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Thirteenth Annual Meeting

American Society for Surface Mining and Reclamation

SUCCESSES AND FAILURES: APPLYING RESEARCH RESULTS TO INSURE RECLAMATION SUCCESS

May 18 - 23, 1996 - Knoxville, Tennessee

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SUCCESSES AND FAILURES: APPLYING RESEARCH RESULTS TO INSURE RECLAMATION SUCCESS

May 18 - 23, 1996 - Knoxville, Tennessee

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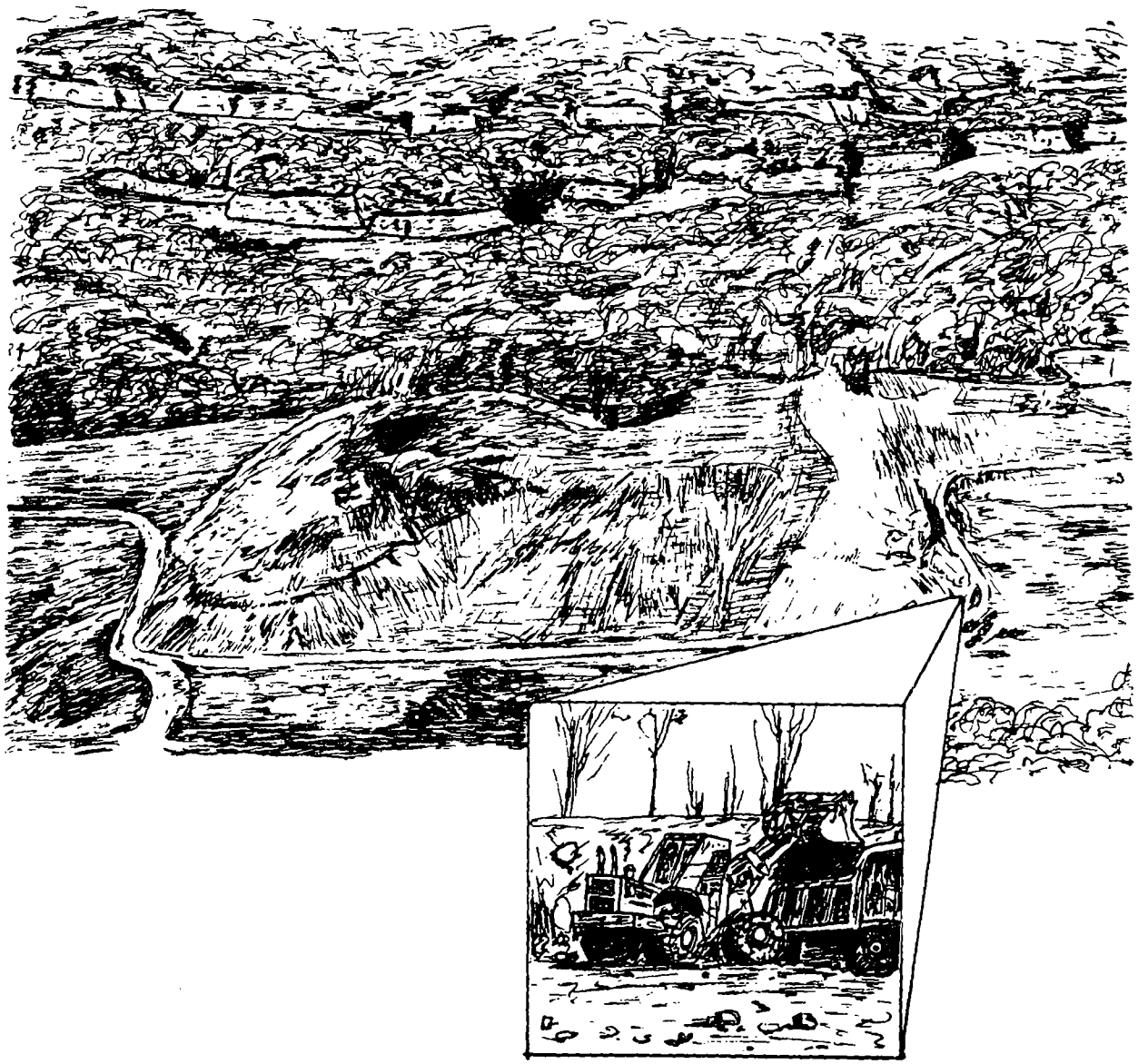

Coal surface mining in Southwest Virginia. Original drawing by J. Michael Schmidt.

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SESSION 1: CONTAINMENT AND MITIGATION OF TOXIC MINE WASTE

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AMERICAN SOCIETY • SURFACE MINING and RECLAMATION •

OXYGEN TRANSPORT AND PYRITE OXIDATION IN UNSATURATED COAL-MINE SPOIL[1]

by

Weixing Guo[2] and Charles A. Cravotta III[3]

Abstract. An understanding of the mechanisms of oxygen (O_2) transport in unsaturated mine spoil is necessary to design and implement effective measures to exclude O_2 from pyritic materials and to control the formation of acidic mine drainage. Partial pressure of oxygen (Po_2) in pore gas, chemistry of pore water, and temperature were measured at different depths in unsaturated spoil at two reclaimed surface coal mines in Pennsylvania. At mine 1, where spoil was loose, blocky sandstone, Po_2 changed little with depth, decreasing from 21 volume percent (vol %) at the ground surface to a minimum of about 18 vol % at 10 m depth. At mine 2, where spoil was compacted, friable shale, Po_2 decreased to less than 2 vol % at depth of about 10 m. Although pore-water chemistry and temperature data indicate that acid-forming reactions were active at both mines, the pore-gas data indicate that mechanisms for O_2 transport were different at each mine. A numerical model was developed to simulate O_2 transport and pyrite oxidation in unsaturated mine spoil. The results of the numerical simulations indicate that differences in O_2 transport at the two mines can be explained by differences in the air permeability of spoil. Po_2 changes little with depth if advective transport of O_2 dominates as at mine 1, but decreases greatly with depth if diffusive transport of O_2 dominates, as in mine 2. Model results also indicate that advective transport becomes significant if the air permeability of spoil is greater than 10^{-9} m^2, which is expected for blocky sandstone spoil. In the advective-dominant system, thermally-induced convective air flow, as a consequence of the exothermic oxidation of pyrite, supplies the O_2 to maintain high Po_2 within the deep unsaturated zone.

Additional Key Words: advection, diffusion, numerical simulation, heat transfer

Introduction

Atmospheric oxygen (O_2) is required for pyrite oxidation and the consequent formation of acidic mine drainage (AMD). Gaseous or dissolved O_2 is necessary for the direct oxidation of pyrite and for the regeneration of ferric iron (Fe^{3+}), which is an important oxidant (Singer and Stumm 1970). Hence, an understanding of the mechanisms of O_2 transport in mine spoil is necessary to design and implement effective measures to control the formation of AMD.

Pyrite oxidation takes place mainly in the unsaturated zone of mine spoil. Although infiltrating water carries some dissolved O_2 into mine spoil, gaseous transport of O_2 generally is more effective than aqueous transport of O_2 because of the low solubility and low diffusivity of O_2 in water (Watzlaf 1992).

Gaseous O_2 transport through air-filled voids in spoil involves both processes of diffusion and advection. Diffusive transport is the movement of O_2 molecules from zones of high O_2 to zones of low O_2 concentration. Steep gradients in O_2 concentration can be formed between the exterior and interior of spoil because of relatively rapid O_2 consumption by oxidation reactions and slow diffusive transport of O_2. If air permeability is low, such as in compacted, shaly spoil, diffusive transport can be the major mechanism

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of O_2 transport. However, if air permeability of spoil is high, such as in blocky sandstone spoil, advective transport of O_2 by moving air can predominate. Thermal gradients within spoil can induce the inward (lateral) flow of oxygenated air to replace outward (upward) flowing air that is warmed by the exothermic oxidation of pyrite (Harries 1969). Advective transport can also result from rapid changes in barometric pressure and from displacement of air by ground water.

Approaches to controlling the formation of AMD include minimizing the exposure of pyrite to fresh air and oxygenated or Fe^{3+}-rich waters (Kleinmann and Erickson 1986; Barton-Bridges and Robertson 1989; Watzlaf 1992). However, the processes that form AMD in unsaturated spoil are complex and involve many hydrological, chemical, and biological factors (Williams et al. 1982). Numerical models have been developed to evaluate the interactions of these factors and the generation of AMD (Cathles and Apps 1975; Jaynes et al. 1984a,b; Pantelis and Ritchie 1992).

In this study, evaluations were made on the relative importance of diffusive and advective transport of O_2 on the distribution of O_2 in coal-mine spoil and the effect of O_2 supply on the formation of AMD. Composition of pore gas, temperature, and water quality were measured at different depths in spoils at two reclaimed surface coal mines in Pennsylvania. A two-dimensional numerical model that was developed to study the relation between O_2 transport and the rate of acid generation (Guo 1993) is used to explain the observed chemical compositions of pore-gas and pore-water samples from the mines.

Field Studies of Oxygen in Mine Spoil

The availability of O_2 in mine spoil is controlled by the dynamic balance between the rate of O_2 consumption and the rate of O_2 supply. Consumption of O_2 results from pyrite oxidation, biological respiration, and other reactions. The rate at which O_2 is replenished at reaction sites is controlled largely by the air permeability, moisture content, temperature, the O_2 partial pressure (Po_2) gradient, and the strength of air convective flow within surrounding spoil. Therefore, the availability of O_2 in mine spoil can vary spatially and temporally.

Field investigations of O_2 in unsaturated spoil have been conducted at several surface coal mines in western Pennsylvania. Although active pyrite oxidation and AMD were problems at all these mines, the spatial distribution of Po_2 in pore-gas samples was extremely variable among and within the mines. Jaynes et al. (1983) reported fairly high Po_2. Po_2 in several boreholes decreased from 21 vol % at the ground surface to greater than 10 vol % at a depth of 12 m. At a second mine, Fielder (1989) reported Po_2 as high as 15 vol % down to 9.1 m; however, in adjacent spoil containing pyritic refuse, Po_2 was less than 1 vol %. At another mine, Lusardi and Erickson (1985) reported annual mean values of Po_2 as low as 1.3 vol % and as high as 10.2 vol % at a depth of 4.6 m in different boreholes. Data for two additional mines that were collected by the authors are representative of these extreme ranges and are described below.

Mine 1. Advective-Transport Dominant

Mine 1 is a reclaimed, abandoned surface coal mine on a broad hilltop (40°55'30"N, 78°22'30"W) in southern Clearfield County (Guo 1993). During 1968-85, the lower Kittanning coal was mined. The lower Kittanning coal and a rider seam with associated coaly shale were pyritic and had total sulfur concentrations of 3.8 and 5.4 weight percent (wt %), respectively. Spoil material was predominantly blocky sandstone and siltstone. Many boulder size fragments, with diameters greater than 1 m, were distributed throughout the spoil, and large voids were encountered during drilling. Seepage water in a roadcut along the western boundary of the mine was identified as AMD from the mine.

In 1989, a total of 35 boreholes (2-20 m deep) were drilled

in spoil at the site. A control hole was drilled in an adjacent, unmined area. Piezometers were installed in the boreholes for measurement of the temperature in mine spoil and for ground-water sampling. Nested gas samplers were installed adjacent to the piezometers in 13 of these boreholes. Gas sampling ports were placed at depths of 0.9 m, 1.5 m, 4.6 m, 6.1 m, 9.1 m, 10.7 m, and deeper (Guo et al. 1994). Pore-gas and water sampling were conducted monthly for 1 and 2 years, respectively.

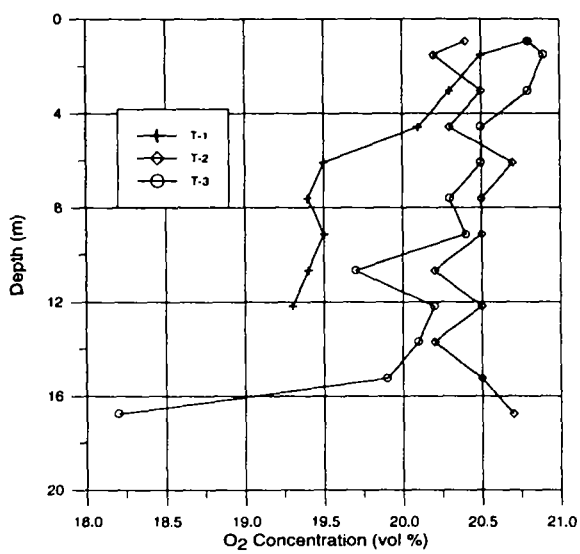

Figure 1. Concentrations O_2 in pore-gas samples from three boreholes (T-1, T-2 and T-3) in unsaturated spoil at mine 1 in Clearfield Co., PA October, 1989.

The chemistry of ground water and seepage water indicates that acid-generating reactions are ongoing at this mine. Water from a perennial seep (N1) had a median pH of 3.4 and had concentrations of sulfate, dissolved iron, dissolved manganese, and acidity, respectively, of 2,200, 45, 52 mg/L, and 290 mg/L as $CaCO_3$ (Durlin and Schaffstall 1993).

