were used to simulate natural conditions in the lysimeter container. Runoff was determined by integrating discharge measurements using a v-notch weir and pressure transducer (at SW8) and dividing by the area of the watershed (47 ha) that contributed to the outflow from the site. A weir was also installed to capture flow from the PALD which was derived mostly from the underground mine workings. Pre-reclamation baseflow was determined through visual hydrograph analysis and subtracted from total discharge measured at SW8. Soil moisture data were collected at 0.3-meter intervals for sites MW4, MW5, MW7, MW8, and MW9 and integrated with respect to depth to determine storage in the vadose zone (Diodato and Parizek, 1994). Soil moisture storage changes for the recharge analysis period were averaged for all five locations to determine the ΔS component in equation 1.

Geochemical characterization

Water samples were collected at all monitoring sites in both the pre- (4/95–10/95) and post-reclamation phases on a quarterly basis with additional samples collected to evaluate possible temporal trends in trace metal concentrations and to evaluate the chemical composition of selected storm runoff events. Major cations Ca, Mg, K, Na, Fe^{2+} , and Fe^{3+} ; minor cations Al, Mn, Ni, Sr, and Zn; as well as trace elements As, Sb, Ba, B, Cd, Cr, Cu, Pb, Se, and Ag were determined using inductively coupled plasma mass spectroscopy and atomic absorption spectroscopy after field samples were filtered and split into aliquots with the metals and trace elements aliquot acidified with HNO_3 . Major anions F^- , Cl^- , NO_3^- , HCO_3^- , and SO_4^{2-} were also determined. Alkalinity and acidity were determined using standard methods, with measured acidity being compared to calculated acidity using the equation described by Watzlaf et al.

(2004) for quality assurance purposes. These comparisons showed good agreement between measured and calculated acidity for all samples containing no alkalinity, generating typical deviations of less than 5%. When alkalinity was present, the difference between measured and calculated acidity was more commonly in the 10 to 30% range for net acid samples and up to 80% for net alkaline samples. Due to this discrepancy, calculated potential acidity was utilized for all water samples to define geochemical characterizations. Field measurements, including pH (sensor calibrated with standard buffer solutions), specific conductivity, Eh (sensor checked with Zobelle’s solution), and temperature were also made on each sample.

Results

Evaluation of hydrologic changes resulting from reclamation

Using the residual water balance method (Equation 1) groundwater recharge was estimated for two periods of equal duration before and after CCB emplacement and for a 1-year period following reclamation (Table 2). In the pre-reclamation period, 25% of the total precipitation was estimated to have contributed to recharge. The post-reclamation calculations indicated that only 16 to 19% of total precipitation was contributed to recharge (~30% reduction) and this is likely due to the emplacement of FSS and the construction of the surface drainage network. This interpretation is supported by a statistical comparison of water level changes at MW7 during the pre- and post-reclamation monitoring periods.

Table 2. Water Balance Calculations

Period	<i>P</i> (cm)	<i>ET</i> (cm)	<i>Ro</i> (cm)	ΔS (cm)	<i>R</i> (cm)
7/14/95 - 10/10/95	15.0	-8.8 (59%)	2.5 (17%)	NA	3.7 (25%)
7/14/00 – 10/10/00	36.3	-24.5 (67%)	6.2 (17%)	-0.1 (0%)	5.8 (16%)
7/14/00 - 7/18/01	94.1	-54 (57%)	19.9 (21%)	1.9 (2%)	18.3 (19%)

Time-series regression analyses were undertaken to evaluate the relative roles of rainfall recharge and atmospheric pressure changes in determining water-level fluctuations in the saturated coal refuse near MW7. Two sample sets were selected; one set for the pre-reclamation period when the refuse was exposed at the surface and readily received rainfall recharge, and one set for the post-reclamation period when the refuse was buried beneath the FSS layer and soil cap. The statistical model being evaluated had the following form:

$$\Delta WL_t = b_0 + b_1 P_t + b_2 \Delta B_t + \rho e_t + \mu_t \quad (2)$$

where ΔWL (cm) is the daily change in water level, P_t (cm) is daily precipitation, and ΔB is the daily change in barometric pressure. The results of the regression analysis are presented in Table 3. Because the data were collected as a continuous time series, the model includes an autocorrelated error term (ρe_t) as well as the usual random error term (μ_t).

As indicated by the statistically significant t-ratios associated with the regression coefficients relating pressure changes and precipitation to water level changes, both of the independent variables were influencing the water-level changes prior to reclamation. A comparison of the regression coefficients themselves indicates that the effect of precipitation was nearly five times stronger than that of pressure change in the pre-reclamation period. In contrast, the results for the post-reclamation period indicate that precipitation events no longer directly influence changes of water level in the buried refuse. Instead, approximately 90 percent ($R = 0.89$) of the variability in daily water-level changes is attributed to changes in atmospheric pressure following reclamation. The regression coefficient relating pressure changes to water-level changes is known as the "barometric efficiency" of the aquifer. The barometric efficiency of the aquifer in the refuse increased almost four-fold from the pre- to the post-reclamation period. The high barometric efficiency that now prevails is indicative of a confined aquifer that does not have a free connection to the overlying unsaturated zone.

Table 3. Statistical analysis of water level changes in refuse aquifer.

Pre-reclamation (1995)		n=140, R=0.56, ρ =0.13	
	parameter estimate	standard error	t-ratio
b_1	-0.21	0.05	-4.55
b_2	1.05	0.19	5.48
Post-reclamation (1998)		n=235, R=0.89, ρ =0.00	
	parameter estimate	standard error	t-ratio
b_1	-0.76	0.03	-28.9
b_2	0.28	0.17	1.70

The effectiveness of the fixated FGD material is supported by perched water above the FSS (Fig. 3) and also by soil moisture fluctuations that are greatest above the cap at MW4, MW7, and MW8 as shown in Fig. 4. MW4 and MW7, located near the center of the capped area (Fig. 1), display the greatest variance in soil moisture in the upper 1-meter of the vadose zone, indicating that soil moisture retention is occurring during wetting periods (as opposed to direct vertical recharge of oxygenated water), and evapotranspiration is prominent during dry periods. The coal ash emplaced at MW8 lies within the zone of saturation whereas the upper 3+ meters of ash utilized as structural fill at MW9 lies in the unsaturated zone.