Pore-gas O_2 concentrations at mine 1 were found to exceed 18 vol % throughout the spoil during the entire year of monitoring. Several depth profiles of Po_2 are presented in Figure 1. Corresponding low concentrations of CO_2 ranged from 0.03 to 1.2 vol %. The amplitude of seasonal variations in Po_2 and Pco_2 was small.

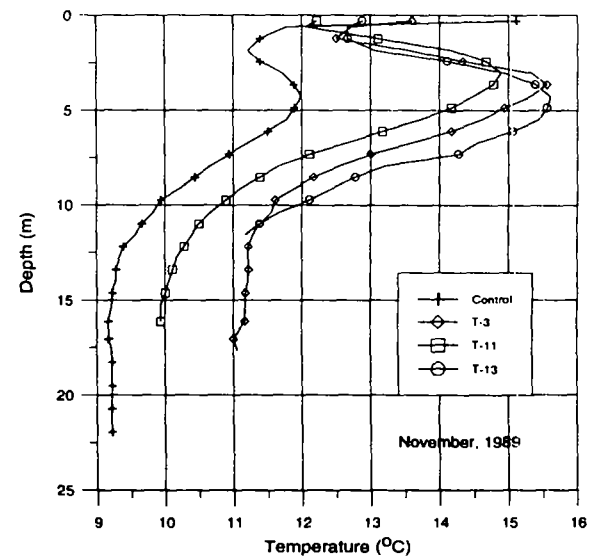

Figure 2. Temperatures in unsaturated spoil at mine 1, November, 1989. Control temperature was measured in a borehole in adjacent unmined area.

Air convection resulting from the temperature gradients induced by the exothermic pyrite-oxidation reactions is the dominant O_2 transport process at this mine (Guo and Parizek 1994). Thermal anomalies were detected in most deep boreholes during the study period. Spoil temperatures were 1-2°C higher than the background temperature measured in the control hole (Figure 2).

Mine 2, Diffusive-Transport Dominant

Mine 2 is a reclaimed surface mine on two adjoining hilltops (41°04'15"N, 79°26'45"W) in southern Clarion County (Cravotta et al. 1994a, b). During 1980-86, the lower Kittanning coal was mined. The lower Kittanning coal and a 1.2-m thick stratum of carbonaceous shale overlying the coal were pyritic, had total sulfur concentrations greater than 2.5 wt % and were laterally continuous across the mine. Spoil at this mine was friable shale with minor siltstone. Spoil fragments generally were less than 10 cm in diameter.

In 1991, a total of 12 clusters of two or more boreholes

were drilled in spoil and underlying bedrock. Several clusters, or monitoring nests, included boreholes for ground-water sampling, pore-water sampling, and pore-gas sampling (Cravotta et al. 1994b). Lysimeters and gas-sampling ports were placed at depths of 1 to 1.5 m, 4 to 4.6 m, 7 to 7.6 m, and 10.7 to 11 m below the spoil surface to enable comparison of chemical concentrations in pore-water and pore-gas samples.

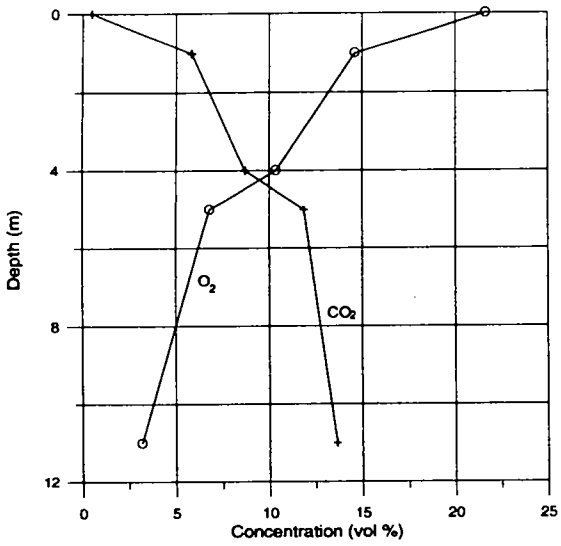

Figure 3. Mean concentrations of O_2 and CO_2 in pore-gas samples from a borehole (Nest 2) in unsaturated spoil at mine 2 in Clarion Co., PA February to December, 1992.

Ground-water samples from the spoil at mine 2 were highly mineralized, with concentrations of sulfate that exceeded 400 mg/L (Lescinsky et al. 1993). Despite this evidence of pyrite oxidation, overall ground-water quality was alkaline because of neutralization by carbonate minerals (mainly siderite) in the saturated zone (Cravotta et al. 1994b). Locally in the unsaturated zone, however, shallow pore water sampled within or near pyritic zones was acidic and had high concentrations of sulfate. For example, in December 1992, pore water sampled from a depth of 4.6 m in the northern part of the mine (nest N2) had pH of 4.0 and concentrations of sulfate, dissolved iron, and dissolved manganese of 5,000, 1.3, and 260 mg/L, respectively.

Data for pore-gas compositions at mine 2 indicate that Po_2 decreases from 21 vol % at the land surface to less than 2 vol % at 10.7 m below the surface, with corresponding increases of Pco_2 with depth (Figure 3). The increase of Pco_2 with depth resulted from the dissolution of carbonate minerals. Because the pore water in spoil is extremely mineralized and dissolved anions are dominated by sulfate, oxidation of organic matter is not considered to be an important control of O_2 and CO_2 at the mine. Cravotta et al. (1994b) concluded that pyrite oxidation rates at mine 2 were controlled by rates of O_2 diffusion and not the amount of pyrite, because concentrations of O_2 declined rapidly at shallow depths in the spoil, independent of pyrite concentrations.

Model Description

A two-dimensional numerical model was developed to investigate the distribution, transport, and reaction of O_2 in unsaturated mine spoil. This model simulates coupled heat transfer and gas flow, dispersive-advective oxygen transport, steady-state ground-water flow, and pyrite oxidation reactions (Guo 1993).

Coupled heat transfer and gas flow in mine spoil are described by the following set of equations:

$$\frac{\partial}{\partial t}(\phi_s \rho_s C_s T + \phi_l \rho_l C_l T + \phi_g \rho_g C_g T) =$$

$$\nabla \cdot (\kappa \nabla T) - \nabla \cdot (\rho_g C_g \vec{v}_g T) - \nabla \cdot (\rho_l C_l \vec{v}_l T) + SS_{energy} \quad (1)$$

$$\phi_g \frac{\partial}{\partial t}(\rho_g) = -\nabla \cdot (\rho_g \vec{v}_g) \quad (2)$$

$$\vec{v}_g = -\frac{k}{\mu_g}(\nabla P + \rho_g \vec{g}) \quad (3)$$

Equation 1 describes heat transfer through solid, liquid and gas phases in spoil; equation 2 represents gas flow through porous medium; and equation 3 is the Darcy law, which links equations 1 and 2. In these equations, ρ is the density $[ML^{-3}]$; C, the heat capacity $[JM^{-1}K^{-1}]$; ϕ, the volume fraction [dimensionless]; T,

temperature [K]; κ, the thermal conductivity of spoil [$JL^{-1}K^{-1}T^{-1}$]; t, time [T]; SS_{energy}, the energy generation and/or consumption rates [$JT^{-1}L^{-3}$] which includes the energy released from pyrite oxidation reactions and the energy transfer due to phase changes; $\vec{v}$, the flow velocity of fluid [LT^{-1}]; g, the gravitational constant [LT^{-2}]; P, the air pressure [$ML^{-1}T^{-2}$]; k, the air permeability [L^2]; and μ, the viscosity of gas [$ML^{-1}T^{-1}$]. Subscripts s, l, and g represent solid, water and gas phases respectively.

To simplify the problem so that it can be solved numerically, several assumptions are made: (1) local thermal equilibrium between all the phases is present at all points, because the temperature difference between the phases generally disappears quickly; (2) thermal conductivity is treated as a constant because the range of temperature changes is relatively small; (3) radiation is negligible; (4) thermal dispersion is insignificant; (5) Darcy's law is valid for gas flow of relatively low velocity, as considered in this study; (6) physical properties of water and solid are constant; (7) gas density is independent of its component composition; (8) air is in equilibrium with pore water and obeys the ideal gas law; and (9) water is assumed to be saturated with respect to the local Po_2 in pore gas in accordance with Henry's law.

Transport of O_2 by dispersion-advection is mathematically described as:

$$\phi_g \frac{\partial C}{\partial t} = \nabla \cdot ([D']\nabla C) - \nabla \cdot (\vec{v}_g C) - \nabla \cdot (\vec{v}_l C) + SS_{oxygen} \quad (4)$$

where C is the concentration of O_2 in air [ML^{-3}]; [D'] is the dispersion coefficient tensor [L^2T^{-1}]; SS_{oxygen} is the source/sink terms including all the O_2 consumed in a unit time inside a block [$MT^{-1}L^{-3}$].

Factors affecting the rate of pyrite oxidation have been expressed numerically by previous investigators using a variety of approaches, of which the shrinking core model is perhaps the most popular one used to simulate pyrite oxidation (Harries 1969; Roman et al. 1974; Madsen et al. 1975; Cathles and Apps 1975; Jaynes et al. 1984a, b). In an unreacted-core model, as used in this study, oxidation proceeds as a narrow front moving into the solid particle and solid reactant concentration remains constant inside the unreacted cores. All the pyrite is assumed to be oxidized in the ash zone outside the unreacted core. Mathematical expressions for these reaction-control mechanisms and for different particle geometry were given by Levenspiel (1972).

Combining of surface-reaction rate and reacted layer diffusion controls gives the reaction rate of a particle containing pyrite:

$$\frac{dN}{dt} = \frac{-1}{\frac{1}{3}\tau_1 N^{-\frac{2}{3}} + 2\tau_2 (N^{-\frac{1}{3}} - 1)} \quad (5)$$

where N is the amount of unreacted pyrite in mole fraction. τ_1 and τ_2 are the time [T] for complete reaction of a particle under surface reaction control and diffusion through ash layer control. Two sets of equations, as defined above, are used to represent the rate of pyrite oxidation by dissolved O_2 and Fe^{3+}, respectively.

Equation 5 defines the reaction rate under non-biological conditions. However, pyrite oxidation can be greatly accelerated by the iron-oxidizing bacterium, *Thiobacillus ferrooxidans* (Singer and Stumm 1970). Increases in pyrite oxidation rates because of biological catalysis are reported to vary from 10 to 50 times (Lorenz and Tarpley, 1963) to 10^6 times (Singer and Stumm, 1970). A simple relation, similar to that given by Jaynes et al. (1984a) is modified and used in this model to account for the activity of iron-oxidizing bacteria.

The mathematical model, described above, was solved using the Newton-Raphson method in a fully-coupled, implicit way based on the block-centered finite difference scheme. Detailed discussion of this model, its numerical procedure, model calibration and testing were presented by Guo (1993).

Results and Discussion

Numerical simulations were performed to evaluate relative effects of advection and diffusion of O_2 in spoil that could cause differences in the spatial distribution of Po_2 at mines 1 and mine 2. One simulation (Test 1) considered effects from both advection and diffusion, and a second test (Test 2) simulated effects from diffusion only, by setting the air permeability to zero. Both tests simulated conditions for 10 years since reclamation and used a generalized geometric configuration (Figure 4). For these tests, 15 cells within the interior of the spoil were simulated to contain 10 wt % pyrite (= 5.3 wt % sulfur), and surrounding cells did not contain any pyrite. This distribution of pyrite represents the selective placement of high-sulfur content overburden into compacted layers above the water table in the backfill. The geometry, initial conditions, and boundary conditions, which were held constant for the tests, are shown in Figure 4. The parameters specified are listed in Table 1.

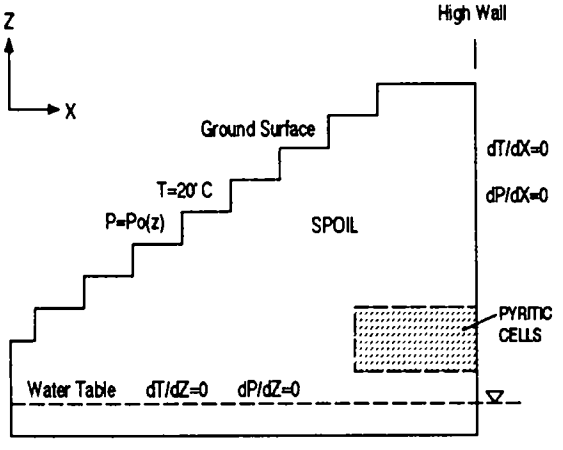

Figure 4. Boundary and initial conditions for numerical simulations. Shaded cells contain 10 wt % pyrite (= 5.3 wt % sulfur) initially; surrounding cells do not contain any pyrite.

For Test 1 where advection is the dominant transport mode, the fraction of pyrite oxidized after 10 years was much greater than that for Test 2 where diffusion is the dominant transport mode (Table 2). This difference in oxidation rate implies that temperature-driven air circulation (Figure 5), induced by pyrite oxidation, is significant and may exacerbate AMD formation because substantial O_2 is available for acid-forming oxidation reactions in mine spoil.