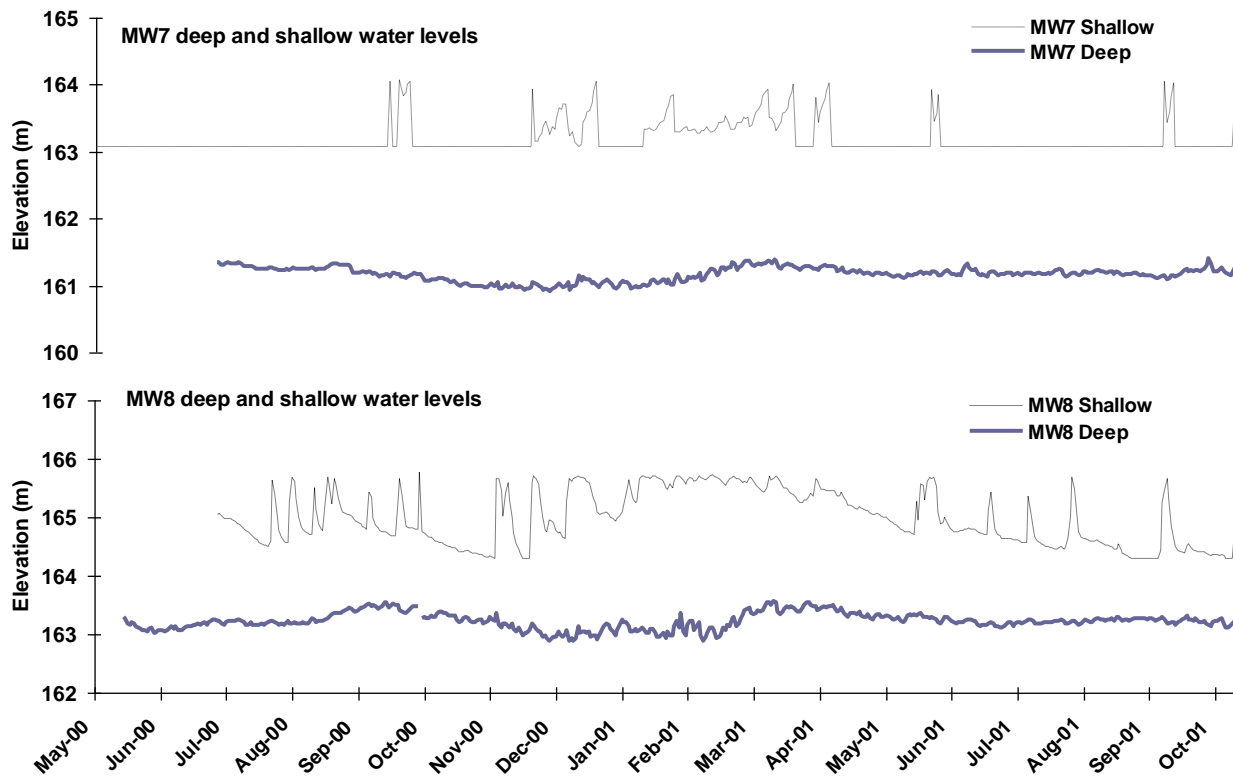


Figure 3. Water level elevations at MW7 and MW8 measured from May 2000 to October 2001 with shallow wells screened atop FSS displaying perched water.

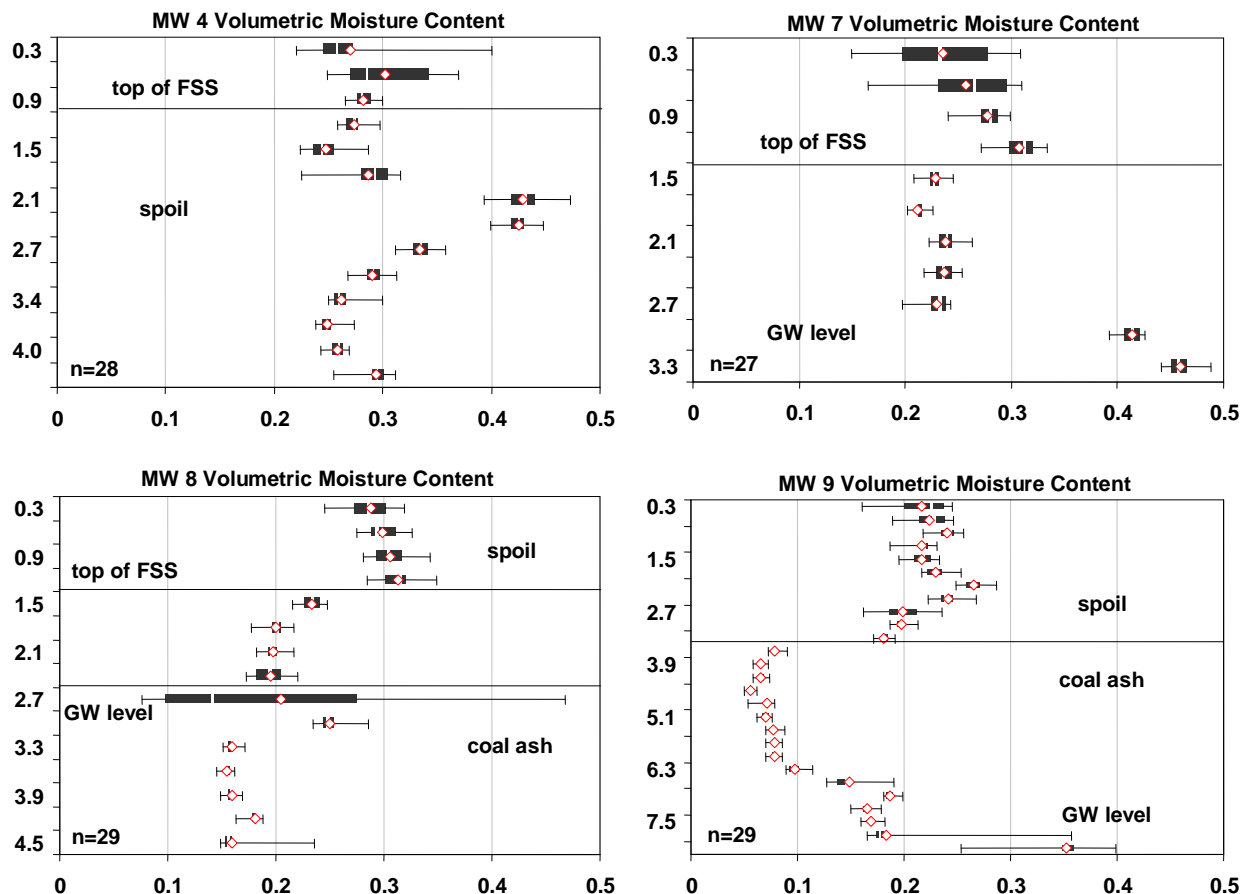


Figure 4. Volumetric soil moisture at MW4, MW7, MW8, and MW9 obtained from bi-weekly neutron probe measurements. Volumetric moisture content is shown on the X axis while depth in meters is displayed on the Y axis. Statistical parameters are shown by whiskers that represent minimum and maximum moisture values, boxes the 25th and 75th percentiles, vertical white lines the median values, and red squares the mean value for each data set.