Thermal anomalies at depth as a result of pyrite oxidation are indicated by both simulated temperature profiles (Figure 6) and observed temperature profiles (Figure 2). Thermal anomalies in pyritic waste materials have also been reported (Bennett et al. 1989; Fielder 1989; Alpers and Nordstrom 1991). Higher temperatures are achieved in the more oxygenated environment, simulated by Test 1, because of greater pyrite oxidation and corresponding amounts of heat generation than for Test 2. Higher peak temperatures for simulations than for field observations are an artifact of concentrating pyritic material into one zone, rather than dispersing it throughout the spoil.

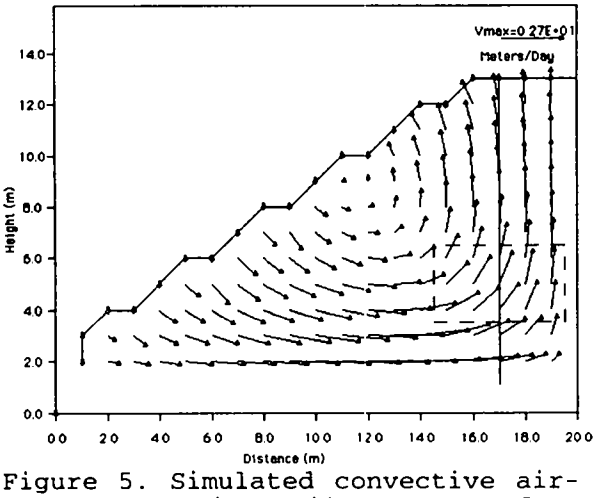

Figure 5. Simulated convective air-flow pattern in spoil 10 years after reclamation (Test 1). Vertical line marks the position of temperature and Po_2 profiles shown in Figures 6-9.

Table 1. Parameters used in numerical simulations.

Bulk density	1650 kg/m^3
Porosity	0.35
Infiltration rate	426.7 mm/yr
Thermal conductivity	2.0 W/m-K
Heat capacity	
Solid	760.0 J/kg-K
Water	4187.0 J/kg-K
Air	1000.0 J/kg-K
Air permeability	0 to 1×10^{-8} m^2
Tortuosity	0.6
Water content	15% by volume
Diffusion coefficient of oxygen in air	1.78×10^{-5} m^2/sec
Longitudinal dispersivity	10.0 m
Transverse dispersivity	1.0 m
Particle size	0.1 cm (radius)
Surface reaction constant	2.5×10^{-5} cm/sec
Diffusion coefficient in reacted zones	1.5×10^{-6} cm^2/sec

Table 2. Summary of Results of Test 1 and Test 2.

Test Number	Test 1	Test 2
Features	Dispersion-Advection	Diffusion Only
Tmax (°C)	27.814	23.006
P_{O_2} min (%)	17.79	4.10
Reacted Pyrite (%)	30.56	12.90

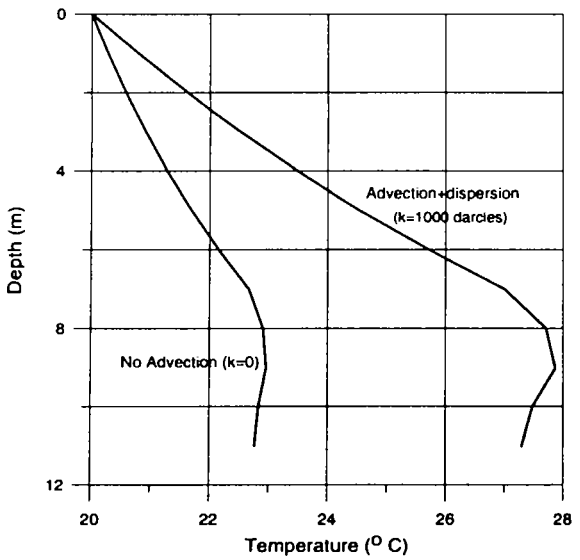

Figure 6. Simulated temperature profiles in unsaturated spoil 10 years after reclamation.

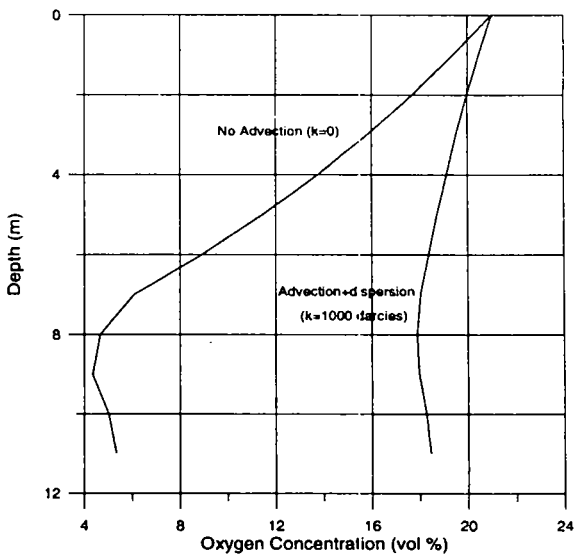

Figure 7. Simulated Po_2 profiles in unsaturated spoil 10 years after reclamation.

Uneven temperatures in mine spoil cause air convection in spoil (Figure 5). If the air flows from the land surface, where Po_2 is high, it brings O_2 into the subsurface. As air convection develops, a greater amount of O_2 is transported into mine spoil and greater rates of pyrite oxidation result, causing a perpetuation of the oxidation cycle. Consequently the temperature will be higher and convective flow is enhanced. However, the system is also controlled by the temperature, because once the temperature reaches about 55°C, bacterial catalysis slows or stops (Cathles 1979) and then the system reaches an equilibrium thermal condition and reaction rate.

Simulated Po_2 profiles for Tests 1 and 2 (Figure 7) are comparable with observed Po_2 profiles for mines 1 and 2, respectively (Figures 1 and 3). The simulated Po_2 profiles indicate that Po_2 remains relatively high throughout the spoil if advective transport takes place, but Po_2 decreases with depth if only diffusive transport takes place.

Results of Test 1 indicate that high Po_2 is attained in spoil when temperature-driven air flow is generated by the heat released from pyrite oxidation. In Test 2 for the simulation of only diffusive transport, Po_2 decreases exponentially from 21 vol % at the land surface to about 4 vol % at 9 m. Once the Po_2 declines to such a low level, the rates of acid-generating reactions are reduced. Results of laboratory experiments indicate the rate of pyrite oxidation reactions decreases when Po_2 is lower than vol 8% (Pugh et al. 1984) to as low as vol 1% (Hammack and Watzlaf 1990). This might be the situation observed in mine 2, where spoil is predominantly shale. The Po_2 remains low while most pore-water samples indicate negligible changes in concentrations of sulfate and iron with increasing depth below about 5 m (Cravotta et al. 1994b). Shale tends to disaggregate and become compacted, and hence shaly spoil (as in mine 2) has relatively low air permeability. In contrast, sandstone spoil (as in mine 1) tends to be very coarse and poorly compacted, which causes it to have higher air permeabilities. Neutralization by calcareous minerals may also contribute to the lower rate of AMD generation in deep spoil at mine 2 than that at mine 1.

If other conditions remain constant (Table 1), the air permeability of spoil is the determining factor for the occurrence of air convection.

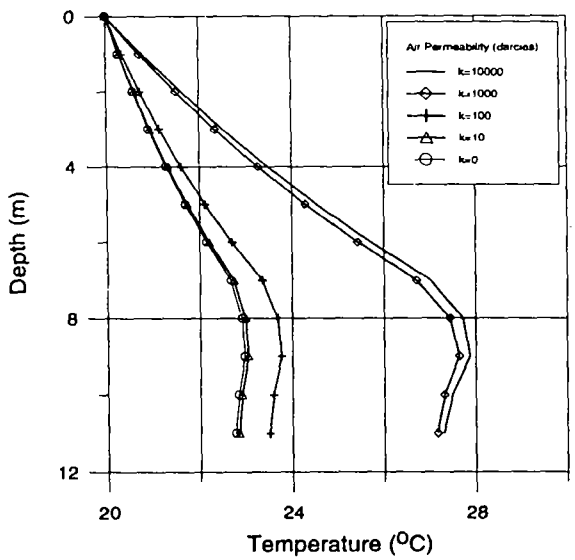

Figure 8. Simulated temperature profiles for different values of air permeability (k) 10 years after reclamation.

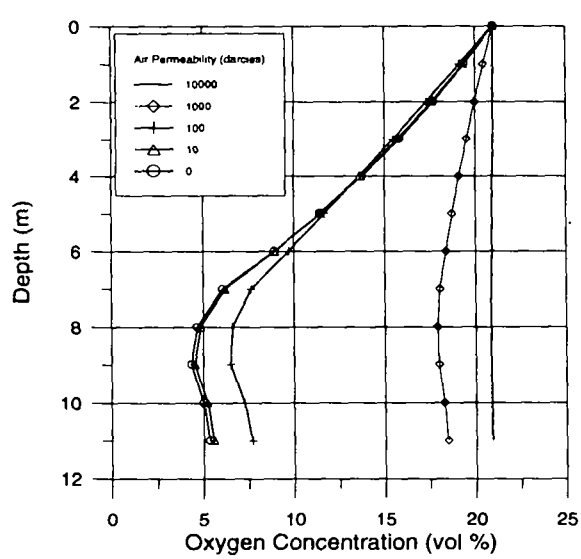

Figure 9. Simulated Po_2 profiles for different values of air permeability (k) 10 years after reclamation.

A series of numerical simulations was made in which only the air permeability was varied from 0 to 10^{-8} m^2. The temperature and Po_2 distributions predicted by these tests are shown in Figures 8 and 9, respectively. It is apparent that diffusion will be the dominant transport process when the air permeability is less than 10^{-10} m^2 (approximately 100 darcies), which is typical for soils. Pantelis and Ritchie (1992) also found that the oxidation rate in a heap with an air permeability of 10^{-10} m^2 is not much faster than that in which diffusion is the only gas transport process, but when the permeability is increased to 10^{-9} m^2, the oxidation rate is substantially faster.

Conclusions

Results of the numerical simulations indicate that differences in the distribution and transport of O_2 in coal-mine spoil can be explained by differences in the air permeability of spoil. Advective transport becomes significant if the air permeability of spoil is greater than 10^{-9} m^2, which is expected for blocky sandstone spoil. In the advective-dominant system, thermally-induced convective air flow, as a consequence of the exothermic oxidation of pyrite, supplies the O_2 to maintain high Po_2 within the deep unsaturated zone. Po_2 changes little with depth if advective transport of O_2 dominates, as at mine 1. In contrast, Po_2 decreases greatly with depth if diffusive transport of O_2 dominates, as in mine 2. Diffusive transport becomes significant if the air permeability of spoil is less than 10^{-10} m^2, which is expected for shaly spoil.

The numerical simulations, combined with field observations of the chemistry of pore-gas and pore-water samples, indicate that unsaturated-zone permeability is a critical variable affecting O_2 flux is spoil and the extent of pyrite oxidation. If the O_2 supply is not restricted, AMD formation can be severe. Thus, engineering practices to reduce O_2 transport into spoil may help to control the formation of AMD. Air permeability and O_2 supply in spoil can be reduced by covering the land surface with a low-permeability layer or by eliminating air-filled voids by compaction or filling with water. Field measurement of unsaturated-zone permeability, temperature, and pore-gas and water chemistry would be helpful for evaluation of the effectiveness of these practices.

Acknowledgments

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PUMPED CO-DISPOSAL OF BLACK COAL WASHERY WASTES IN AUSTRALIA[1]

by

David J. Williams[2]

Abstract. The washing of black coal results in the production of coarse reject and tailings wastes, which historically have been disposed of separately. Loose-dumped coarse reject is highly porous, with largely air-filled voids. Tailings, disposed of as an aqueous slurry, have a high porosity with water-filled voids. By co-disposing coarse reject and tailings, the voids between the coarse reject particles can be filled with tailings. The mixture settles, drains, and gains strength and stiffness rapidly. It occupies less volume than the two waste products disposed of separately, potentially increasing the water return, and also facilitating rehabilitation. This paper briefly describes various co-disposal methods, detailing the behaviour, advantages, disadvantages, and cost implications of the co-disposal of coarse reject and tailings by combined pumping.

Additional Key Words: beaching, coarse reject, costs, tailings.

Introduction

Prior to mining and mineral processing, ore bodies comprise a heterogeneous mixture of different sized particles of different mineralogies. The mining and processing of the ore involves the separation of materials according to their particle size and mineralogy, and results in coarse and fine grained waste materials, which historically have been disposed of separately.

The washing of black coal to meet market specifications results in granular coarse reject (up to 100 mm [4 in] in size) and fine grained tailings (typically silt-sized). The coarse reject is conventionally loose-dumped, while the tailings are conventionally pumped as an aqueous slurry to a storage impoundment. The coarse reject disposed of alone has a high porosity (about 45 %), with largely air-filled voids. The tailings disposed of alone have an even higher porosity (about 55 % after they have settled), with water-filled voids. When coarse reject and tailings are co-disposed, the coarse reject tends to settle in loose contact, with tailings filling the voids between the coarse particles. Co-disposed washery wastes settle, drain, and gain strength and stiffness rapidly. They occupy less volume than the two waste products disposed of separately, potentially increase the water return, and can progressively and more readily be rehabilitated to a high level of future land use.

There are several methods of co-disposing of black coal washery wastes. This paper discusses these briefly, before concentrating on the method most often adopted in Australia, which involves the combined pumping of coarse

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reject and tailings. Beyond the coal mining industry, the co-disposal of overburden waste rock and tailings is being considered at a number of gold mines in Australia.

Methods of Co-Disposal of Coal Mine Washery Wastes

The co-disposal of coal mine washery wastes can be achieved by the following methods.

1. Co-disposal of dewatered tailings filter cake, produced by a belt press filter, and coarse reject is employed at many mines worldwide, including Mount Thorley in the Hunter Valley Coalfields of New South Wales, Australia. The dewatering of the tailings avoids the necessity to recover water after disposal, and facilitates the handling and rehabilitation of the combined wastes. However, the capital and operating costs are high (McKee 1992; in Williams 1992). The main operating costs are attributable to a high flocculant dose (up to 0.5 % by mass of dry tailings; up to 10 times that required for conventional thickening) and labor to maintain the operation of the belt press filters.
2. Combined pumping of a coarse reject/tailings mixture is employed at many coal mines worldwide (several in the USA, three in Queensland, one in New South Wales, and two in Indonesia), and is proposed at numerous existing and future mines. This method is described in detail later in the paper.
3. Pushing coarse reject into uncrusted, wet tailings has been successfully trialed at Ulan Coal Mines in New South Wales, Australia (Williams 1992). There is an optimum time after tailings disposal at which the coarse reject should be added to achieve good mixing. In the field, this optimum time is between one and two days. If the coarse reject is added immediately after tailings disposal, the coarse reject particles will reach the base of the storage before the tailings, and form a segregated layer. If the delay before the addition of coarse reject is too long, the tailings will have settled to the base of the storage, forming a segregated layer. If the tailings are allowed to settle and their surface allowed to desiccate and crust, the coarse reject will be supported by the tailings crust until sufficient coarse reject builds up to punch through it, resulting in bow-wave failure. Operationally, coarse reject must be added continuously, trailing behind the tailings discharge pipe as it is moved around the perimeter of the storage. A stable, trafficable deposit is formed as coarse reject emerges above the tailings surface. Operationally, this method requires a high management/labor input, to move the tailings discharge pipe and to ensure good mixing of the coarse reject. A stable/trafficable surface is possible only if there is sufficient coarse reject for it to rise above the tailings surface (a coarse reject to tailings ratio in excess of about 3 to 1 by dry mass).
4. Pouring tailings slurry over thin (about 0.3 m [1 ft] thick) layers of uncompacted coarse reject has been successfully trialed in South Africa (van Rooyen 1992). This method is effective in sealing coarse reject against possible spontaneous combustion, which is a feature of conventionally loose-dumped South African coarse reject. However, it requires a high management/labor input to place the coarse reject in thin layers and to move the tailings distribution pipes and is suited to only a narrow range of coarse reject to tailings ratios. It is unsuitable for coarse reject to tailings mass ratios of less than about 3 to 1.
5. The co-disposal of thickened tailings (using high-compression or paste thickening) and coarse reject

has been proposed. High-compression thickeners can achieve up to 65 % solids by mass and paste thickening can achieve 65 to 70 % solids, approaching that achieved by belt press filters. This should remove the need to recover water after disposal. The mixing of the thickened tailings and coarse reject could be facilitated by gravity, conveyor, pumping, or truck delivery of the thickened tailings. This method is yet to be trialed, but is being considered by a number of coal and gold mines. In the latter case, overburden waste rock would be co-disposed with thickened tailings.

Co-Disposal by Combined Pumping

Co-disposal by the combined pumping of a mixture of coarse reject and tailings is technically feasible from a pumping point of view, but operational refinements are required.

Pumping Characteristics

The conveyored coarse reject, conventionally thickened tailings (at about 30 % solids by mass), and make-up water are combined in a simple hopper. The wastes are then pumped by a high pressure (up to 3,000 kPa [435 psi]), large bore (approaching the internal diameter of the pipeline), centrifugal gravel pump at about 30 % solids, and at high velocity (2.5 to 4 m/s [8 to 13 ft/s]), to reduce the potential for the pipeline blocking. Pipeline blockages are usually readily cleared by increasing the line pressure. As a result of the relatively low solids concentration, a huge volume of water is required for pumped co-disposal, necessitating efficient water recovery. Particle sizes of up to 100 mm [4 in] (usually limited to about 50 mm [2 in]) can be pumped in a 200 mm [8 in] diameter pipeline, although the pipeline internal diameter should ideally be 4 or 5 times the maximum particle size. Pumping distances in Australia currently vary between 0.3 km [0.2 miles] and 2.5 km [1.6 miles]. Pumping distances greater than about 5 km [3 miles] may well make combined pumping uneconomic on the basis of pipeline capital and operating costs. Pumping heads currently range up to about 10 m [33 ft]. To maintain stable pumping conditions, some fines are required. Coarse reject to tailings mass ratios of between 1:1 and 6:1 have been successfully pumped. Pump and pipeline wear is high.

Beaching Behaviour

Co-disposed wastes beach at a slope of up to 1 (vertical) in 10 (horizontal), compared with a tailings only beach slope of about 1 in 100, and co-disposed wastes can be pushed up to slope angles approaching the angle of repose of the coarse reject material. On co-disposal, hydraulic sorting of the particles occurs, according to their particle size and specific gravity. These two effects may counteract, with the result that there may be little change in particle size down the co-disposal beach, with high specific gravity particles settling close to the discharge point and low specific gravity (coal-rich) particles settling at the far end of the co-disposal beach (Williams 1992).

Some wash out of fines (up to two-thirds of the tailings, the proportion increasing with decreasing particle size) is inevitable on pumped co-disposal. This results, in part, from the relatively low solids concentration and high velocity at which the washery wastes are pumped to limit the potential for pipeline blockages. Fines wash out increases with the inevitable gap-grading between the coarse reject and tailings, and with increasing plateness of the coarse particles. Crushing of the coarse reject, which would reduce the gap-grading and plateness of the particles, may be of benefit, provided that it does not substantially increase the proportion of fines in the mixture. For coarse reject to tailings ratios of less than 3:1, a mixture with a ratio

of about 3:1 will form on the co-disposal beach, with the residual fines washing out to form a tailings-like beach. To minimise the impact of fines wash out, co-disposal directed upslope is preferable to downslope co-disposal. Upslope co-disposal retards the flow of fines, allows them to be covered by later co-disposal, and avoids the construction of a major downstream containment. Water return of 75 to 90 % is possible, the percentage decreasing with increasing initial solids concentration.

The co-disposal mixture has a permeability (typically about 0.0001 m/s [0.0003 ft/s]; Williams and Kuganathan 1993) mid-way between that of coarse reject only and tailings only, at about 1,000 times that of the silt-sized tailings alone. Drainage paths exist around the coarse reject particles in loose contact, against which the angular tailings particles form a loose packing. The flow on the co-disposal beach is about 100 mm [4 in] deep, making the surface almost immediately trafficable. The co-disposed mixture rapidly achieves a dry density of up to 1.5 tonne/cubic m [94 lb/cubic ft], at a gravimetric moisture content as low as 15 % and a porosity as low as 30 %. The overall physical characteristics of coarse reject disposed alone, tailings disposed alone, and co-disposed wastes are compared in Table 1.

Table 1: Overall dry density, moisture content and porosity of coarse reject only, tailings only, and co-disposed wastes.

MATERIAL	DRY DENSITY (t/m^3) [pcf]	MOISTURE CONTENT (%)	POROSITY (%)
Coarse reject only	1.2 [75]	5-10	45
Tailings only	0.8 [50]	70	55
Co-disposed	1.5 [94]	35-40	35-40

Relative Ease of Pumped Co-Disposal

The relative ease of pumped co-disposal depends on the following parameters.

1. Coarse reject to tailings mass ratio: Generally, the higher the ratio the better, since it reduces the total fines wash-out and results in a steeper co-disposal beach.
2. Particle size distribution of mixture: There is, inevitably, little overlap between the coarse reject and tailings sizes. The more pronounced the gap, the less efficient is the trapping of fines.
3. Coarse reject particle shape: Rounded particles are more readily transported and pack and trap fines better than platey particles, but give a flatter beach slope.
4. % solids: Provided pipeline blockages are avoided, the higher the solids concentration the better. This results in a steeper beach slope and a reduced volume of water in circulation.
5. Flow velocity: Provided pipeline blockages are avoided, the lower the flow velocity the better. This results in a reduced wash out of fines.
6. Geometry of deposition: A steep initial beach slope results from burial of the discharge, until a blow-out occurs.
7. Level of decant pond: This affects water clarity and discharge into water will slow the segregated fines.
8. Direction of discharge: Upslope pumped co-disposal is far preferable to downslope, preventing the segregated fines from running and allowing them to be progressively covered. However, it requires more management/labor input.

Potential Problems with Pumped Co-Disposal

The potential problems with pumped co-disposal include the following.

1. Excessive pump wear: This can be largely overcome by using a larger pump bore, several times the size of the largest particle and similar to that of the pipe.
2. Excessive pipe wear: Pipe wear will be highest where the line pressure is greatest (near the pump). High density polyethylene (HDPE) and mild steel pipes wear excessively, but may be appropriate for short pumping distances (< 0.5 km [0.3 miles]). Ceramic- and polyurethane-lined pipes perform well, at a higher capital cost, while the more brittle basalt-lined pipes perform less well. A 10 mm thick lining may be optimal.
3. Pipeline blockages: High pressure (3,000 kPa [435 psi]) pumps largely overcome pipeline blockages.
4. Long pumping distances: As a rule-of-thumb, an additional pump should be added (preferably along the line, although this necessitates remote power) for every km length of pipeline.
5. Variable coarse reject/tailings feed: High pressure pumps provide a robust system, capable of handling variations in the coarse reject/tailings feed.
6. Moving discharge point(s): Moving the discharge point or points is recommended to minimise rehandling of the mixture and for better control. It can be done by dragging the pipeline or, preferably, by using a number of discharges in rotation and adding lengths of pipe as required. The optimum number of discharges is dependent on the storage geometry.
7. Water recovery and recirculation: Efficient water recovery for recirculation is vital, requiring substantial management/labour input. The clarity of the return water is relatively unimportant.

Co-Disposal Storages

New mines will likely co-dispose above-ground, either upslope or into paddocks, while existing mines may have dis-used ramps or pits available. Upslope co-disposal above-ground offers the potential to store up to five times the volume of washery wastes stored by downslope co-disposal, for the same height of downstream containment. Co-disposal offers a more efficient use of available in-pit storage volume than the separate disposal of coarse reject and tailings. Co-disposal into ramps should preferably be upslope, to maximise the available storage volume, reduce the impact of fines segregation, and increase the stability of the embankment separating the ramp from the pit.

Water Recovery from Pumped Co-Disposal

Co-disposal by combined pumping requires the recovery and circulation of large volumes of water. It is therefore vital that water recovery be efficient. Water clarity is less important, since the water will be recirculated with the co-disposed wastes. Various methods have been or can be used to recover and recirculate water from the co-disposed wastes. For in-pit and downslope co-disposal, pumping from a conventional decant pond is appropriate. For upslope co-disposal, underdrainage is required in addition to pumping from an upslope decant pond.

Costs of Pumped Co-Disposal

The costs of pumped co-disposal are offset by the elimination of tailings dams and coarse reject dumps. Jeebropilly Colliery, in the Ipswich Coalfields of south eastern Queensland, Australia, operates pumped co-disposal for about 15 % of the cost of the previous separate disposal of tailings to a dis-used pit and coarse reject to a dump. However, Jeebropilly has dis-used pits available for co-disposal, and experienced difficulties with the dumping of their highly degradable, clay-rich coarse reject. As a guide, pumped co-disposal should cost of the order of half the cost of conventional separate disposal, depending on a range

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of site specific and other factors. For both new and existing mines, the costs of pumped co-disposal must be weighed against the costs of conventional separate disposal.

Cost items associated with conventional coarse reject dumps include the following.

- Coarse reject hopper.
- Truck capital, operating, maintenance, and replacement costs.
- Land cost.
- Construction and maintenance of haul roads.
- Collection of seepage from the dump, and treatment if necessary.
- Rehabilitation costs. To reshape and cover a coarse reject dump costs up to $A 75,000/ha [$US 22,500/acre], but the costs can be greatly reduced by progressive reshaping and rehabilitation.

Cost items associated with conventional tailings dams include the following.

- Thickener capital, operating, and maintenance costs.
- Tailings dam construction and raising.
- Land cost.
- Capital, operating, and maintenance costs of pumps and pipeline(s).
- Water recovery and return.
- Rehabilitation costs. To cover crusted tailings costs in the range from $A 40,000 to $A 80,000/ha [$US 12,000 to $US 24,000/acre].
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Cost items associated with pumped co-disposal include the following.