Changes in water chemistry following reclamation

Average values of water chemistry from monitoring stations of the pre- and post-reclamation periods are listed in Table 4. When comparing the pre-reclamation surface water discharge (SW4) with those of the post-reclamation period (SW8), one can see that most of the major geochemical indicators for AMD were significantly improved following reclamation. Total Fe concentrations in site effluent have decreased from 200–300 mg/l prior to reclamation to less than 10 mg/l in 2007 (Fig. 5). Alkalinity at the site outlet was greatest during the 1-year period following reclamation (50–170 mg/L), but has more recently been absent or present at lower concentrations (Fig. 5). At each of the locations where CCBs were used, there were increases in pH and alkalinity, with the exception of MW7 where alkalinity remained absent following

reclamation. Concentrations of total Fe in samples collected from MW7D have decreased 5-fold following reclamation to less than 800 mg/L in 2007. Together, these findings are consistent with the conclusions of Branam et al. (1999, 2005) as well as Harper and Olyphant (1999) and lend support to the contention that the FSS is preventing further recharge and leaching of the pyritic refuse.

Table 4. Average values, ranges of values, and numbers of samples (n) at selected water-monitoring sites (see figure 1 for site locations).

	SO ₄ (mg/l)	Fe (mg/l)	Acidity (mg/l)	Alkalinity (mg/l)	pH	SpC (µmhos/cm)	TDS (mg/l)
Pre-reclamation water chemistry (April through August, 1995)							
SP1 n=3	1380 1220-1540	76 65-82	369 193-720	11 0-34	4.1 3.7-5.1	1958 1927-1988	2033 1900-2100
MW7 n=3	12967 8200- 17500	4433 2800- 5700	11732 7507- 15817	0 0-0	1.4 1.1-1.8	22093 13700-32800	23333 12000- 35000
SW4 n=4	2353 2280-2500	243 190-330	714 451-901	0 0-0	2.9 2.8-3.1	3215 3030-3350	3650 3300-3900
SW2 n=4	550 370-690	34 6-80	278 92-523	0 0-0	2.7 2.6-3.1	1479 930-1758	848 600-970
SW1 n=4	186 94-240	6 1-18	104 11-350	0 0-0	4.2 4.0-5.3	460 392-508	338 230-420
MW5 n=4	2695 2520-2880	285 250-320	885 825-979	0 0-0	3.1 3.0-3.4	3555 3390-3790	4275 3700-4700

Post-reclamation water chemistry for comparable monitoring sites (November 1996 through June 2007)

SP2A	1463 722-1680 n=20	86 64-120 n=19	159 122-217 n=20	267 218-323 n=20	6.4 6.0-7.3 n=20	2531 1836-2810 n=20	2632 2336-3000 n=17
MW7	8119 2192- 15900 n=20	2421 785-5700 n=18	5839 1560- 13413 n=20	0 0-0 n=20	2.3 1.6-4.0 n=20	8890 4103-20800 n=20	15122 6405-29000 n=17
SW8	1579 625-2360 n=20	28 2-83 n=19	139 0-350 n=19	32 0-140 n=20	3.5 2.7-7.0 n=20	2448 1408-3389 n=20	2452 1600-3600 n=17
MW8	1794 1650-2270 n=20	1 0-6 n=19	20 0-70 n=20	55 29-123 n=20	7.0 6.2-8.7 n=20	3014 2516-3269 n=20	4192 2686-6823 n=17
MW9	1687 1370-1972 n=18	145 16-197 n=18	372 43-578 n=17	234 104-460 n=18	6.0 5.6-7.1 n=18	2740 1832-3172 n=18	3009 2400-3600 n=17
MW5S	2745 2175-4740 n=17	275 46-482 n=17	524 110-913 n=17	110 0-330 n=16	5.2 4.3-6.8 n=17	3726 2872-4950 n=17	5019 3500-9100 n=15

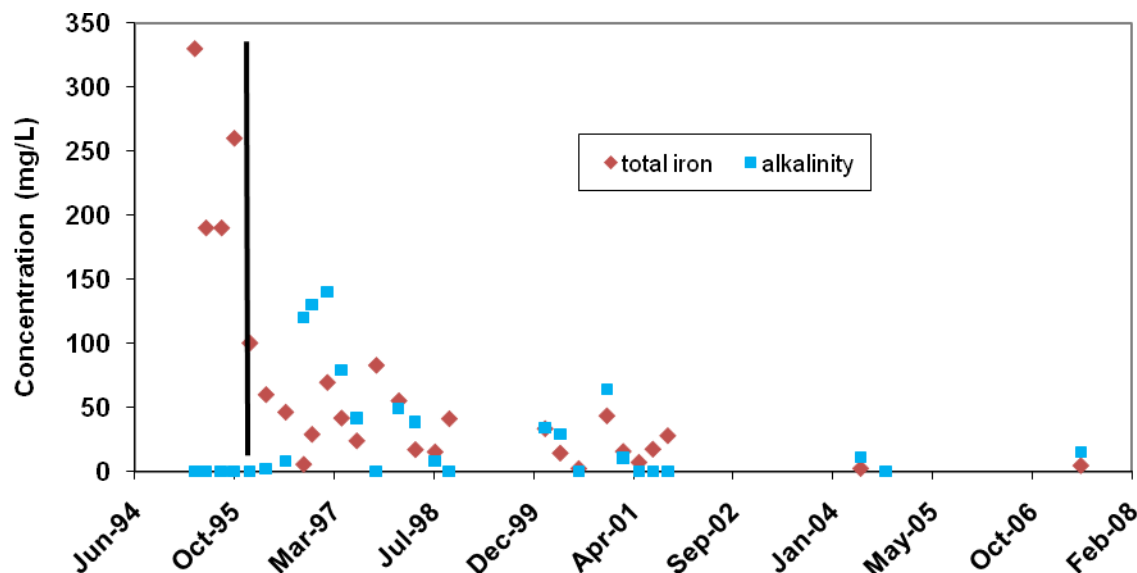


Figure 5. Time series plot of total iron and alkalinity concentrations at the site outlet (SW4: pre-reclamation, SW8: post-reclamation). Water discharging from the site contained no alkalinity prior to emplacement of CCBs and iron concentrations were significantly higher before reclamation commenced.

Field chemistry data collected between April of 1995 and June of 2007 indicate that improving water quality trends exist based on both the SpC and pH of the central refuse aquifer (Fig. 6). The increasing pH trend is even more significant when one considers the logarithmic nature of the scale, which indicates that, by raising the pH from approximately 1.5 units pre-reclamation to about 3.5 units following reclamation, there has been a 100-fold decrease in hydrogen ion activity. This is further supported by MW7 total acidity concentrations (Fig. 7) that incorporate proton activity, mineral acidity caused by dissolved metals, and organic acidity resulting from dissolved organic compounds.

Although concentrations of most all of the major chemical constituents associated with AMD (e.g., SO_4^{2-} , Fe, and As) significantly decreased following reclamation at MW7, the trace element B increased immediately following CCB emplacement (Fig. 8). Concentrations of the trace element also spiked at MW9 where a thick sequence of coal ash was used as structural fill, but by 2004, B concentrations at these sites had receded to pre-reclamation levels. Post-reclamation increases in B at monitoring well MW8 have not returned to pre-reclamation levels and instead appear to be increasing reaching 14 mg/L in June 2007. Both coal ash and fixed

desulfurization sludge were utilized in this area, and there could be relatively unique geochemical reactions occurring as a result.