- Tailings thickener capital, operating, and maintenance costs. Where pumped co-disposal has been adopted, a conventional tailings thickener has been retained. There is scope for some of the tailings to by-pass the thickener, avoiding the need to add make-up water, with consequent cost savings.
- Construction of a hopper for combining the coarse reject, thickened tailings, and any make-up water. This can be a modest, low-cost facility.
- Land cost. This will be less than that required for the separate disposal of coarse reject and tailings, because of the reduced storage volume occupied by co-disposal.
- Capital, operating, and maintenance costs of pumps and pipeline(s). These will be substantially higher than the corresponding costs for tailings only, and may be the largest cost item. A low-cost pipeline may be adopted for emergency purposes. Instrumentation should be installed to warn of possible pipeline blockages.
- Operating costs associated with moving the co-disposal discharge.
- Costs of recovery and recirculation of water, preferably involving a separate water storage dam.
- Operating costs associated with re-shaping the co-disposal deposit.
- Rehabilitation costs. These will be far lower than those for separate coarse reject dumps and tailings dams. A nominal cover after progressive re-shaping may be all that is required.

For existing mines, a number of situations offer opportunities to switch from conventional separate disposal to pumped co-disposal. These include an exhausted coarse reject dump and/or tailings dam, coarse reject haul truck(s) requiring replacement, and expansion of the mine, and mining voids becoming available. For new mines, pumped co-disposal can take advantage of not being locked into existing plant and methods, a comprehensive cost-benefit analysis, flexibility in the siting of waste handling plant and waste storages, and rehabilitation requirements.

Rehabilitation of Pumped Co-Disposal

Pumped co-disposal offers the potential for progressive, ready rehabilitation to a high level of future use. The co-disposal discharge

should be shifted to build up the desired final land form, by regularly moving a single discharge, or using a number of discharges in rotation, extending out over the deposit as it builds up. Upslope co-disposal lends itself to progressive rehabilitation, since it minimises the impact of fines segregation, maximises the use of hydraulic placement, and allows the downstream slope to be built up, using co-disposed material, to the desired final slope angle.

Concluding Comments

There are several different approaches for the disposal of black coal mine washery wastes. The pumped co-disposal of the combined coarse reject and tailings is an economical alternative to the conventional separate disposal of the two waste products, which has been adopted at many coal mines worldwide and is being considered at numerous other mines. It will find application at existing and new mines where its potential benefits are recognised and can be realised. Pumped co-disposal has a number of problems including pipe wear and the segregation of fines on the beach. However, an appreciation of these problems in the design and operation of pumped co-disposal will minimise their impacts.

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SITE CHARACTERIZATION AND CONTAINMENT/REMEDIATION OF ACID MINE DRAINAGE AT AN ABANDONED MINE WASTE DUMP

by

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Abstract. This paper focuses on the preliminary results of laboratory tests to evaluate a new suspension grout consisting of a mixture of a naturally occurring lignite coal based wax "montan wax", sodium bentonite "pure gold grout", and water. The test program assesses the suitability of the grout for creating subsurface containment barriers in coal waste dump sites for acid mine seepage control to surface and ground waters. The laboratory activities evaluated the reduction in permeability that could be achieved in a coal waste dump site under optimum conditions and the compatibility of the grout with representative waste from the test site. Information on geological, geochemical and geophysical about the test site is presented. Laboratory formulation of the grout is complete and simulation of field condition is in progress. Pregrout geophysical surveys for determination of hydrogeologic conditions at the site are also completed. Based on geophysical surveys, a grout curtain is proposed which will consist of two rows of grout placement holes in an array across the seepage area toward Belt Creek in Montana. Post-grout geophysical survey will be carried out immediately after grouting work. Performance of the grout curtain will be monitored by collection of water samples from monitoring wells in the Belt Creek and seepage area.

Additional Key Words: grouting, sealing, barriers, seepage control, grout formulation, montan wax, bentonite, permeability, compatibility tests, coal waste dump site, ground water contamination, geophysical, geochemical, etc.

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Introduction

Acid mine drainage (AMD) from past mining practices is a major source of contamination of rivers, streams, and aquifers throughout the United States. The migration of surface and groundwater through active and abandoned mine workings and tailing piles, combined with bacterial oxidation of sulfides, is the major cause of the formation of AMD. In the eastern United States, there are thousands of abandoned surface and underground, high-sulfur, coal mines. As a result, AMD is a significant source of pollution of surface and ground water. However, in the western United States, AMD is primarily the result of hard-rock mining. For example, the state of Colorado has over 20,000 abandoned mines, some of which are sources of contamination of rivers and streams.

Montan Wax is essentially a mixture of waxy constituents consisting of monohydric alchohol esters and high-molecular-weight acids with some resinous and asphaltic material.

The idea of using Montan wax to construct containment barriers began because of serious environmental problems in Bitterfeld, Germany. MIBRAG is a large industrial company with headquarters in Bitterfeld that developed the montan wax grout. MIBRAG was responsible for lignite coal mining operations in the area. Montan wax deposits are present in these coal fields.

In the Bitterfeld area, there are several industrial waste disposal pits where leachates are entering the local aquifer, contaminating the drinking-water supply. (Voss, et. al., 1994). MIBRAG researchers began testing the feasibility of constructing containment barriers with montan wax grouts in several field demonstration projects at the disposal sites. The sites contained unconsolidated soils with relatively high rates of hydraulic conductivity (e.g., 10^{-3} cm/s). After permeating the soils with a mixture of montan wax, water, and bentonite, the conductivity of the soil was reduced by as much as five orders of magnitude (Caldonazzi, et. al., 1993).

Based on the positive results of the field demonstration projects in Germany, the U.S. Department of Energy's (DOE) Office of Technology Development (OTD) is evaluating the montan-wax emulsion as a containment material for remediating contaminated DOE sites. Hydraulic conductivity tests have been conducted on soil specimens from the Hanford and Sandia sites. Laboratory studies, namely hydraulic conductivity, penetration rate, waterproofing characteristics, chemical compatibility, and durability have shown that these soils are groutable with montan wax emulsions. After permeating the soils with straight montan wax (no added bentonite), the hydraulic conductivity of soil has been reduced by as much as five orders of magnitude. (Voss, et al., 1993)

Chemistry of Montan Wax

Montan wax is found in lignite which is not far advanced into the carbonization process. The resistant fossil plant leaf wax has remained uncoalified as a result. It is thought that montan wax results from fossil palms similar to modern carnauba wax palms. Montan wax is recovered by pulverizing the lignite and placing it in a solvent extractor for removal of the plant wax. Montan wax is a complicated mixture of primarily wax, resin, and asphaltene - like substances. The typical dark color of the base montan wax is due to the small percentage of asphaltene - like substances. Montan wax has a distribution of C24 - C32 carbon chain esters of long-chained acids and long-chained alcohols. Hydrocarbons, free alcohols, and ketones exist only in small quantities.

Because the wax is extracted from the mining of lignite, it is classified as a mineral wax. But, it is a fossil plant wax, not a petroleum wax, and the properties are comparable to natural plant waxes. The asphaltene-like material contained in the wax is so named because it is black; it does not have the same chemistry as asphalt.

Montan waxes are used for their hydrophobic and lubricating properties as well as their hardness and high melting point. Montan wax has a natural acid number and a long chain hydrophobic molecule with a short hydrophilic section. For water-proofing of aggregates, the hydrophilic section wets the aggregates and the hydrophobic section provides the water repellency. This property is not found in paraffin waxes.

Montan wax grout in comparison to conventional cement grouts offers superior sealing characteristics because ground water does not wash it out and pore cavities are sealed by wax deposition. If large pore sizes exist in the rock mass, bentonite is added to the wax to improve sealing performance. Montan wax is not brittle and has non-desiccant characteristics. It is environmentally safe and chemically and micro-biologically stable. (Karabon, 1978)

General Characteristics and Uses of Montan Waxes

Montan wax is composed of hard, friable, lustrous materials ranging in color from black to amber. The natural ester wax is derived from fossilized plants and is a complex mixture of components. Montan wax products in the United States are named ALPCO by the producer of crude montan wax. ALPCO produces several waxes. ALPCO 16, 1600, and 1650 are suitable for environmental applications. For formulations of montan wax and bentonite, only ALPCO 16 is being used. ALPCO 16 is a black, hard and lustrous wax. Its moderate acid number makes it easily saponified and suitable for emulsions for mold releases, polishes, water-proofing applications, asphalt emulsions, cable dressing, lubricants and grease. ALPCO 16 is a modifier for asphalts, temporary soft coatings,

sealants, and undercoating materials.

ALPCO 16 has a specific gravity of 1.01 at 25 C°, bulk density from $4.9x10^2$ to $5.3x10^2$ kilograms per cubic meter, melting point from 81 to 85 C°, and flash point of 345 C°. All ALPCO waxes are resistant to oxidation but will biodegrade. All ALPCO waxes are non zardous by U.S. Department of Transportation regulations.

Potential Benefits of Montan Wax Grout as a Sealing Material

When injected into the fractured rock mass, montan wax emulsions containing polymer silicate creates an elastic block which firmly bonds with the pore walls. The sealing characteristics of montan wax grout can be adjusted for mobility, precipitation, and configured for the individual geological conditions. An example of this type of application is sealing of the drill hole walls by introducing montan wax emulsion into the drilling fluid containing polymer silicate. As a result, the introduction of heavy metals into the drill hole while drilling is stopped. Montan wax emulsion, containing bentonite, and containing large pore openings is also used for soil stabilization. This application improves water isolation properties of the soil mass even in high permeability environments.

The properly designed montan wax grouts are insoluble in ground water, neutral, and biologically nondegradable in the saturated fractured rock mass environment. The montan wax does not stratify during hardening nor stabilize during injection and it has excellent flow-ability characteristics. It develops strength quickly after injection and it is used for soil stabilization of hazardous waste sites in the Lausitz region, Bad Erna, Germany. Sealing characteristics of montan wax placed within fractures for isolating ground waters in the central German mining area of Lochau and Meserburg - OST have been documented, (Voss, et al., 1993).

The montan wax grouting technique is used for encapsulation of contamination sources or to prepare it for a subsequent restoration purpose. For control of contaminant migration in lateral, horizontal and inclined directions, injection of montan wax is practiced in the brown coal fields of the former East Germany.

General Characteristics of Pure Gold Grout (Bentonite Clay Grout)

Pure Gold Grout (PGG) is a specifically formulated, high solids, sodium bentonite grout. PGG is an easy mixing, single component, organic free bentonite clay grout. It was engineered to form a contaminant resistant seal without affecting ground water chemistry. PGG is a technically superior replacement for traditional cement grouts. It will form a low permeability and a flexible clay seal which impedes interaquifer fluid movement and infiltration of surface contaminants into the borehole.

PGG is commonly used in sealing boreholes or the annular space of monitoring wells to control contaminant infiltration and preserve ambient groundwater quality. PGG's pH ranges from 5 to 6-standard units. PGG is contaminant free and chemically stable. PGG has a low permeability with test results ranging from $1x10^{-7}$ to $1x10^{-9}$ cm/s against permeants ranging from a pH of 2 to 12. PGG has a low formation loss, and minimum flow through highly permeable soils. It remains flexible, maintains putty-like consistency over time and will re-hydrate. PGG is mixed with a grout mixer and trimmed with a positive displacement pump. It can be mixed in a slurry up to 30% solids, remains placeable for up to two hours, and sets in eight hours. PGG is manufactured by American Colloid Company. (CETCO, 1994).

Site Characterization

Site characterization for planning the grout test involved drilling, coring and mapping, the installation of monitoring wells, the collection of geochemical data from the monitoring wells, diversion ditching and performing geophysical surveys.

Site Survey

The field site selected for this study is the Anaconda Coal Mine's reclaimed coal processing waste dump near Belt, MT. AMD is a major source of contamination of Belt Creek and the unconfined aquifers at the site. The mine was an active underground coal mine from the 1890s to the 1920s, and at one time, was the largest coal mine in Montana, with daily production of 2,500 tons. The mined coal was transferred about 914 m from the adit to a coal processing plant. Washed coal was coked and coal waste dumped on the banks of Belt Creek. In the 1980s, $6.5x10^5 m^2$ of the site was reclaimed by the Abandoned Mine Reclamation Bureau, Department of State Lands, State of Montana.

The 8,100 m^2 testing site is bounded by Belt Creek, an all-weather road, and a diversion ditch which discharges flow of about 378 lpm from the mine adit into Belt Creek. AMD from the diversion ditch flows through the mine waste dump and seeps to Belt Creek. Aquatic life has disappeared up to 8 kilometers down the creek. The site is presently used by the Department of Transportation, State of Montana to store railroad ties and gravel for making asphalt. The site is leveled, and a foot of top soil covers the area.

Drilling

Six hydrologic monitoring wells (MW) have been drilled using a hollow stem auger drill (Figure 1). The wells were all drilled to bed-rock and range from 4.2 to 7.2 m (Table 1). During drilling, subsurface materials were characterized by examining materials coming up the outside of the auger flights. Split spoon sampling was not conducted during drilling. Materials encountered were in no way uniform throughout the site and ranged from gravel to clay to

coal to a reddish "muddy" material which may have been decomposed, liquified scoria and mine waste. Drilling at every location except MW-4 was stopped when augers began scraping on a compact solid surface at depth. It was postulated that this solid surface may be cemented gravel underlying the coal wastes. The cemented gravel could be caused by precipitation of minerals due to pH changes in the water migrating through the coal wastes into native materials. A surface exposure of cemented gravel is located adjacent to Belt Creek a short distance upstream from the diversion ditch discharge point. At MW-4 gravel, clay and silt were encountered at depth rather than the "cemented gravel". Once the drilling was complete, groundwater monitoring wells were installed with 10 cm diameter 40 PVC pipe. Three meter screened sections were used in each well to intercept the groundwater table. Screened sections were filter packed with sand. Above the sand, bentonite chips were used to fill the remaining annular space. A flushmount with concrete seal was then installed at the surface. After installation activities, the wells were developed by blowing water and sediment from the well using pressurized air from the drill rig and then surging and bailing with a disposable bailer to purge the wells of fine material (Tetra Tech, Inc., 1995). This work was sponsored by the Department of Environmental Quality, Reclamation Division, State of Montana. Drilling of monitoring wells was contracted to Boland Drilling, Great Falls, MT., and hole logging was done by the Tetra Tech, Inc. Helena, MT.

Geochemical Data

Water samples collected from monitoring wells were analyzed and scanned for their metal contents. Conductivity ranged from 8,200 (MW-4) to 26,000 (MW-3) micro mhos/cm and pH ranged from 1.9 (MW-3) to 4.0 (MW-4). At about 6 meters from MW-2 (Figure 1) on the bank of Belt Creek, decolorization and bubbling of water was in progress and a water sample was collected from this location. Conductivity and pH measured for this location were 566 micromohs/cm and 4.7 respectively. An additional water sample was taken from surface diversion ditch and it had conductivity and pH of 2,800 micromohs/cm and 2.4 respectively. Water samples taken from Belt Creek, upstream from the test site have conductivity and pH content of 415 micromohs/cm and 7.4 respectively. Water analysis for the contaminated samples show that pH was quite low and dissolved solids moderately high. The dissolved solids are largely composed of sulfates. Iron, magnesium and calcium are the most common cations detected, but there are probably other cations present that were not tested for. The main concern is the low pH but there were small amounts of toxic metals, mainly lead and zinc. Lead and zinc metal ions were detected in all of monitoring wells above limits for drinking water established by Environmental Protection Agency in mg/l.

Compatibility Test Results

Chemical compatibility testing was conducted in order to provide a preliminary indication of the performance of the grout when exposed to a variety of metals that are representative of typical AMD constituents present at the coal waste dump test site. The analysis of water samples is not complete enough to simulate exactly the potential effects on the grout; however, the simulated solution's pH is 2, which will be a tougher test for the grout, but the total dissolved solids will have about 2,000 ppm, which would make an easier test for the grout.

Grout Cost Comparison

Cost estimation for grouting projects is an art and much depends on site-specific conditions, materials availability, grouting contractors mobility and experience. However, the cost data, presented in Table 2, are for material cost comparisons purpose for the various grouts. These costs include the cost of material and shipment to Denver only, and exclude labor cost for preparation of grouts.

Portland Cement Type II grout is the cheapest one but it has its own limitations for environmental applications. The second cheapest grout is montan wax and bentonite clay-based (Pure Gold Grout) materials with unique properties. These materials may have a wide application in the environmental field for containment and stabilization of abandoned coal, uranium and hard-rock mines. The most expensive grouting materials are microfine cement and sodium silicate. These materials are routinely and successfully used at civil engineering projects, however, economically they would have limited application in large volume mining operations for AMD control.

Laboratory Formulation

American Colloid Company has been active in formulation of montan wax and bentonite grout for acid mine seepage control at the test site. The first phase of this effort focused on formulation of grout in the laboratory; and the second phase of this effort will consist of field injection of the formulated grout in the Anaconda Coal Mine waste dump site near Belt, MT., where performance of injected grout will be determined by post-grout geophysical surveys and hydrologic monitoring.

Laboratory formulation of montan wax and bentonite grout is complete. A total of nineteen formulated samples of montan wax and bentonite (Pure Gold Grout) were prepared in the laboratory and all of these samples were subjected to standard cone penetration testings. Pure Gold Grout (PGG) is the type of bentonite grout used for laboratory testing. PGG is used as the standard basis for core penetration testing of all formulated samples. Two formulated samples out of nineteen were closely comparable to PGG's cone penetration results. The results of these two formulated samples are presented in Table 3.

The two formulated samples are composed of 1) 1% montan wax, 30% PGG, and 69% deionized water, 2) 5% montan wax, 25% PGG, and 70% deionized water. Permeability studies resulted in the measurement of hydraulic conductivity of 2.04×10^{-8} cm/s after 142 days for 1% Montan wax mixture.

The permeability measured for 5% Montan wax was 1.12×10^{-7} cm/s after two weeks. For both grout formulations, falling head permeability tests were used and the permeant had a pH of 2. Furthermore, 1% montan wax, PGG 30%, and 69% deionized water formulated grout was mixed with augered coal waste dump samples from the testing site at a ratio of 1:1. This mixture is permeated with tap water in the laboratory. The average hydraulic conductivity measured after 56 days was 2.71×10^{-8} cm/s. This test is still in progress. This is considerably less than the level of 10^{-6} cm/s specified by EPA for use in liner systems. Nevertheless, the reduction in permeability afforded by the montan wax based grout is significant and it may be a useful barrier material for applications where the containment time is limited, or where a lower degree of containment can be tolerated.

Geophysical Surveys

AMD from the sealed adit of the abandoned Anaconda coal mine is directed east through a drainage pipe downslope to a diversion ditch which brings AMD from other mines upstream. Flow is then directed northeasterly and discharges into Belt Creek, which flows in a northwesterly direction at this location. The diversion ditch is constructed on top of wastes from coal processing and angles toward Belt Creek where it discharges over a riprap embankment to prevent erosion. Consequently, a positive hydraulic gradient is established along the ditch which is approximately normal to the topographic gradient of the old surface drainage.

The seepage of AMD that has been observed in Belt Creek, upstream from the discharge point of the diversion ditch, thus could be due to leakage from the ditch, or the near surface groundwater moving through the waste materials, or a combination of both. Geophysical surveys were performed on the site to characterize the hydrology and attempt to delineate the primary source of the AMD seep into the creek.

Electromagnetic (EM34) surveys were performed to map the subsurface conductivity of the site, and self-potential (SP) surveys were made to determine direction of significant ground water movement in the area. Additionally, a ground-penetrating radar method was evaluated as a potential characterization technique. The EM34 data were taken using the 10m coil spacing in the horizontal and vertical dipole configurations to investigate both the fill material, which varied in thickness from about 3m to 8m, and the buried geologic formation beneath the fill. The horizontal dipole configuration has a depth of investigation up to approximately 7.5m, whereas the

vertical dipole configuration responds to conductive layers up to 15m depths.

Geophysical surveys were made in August and again in October, 1995. Surface conditions at the site were significantly different between the two dates. These conditions affected the SP surveys with excessive noise because of the electrode coupling problems. Also, the surface of the site has been capped, revegetated, and used by the County for storage of gravel and asphalt for road construction, compounding the coupling problem. EM34 surveys, however were highly repeatable, indicating little change in subsurface groundwater conditions between the two survey dates.

Figures 2, 3, and 4 illustrate the horizontal dipole, vertical dipole, and SP data, respectively, from the later surveys in October, 1995. Included on the maps are the survey line data points (small circles), sample well locations, and the approximate boundary of the property (heavy solid line). The boundary line is a barbed wire fence which affects the response of the EM34, and data near this boundary is not diagnostic; however, in the central portion of the site on the horizontal dipole data map (Figure 2), a high conductivity ridge originating at the diversion ditch between 50-60m, and oriented toward the approximate location of the seep into Belt Creek, is apparent. The vertical dipole data (Figure 3), which is responding proportionately greater to deeper ground conductivity, does not show a prominent high anomaly at this location, and the SP map (Figure 4) is also relatively non-descript.

Interpretation of these geophysical surveys indicates potential leakage from the central portion of the diversion ditch, accumulation and very slow percolation of the already acidic water through the coal processing wastes just above the bedrock surface (which is an impermeable cemented conglomerate), dissolution of metals in the wastes, and seepage below the water level at the bank of Belt Creek. The self potential data is non-diagnostic because the rate of flow and hydraulic gradient (pressure) is minimal, generating no streaming current. Horizontal dipole survey data indicate the primary water movement is at or above the bedrock layer, and the vertical dipole data indicates little likelihood of communication with any deeper groundwater regime.

The conclusion from the interpretation of the geophysical surveys is that the objective of the project, i.e., evaluating the effectiveness of a montan wax bentonite grout curtain (Figure 5), can be achieved by configuring the placement holes in an array across the channel of acid mine drainage and monitoring the effect on the location and rate of seepage into Belt Creek.

Summary

The laboratory results of the preliminary evaluation of montan wax and bentonite (Pure Gold Grout) for constructing

containment barriers at an unconsolidated coal waste dump site for seepage control is promising. Permeability testing for a formulated 1% montan wax, 30% PGG, and 69% dionized water had average hydraulic conductivity of 2.04×10^{-8} cm/s after 142 days. The permeant had pH of 2. This formulated grout is mixed with an augered coal waste dump sample in the laboratory at a ratio of 1:1. This mixture was permeated with tap water. The average hydraulic conductivity measured was 2.71×10^{-8} cm/s after 56 days. This test will be repeated soon and permeance will be replaced with an acid solution of pH2. Laboratory measured hydraulic conductivities exceed the maximum level of 10^{-6} cm/s specified by EPA for use in liner systems. Geophysical surveys were performed at the site to characterize the hydrology in an attempt to delineate the primary source of the AMD seep into the creek. Interpretation of these geophysical surveys indicates potential leakage from the central portion of the diversion ditch, accumulation and very slow percolation of the acidic water through the coal waste dump just above the bed rock surface, dissolution of metals in the wastes, and seepage below the water level at the bank of Belt Creek.

In summary, the interpretation of geophysical surveys have resulted in configuring the grout placement holes in an array across the acid mine drainage channel. Grouting work will be completed soon and a post-grout geophysical survey will be carried out for determination of grout effectiveness. Finally, performance of grouting work will be monitored by collecting water samples from wells and by monitoring the rate of seepage into Belt Creek.

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TABLE 1. LABORATORY FORMULATION RESULTS FOR MONTAN WAX(MW) AND PURE GOLD GROUT(PPG)

Grout Type	PGG gram	MW ml	Deionized water ml	Cone Penetration** 15 min mm	24 hr mm	Hydraulic Conductivity cm/s
PGG Grout Standard	300		700	>100	24	1.0×10^{-7} - 1.0×10^{-6}
MW/PGG 1	300	10	690	>100	22	2.04×10^{-8} after 142 days*
MW/PGG 2	250	50	700	>100	36	1.12×10^{-8} after 14 days*

*Simulated site water sample had a pH 2.0 Falling Head Permeability Test were based on Rigid Wall Method (ASTM: D2434-68)
**Standard Test Method for Cone Penetration of Lubricating Grease (ASTM: D217-86)

TABLE 2. GROUT COST ESTIMATE*

Grout Type	$/m³
Montan Wax (MW) and Pure Gold Grout (PGG)**	
1: 1% MW, 30% PGG, 69% Deionized Water	120
2: 5% MW, 25% PGG, 70% Deionized Water	90
Portland Cement Grout Type II, mixed at ratio of 1 to 1	70
Microfine Cement, mixed at ratio of 1 to 1	1,170
Sodium Silicate	1,250

*These cost estimates include cost of material and shipment to Denver and excludes labor costs for preparation of grouts.
**Pure Gold Grout is a registered trade mark of the American Colloid Company.