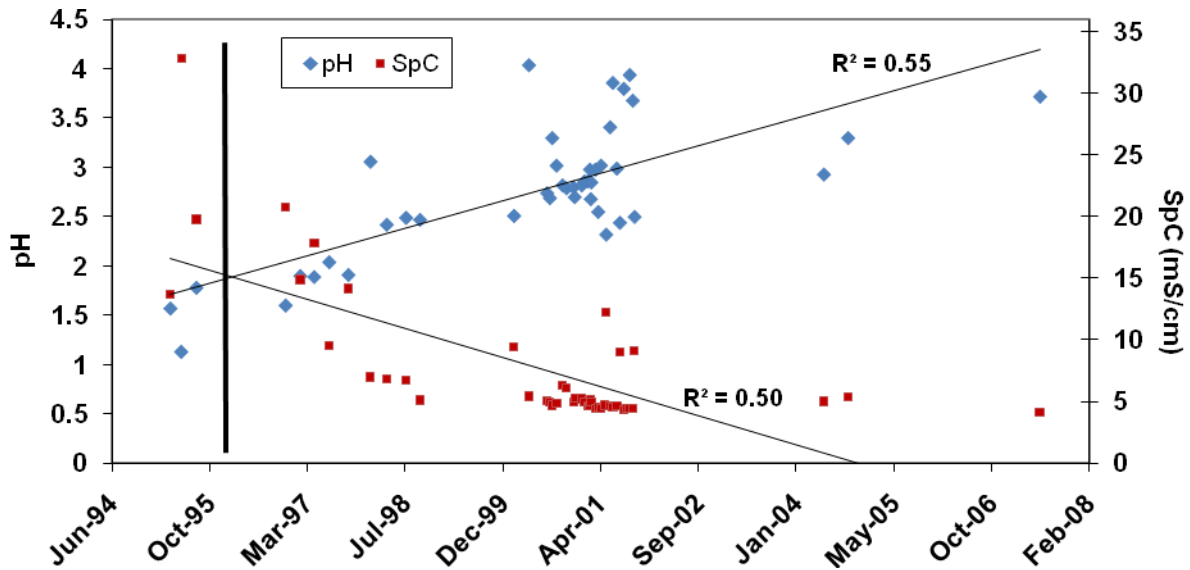


Figure 6. Field chemistry data collected between 4/95 and 6/07 at MW7. The vertical black line indicates the onset of reclamation activities. Linear regression trend lines are plotted for both data sets.

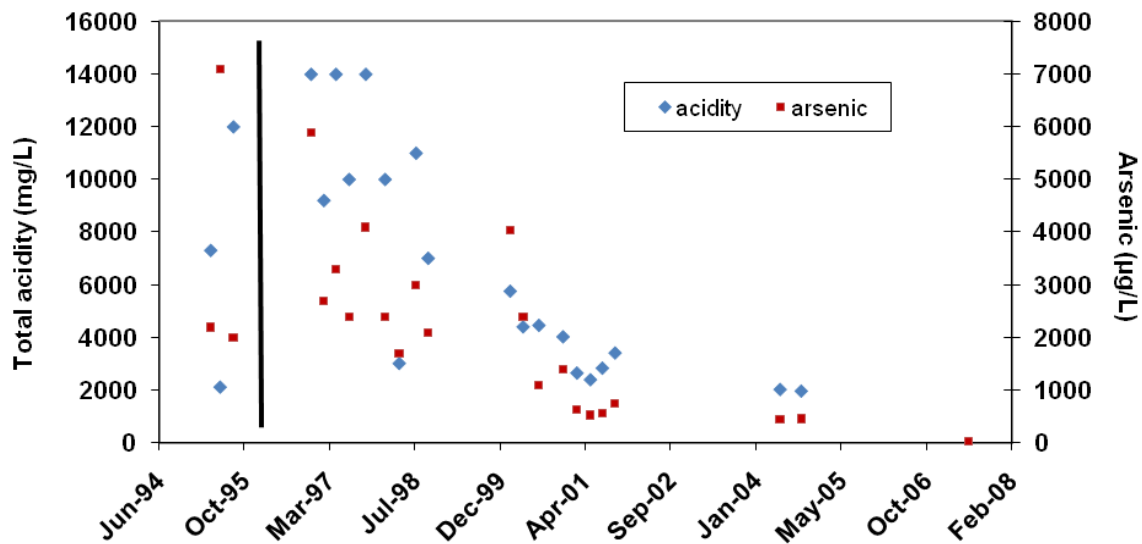


Figure 7. Total acidity and arsenic concentrations at MW7 before and after reclamation (black line indicates start of reclamation).

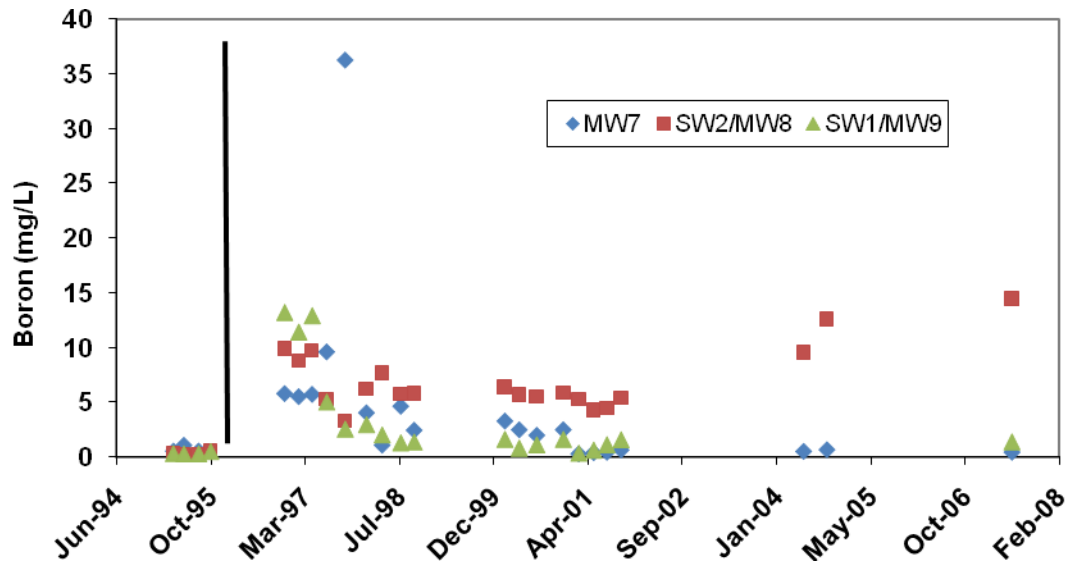


Figure 8. Boron concentrations prior to and following reclamation at MW7 (refuse aquifer), SW2/MW8 (pre-reclamation highwall lake and post-reclamation monitoring well installed in coal ash fill overlain by fixed desulfurization sludge), and SW1/MW9 (pre-reclamation highwall lake and post-reclamation monitoring well installed in coal ash fill without cap).

Arsenic concentrations have existed at or above the MCL standard for drinking water as established by EPA at three sites following reclamation: MW5S, MW8, and MW9. As noted by Branam et al. (1999), As concentrations at MW5S (near the east edge of the FSS cap and refuse pile) increased significantly immediately following reclamation (Fig. 9). However, more recent samples collected between February of 2000 and June of 2007 have indicated a substantial decrease to near pre-reclamation levels at MW5S with the latest sample yielding a concentration of 1 $\mu\text{g/L}$. MW8 and MW9 also display elevated concentrations when compared to their pre-reclamation levels with concentrations at MW8 continuing to slightly exceed the EPA maximum contaminant level in 2007 (Fig. 10).

Trace metals with established EPA MCLs (e.g., As, Se, Cu, and Cr) were all well below the established guidelines at the site outlet (SW8) following reclamation and other trace elements associated with CCBs (e.g., B and Mo) displayed no noticeable increase in surface water leaving the site.

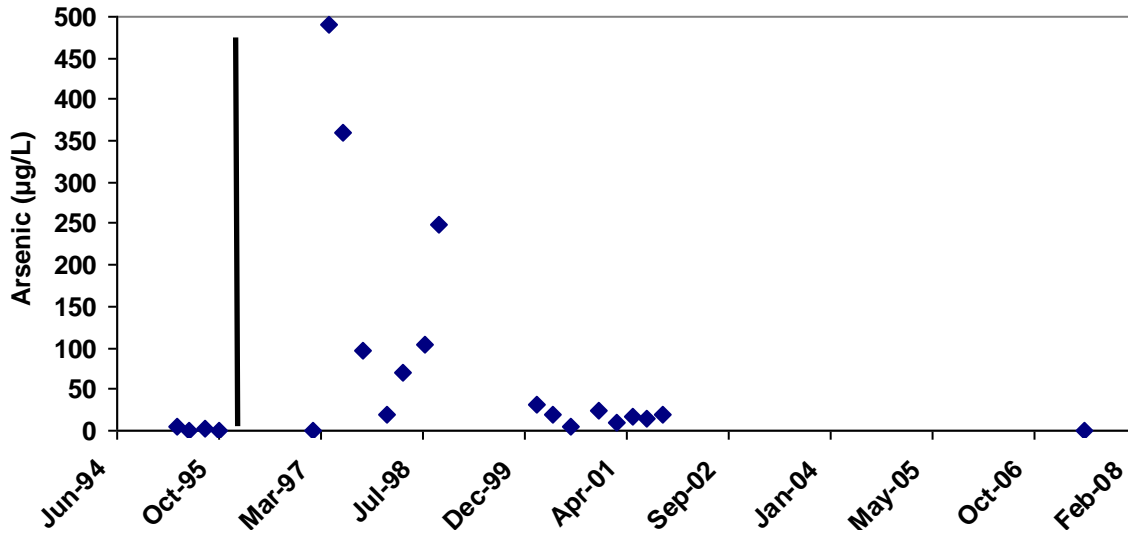


Figure 9. Arsenic concentrations at MW5S (a shallow well installed near the edge of the FSS cap and coarse pyritic refuse deposit).