TABLE 3. MONITORING WELL DATA FROM ANACONDA MINE RECLAIMED WASTE DUMP, BELT, MONTANA

Monitoring Well	pH	Conductivity micromhos/Cm	Temperature C°	Well Depth m	Well Evaluation m	Depth to Water m	Water Evaluation m	Depth of Water m
MW - 1	3.9	9,000	11.8	4.2	30.3	3.7	26.6	0.48
MW - 2	3.4	14,500	10.7	4.8	29.8	3.2	26.7	1.64
MW - 3	1.9	26,000	12.2	4.4	30.6	3.5	27.0	0.79
MW - 4	4.0	8,200	10.6	7.2	31.4	3.4	27.0	2.84
MW - 5	2.4	8,900	10.1	5.2	31.9	4.8	27.0	0.06
MW - 6	3.8	12,000	10.3	5.4	32.3	4.6	27.6	0.70

This information was collected from testing site on August 09, 1995

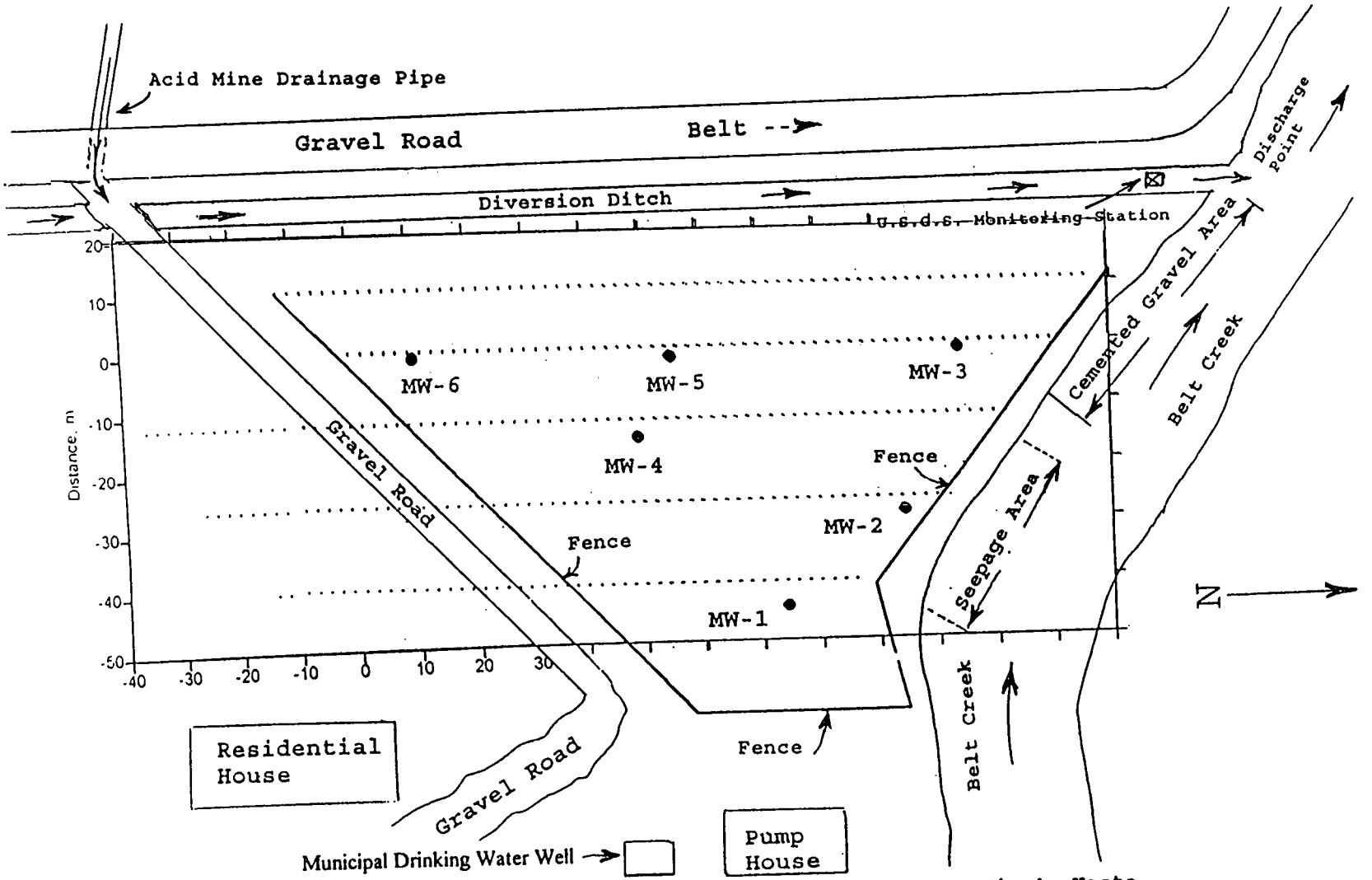

Figure 1. Monitoring Well Locations at Anaconda Coal Mine's Waste Dump Site, Cascade County, Belt, MT.

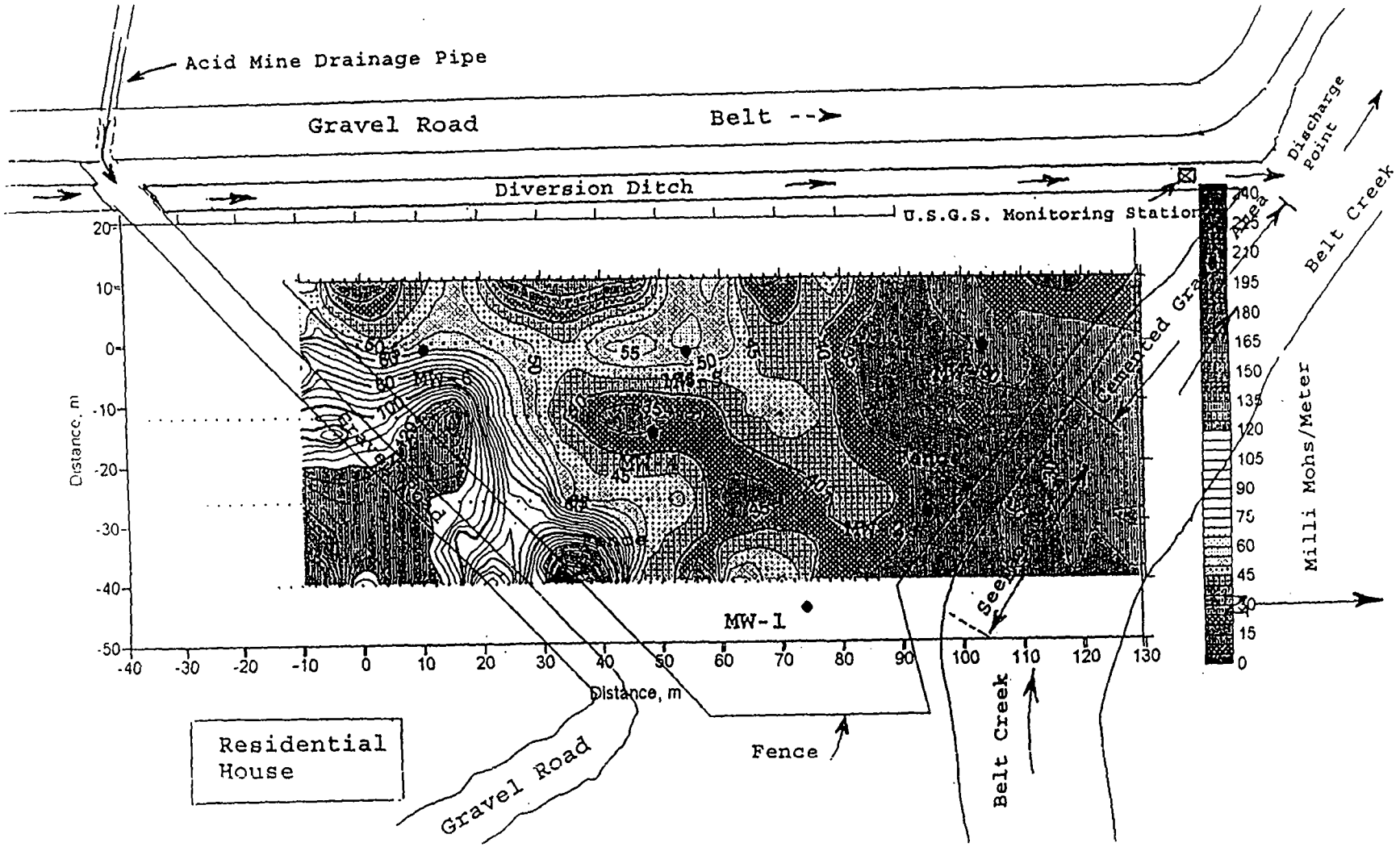

Figure 2. Geophysical Survey, EM34, Horizotal Dipole, Anaconda Coal Mine's Waste Dump Site, Cascade County, Belt, MT.

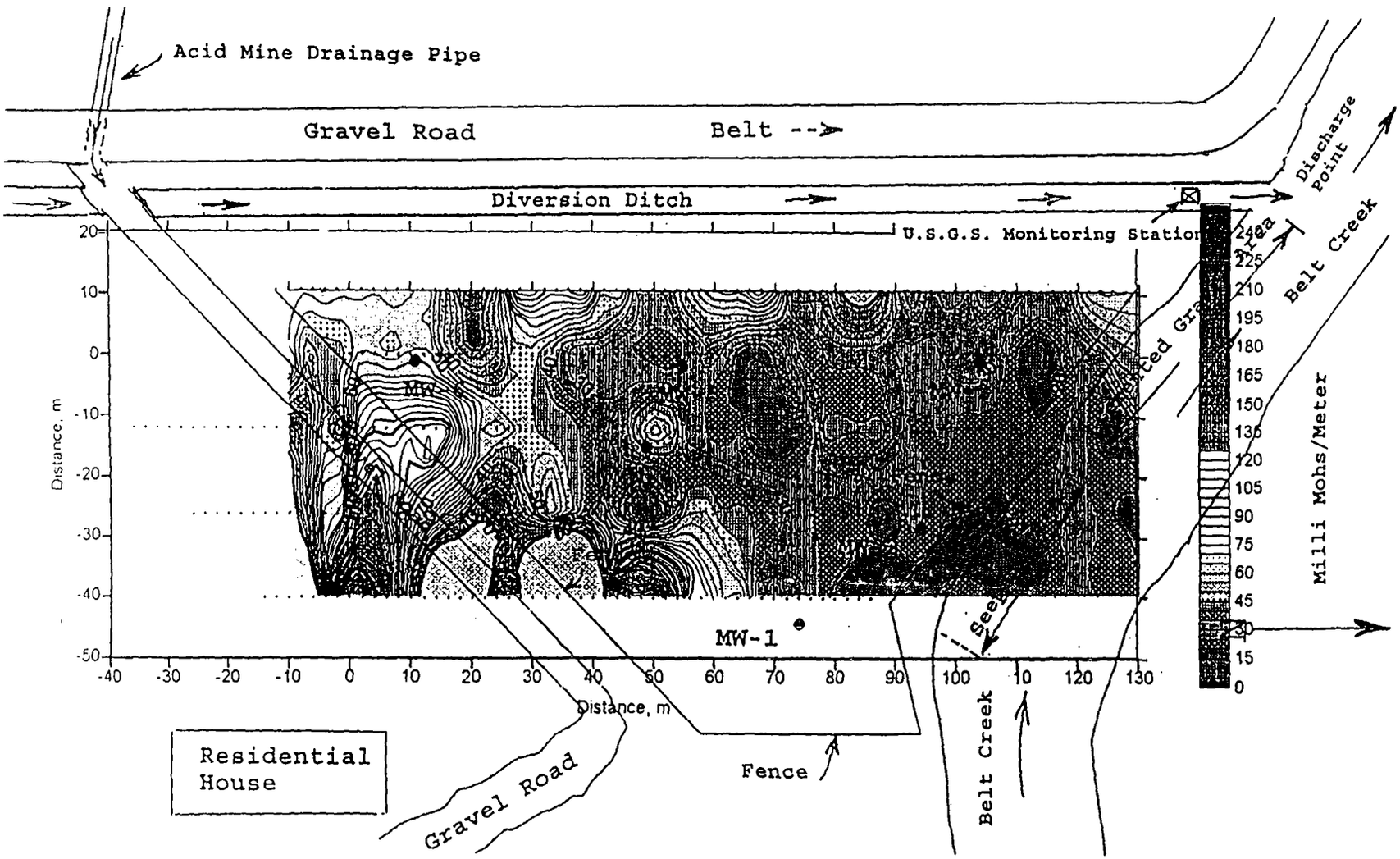

Figure 3. Geophysical Survey, EM34, Vertical Dipole, Anaconda Coal Mine's Waste Dump Site, Cascade County, Belt, MT.