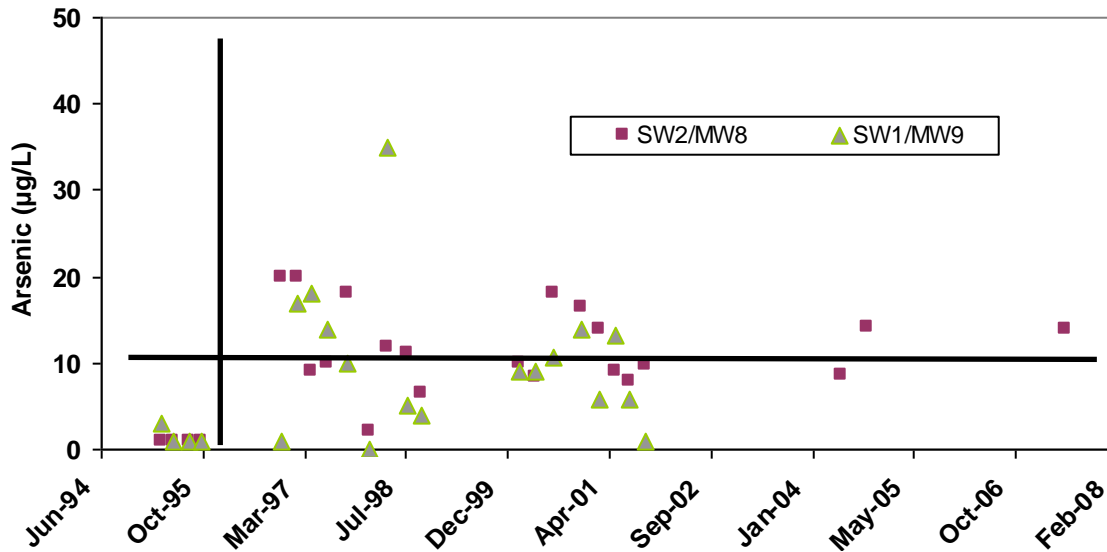


Figure 10. Arsenic concentrations at SW2/MW8 (pre-reclamation highwall lake and post-reclamation monitoring well installed in coal ash fill overlain by fixed desulfurization sludge) and SW1/MW9 (pre-reclamation highwall lake and post-reclamation monitoring well installed in coal ash fill without cap). The horizontal line indicates the EPA MCL for arsenic in drinking water.

Conclusions

The 30% reduction in groundwater recharge, estimated using the water balance approach, is expected when one considers that the FSS cap material was applied to 19% of the study area. The presence of perched water above the cap and limited variance in soil moisture data below

support the contention that the material is acting as an effective recharge barrier 4 to 5 years following reclamation based on data collected from 2000 to 2001. These data coupled with increases in barometric efficiency of the central refuse aquifer indicate that the infiltration of oxygenated meteoric water, which is necessary to perpetuate pyrite oxidation and AMD production, has been significantly limited following reclamation.

Improvements in water quality within the refuse aquifer have resulted from the emplacement of FSS and the subsequent reduction in oxygenated water percolating through pyritic refuse. In addition to lowering concentrations of sulfate, total iron, and total acidity, reclamation efforts have produced improving long-term trends for both SpC and pH. Compared with pre-reclamation data, post-reclamation As concentrations are elevated near final cut pits filled with coal ash suggesting that leaching may be occurring from ash used as structural fill. As noted by Branam et al. (1999), B is a relatively conservative tracer associated with CCBs and its concentrations are higher at the ash-filled pits, further indicating that the coal ash is the source of trace elements being leached at these locations. Similarly, B concentrations increased at MW5S following reclamation implying that CCBs are likely the source for As at this site (Branam et al., 1999). However, data collected 4 to 10 years after reclamation indicate that As leached from the FSS is depleted quickly over time until no further As is leached from the cap material. The results of this study illustrate numerous positive aspects of using CCBs in reclamation. Direct recharge to the central refuse aquifer has been significantly reduced using the FSS cap material and coal ash used as structural fill has provided alkalinity and increased pH for multiple former AMD reservoirs at the site while no considerable leaching of toxic trace elements has taken place.

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