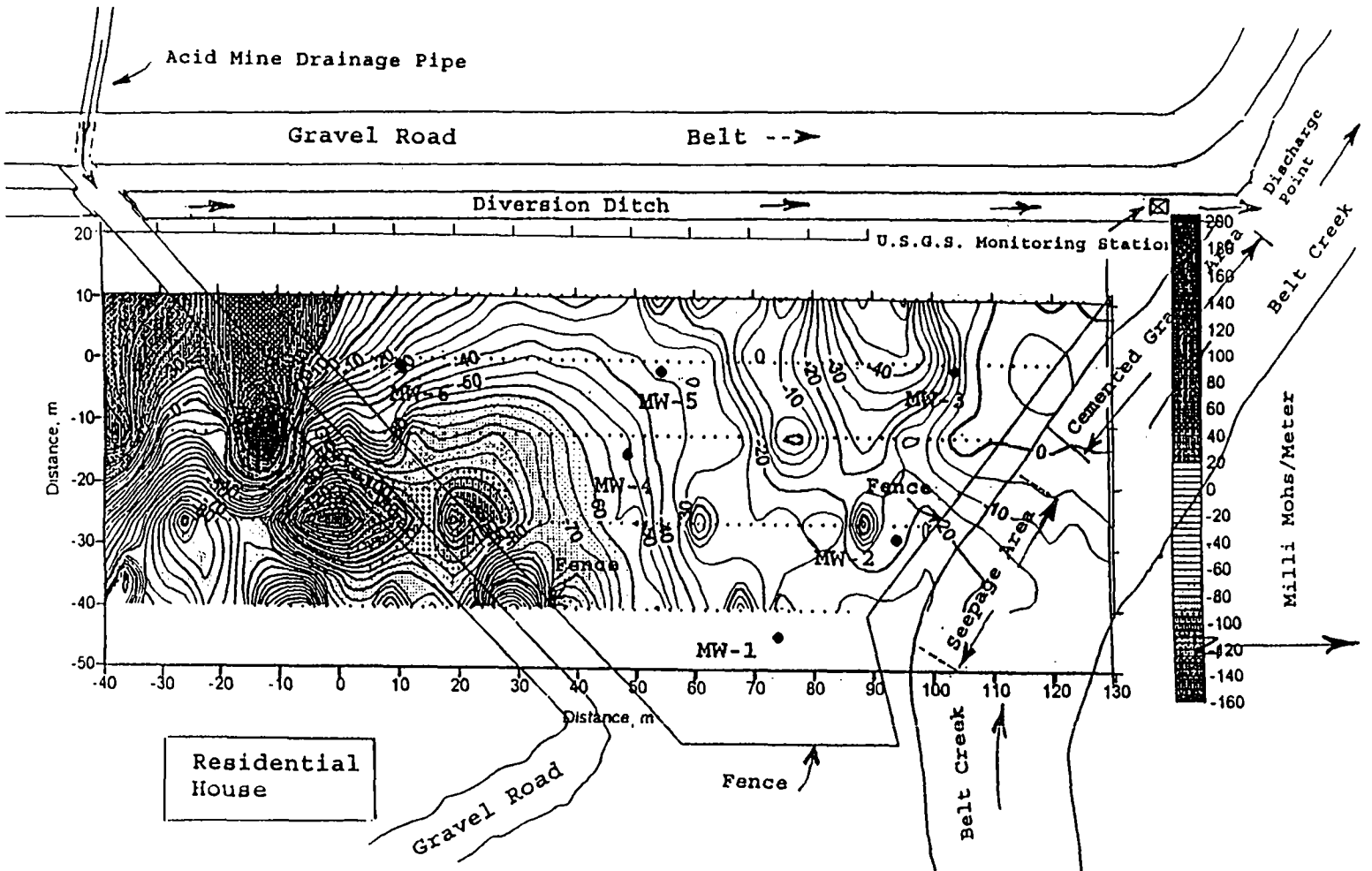

Figure 4. Geophysical Survey, Self-potential, Anaconda Coal Mine's Waste Dump Site, Cascade County, Belt, MT.

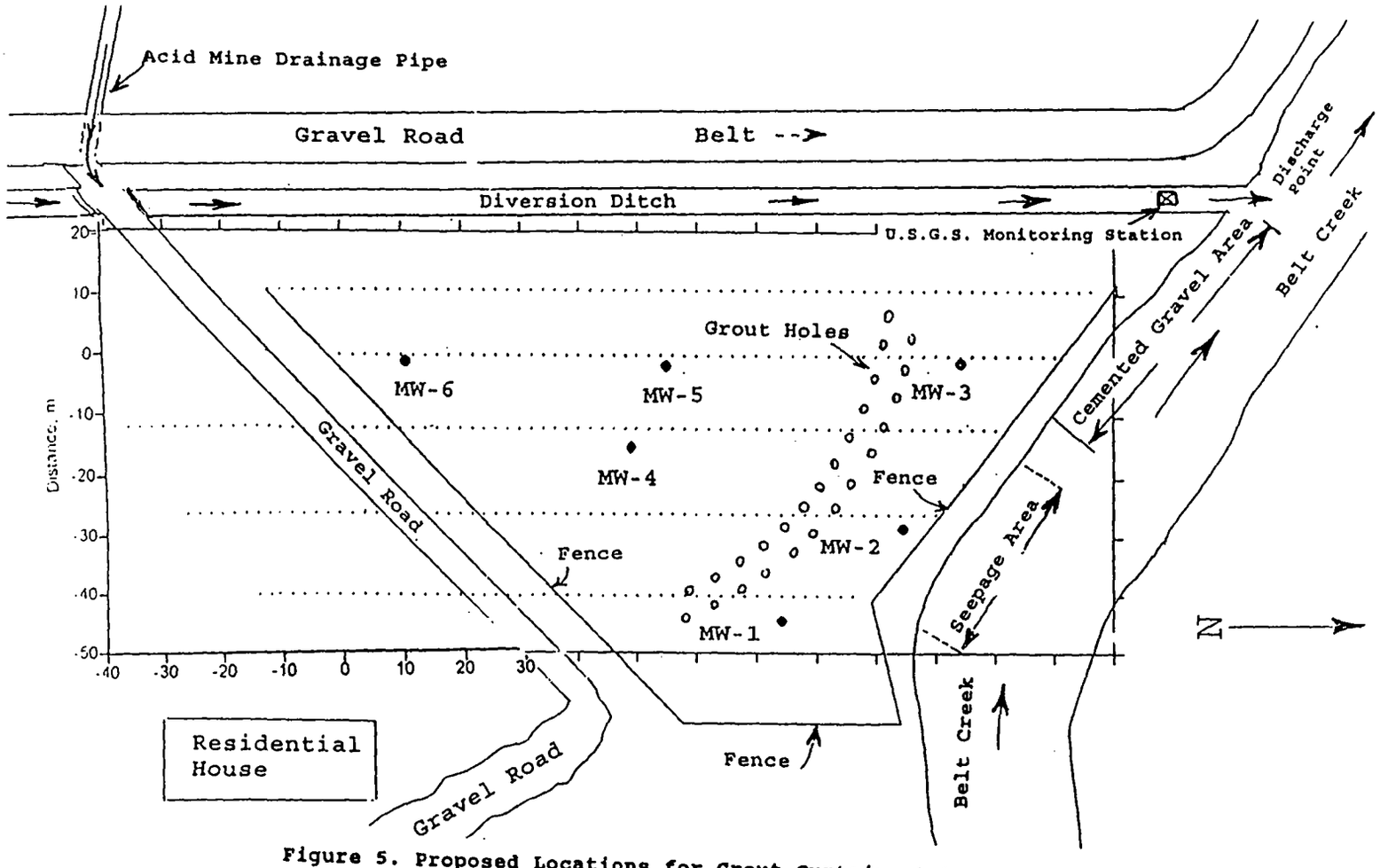

Figure 5. Proposed Locations for Grout Curtain at Anaconda Coal Mine's Waste Dump Site, Cascade County, Belt, MT.

SESSION 2: MINE HYDROLOGY

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Shallow Ground Water Flow in Unmined Regions of the Northern Appalachian Plateau: Part 1. Physical Characteristics[1]

by
Jay W. Hawkins, Keith B.C. Brady, Scott Barnes, and Arthur W. Rose[2]

Abstract. At a Pennsylvania study site, ground-water flow and aquifer properties in undisturbed strata are controlled by fracture frequency and aperture development. Analysis of images from a borehole video camera illustrate that the frequency of horizontal and vertical fractures decreases non-linearly with increasing depth. A highly transmissive fractured zone extends from 0 to roughly 15 meters in the strata underlying hilltops and hillsides. This zone exhibited hydraulic conductivities over 100 times higher than strata lying 8 meters or more deeper. These extreme hydraulic conductivity changes permit a temporary perched water table to arise from rainfall events. Surface water quickly enters near-surface fractures and flows rapidly downward and laterally through the shallow fractured zone. Well hydrographs illustrate that the residence time for much of this water is relatively short-- a few days up to a week. Most of this shallow ground-water flow emanates at cropline springs, while most of the remainder continues as shallow ground-water flow and a small portion of the ground water enters the deeper flow system. In the deeper system, the flow rate is substantially slower with a much longer residence time than the shallow zone. This is caused by a low hydraulic conductivity (*i.e.* geometric mean of 1.1×10^{6} m/s). The longer residence time allows the deep-flowing ground water more time to react with the minerals in the strata, increasing conductance and dissolved solids concentrations, relative to the shallow ground water. Specific conductivity profiles in the uncased boreholes illustrate that the conductance of the ground water increases with depth. The increases in conductance observed in some cases were gradual, while in other cases conductance increases were very discrete, corresponding to major fractures intersected by the borehole.

Additional Key Words: Appalachian Plateau, shallow ground-water flow, fracture frequency.

Introduction

The development of stress-relief fractures in undisturbed strata in the Appalachian Plateau has been widely reported (Ferguson, 1967; Ferguson and Hamel, 1981; Wyrick and Borchers, 1981; Wright 1987; Merin, 1989). Stress-relief fractures are created by the removal of rock mass by natural erosional processes. Stress-relief fractures that form in the valley walls are vertical or near-vertical and commonly parallel the valley orientation. Bedding-plane separations and small thrust faults form in the valley bottoms from the compressional stresses.

The frequency and depth extent of stress-relief fractures are directly related to the rock types, degree of induration, rock-layer thickness, and depth. Well indurated units in the valley walls such as thick sandstones and limestones tend to have a lower fracture frequency than less competent shales and claystones (Ferguson, 1967). A study of sandstones revealed a logarithmic increase in distance between fractures and increasing rock-layer thickness (Rats, 1964). Fracture

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spacing in Appalachia was observed at a fraction of a meter near the surface, increasing to over ten meters at depths up to 90 m (Trainer, 1983). Merin (1989) noted that bedding-plane fracture spacing increased non-linearly with depth, with spacing between fractures ranging from 1.3 cm to over 11 m. He saw no relationship between frequency and increasing depth for most vertical joint sets. Ferguson and Hamel (1981) observed that fractures in competent units tend to terminate at bedding contacts in softer units and that fracturing frequency decreases with increasing distance into the valley walls in the Appalachian Plateau. Borchers and Wyrick (1981), working in southern West Virginia, observed that stress-relief fractures "are a near-surface phenomenon" and that they decrease with depth. They doubted whether these fractures extend below 30 to 60 m. Kipp and Dinger (1987) noted that most fractures were within 46 m of the surface in eastern Kentucky. They reported that highly fractured rock units near the surface were relatively unfractured when more deeply buried beneath ridges. Wright (1987) also observed a significant reduction in fracture frequency and development with increasing depth in southwestern Virginia.

Ground-water flow in strata of the Appalachian Plateau is mainly controlled by the permeability created by stress-relief fractures. Brown and Parizek (1971) observed that sandstones and coals were the highest-yielding aquifers at two sites in central Pennsylvania. Sandstones, coals, and well indurated-units are more transmissive because they are self supporting and can hold fractures open. Softer units (*e.g.*, shales and claystones) are less permeable, because the fracture apertures are narrower (Peffer, 1991). These units are somewhat plastic and self healing; when fractures are created they have relatively small apertures or they tend to close up (Hawkins, 1995).

In a ground-water model developed by Kipp and others (1983), precipitation infiltrates into near-surface open fractures and percolates down until a confining unit is reached. The water then flows horizontally toward the outcrop, where it intersects fractures in the confining unit that allow downward flow, or where it emanates as a cropline spring. The remainder of the ground water continues in a stairstep fashion downward and toward the valley floor. Brown and Parizek (1971) observed a similar flow pattern. They divided the ground-water flow system into three components: 1) an "upper flow system" encompassing topographic highs down to the underground coal mines, 2) a "middle flow system" which discharges laterally and towards the stream valley, and 3) a "lower flow system" that represents the regional ground water system discharging to major river systems. Abate (1993) developed a conceptual ground-water model entailing a series of perched systems above each coal seam at the study site. In this paper, the conceptual ground-water system of this site is expanded and modified using additional testing techniques and data. Previous research and work by the authors at numerous other sites in the Appalachian Plateau indicate that the model developed has widespread applicability.

Site Background

The study site, the Kauffman mine, is located in Boggs Township, Clearfield County, Pennsylvania, midway between the towns of Clearfield and Houtzdale. A map of the site is located in the second part of this paper (Brady and others, 1996). The site is located between Camp Hope Run to the north and Sanbourn Run to the south, within 300 m of the regional drainage system, Clearfield Creek. The site is in a broad upland setting dissected by northwest-flowing tributaries. Topography is fairly steep, with a relief exceeding 200 m above drainage. The soil is relatively thin, sandy, well drained, and rocky. The predominant vegetation are hardwood trees with an undergrowth of ferns and mountain laurel.

Strata of the site and adjacent areas range from Mississippian sandstones at the level of Clearfield Creek to Pennsylvanian siltstones and sandstones forming the hilltops. Mining affects the Middle and Lower Kittanning formations, with the Lower Kittanning No. 2 and No. 3 coals being the target seams. These seams are usually separated by a carbonaceous claystone about 0.5 m thick. Above the No. 3 coal are dark gray to black shales and siltstones. Overlying the dark shales

are gray, thick cross-bedded channel sandstones. In places, these sandstones rest directly on top of the No. 3 seam. About 7.5 m above the No. 3 coal is a thin rider seam, that may be the Lower Kittanning No. 4 coal. Above the rider seam, the strata are predominantly sandstones and siltstones with minor occurrences of clay, shales, and coals. In the western portion of the site, some of the strata within the Lower Kittanning overburden are alkaline. The strata dip slightly (< 2°) toward the west-northwest. Ground water moves primarily through fractures in the rock and cleat in the coal.

At several locations across the site, piezometer nests were installed at 3 or 4 discrete levels. The "A" wells were completed with the 1.5 m screened interval open to the Lower Kittanning coal and enclosing strata. The "B" wells were installed with a 1.5 m screened interval enclosing the Clarion No. 2 coal beneath the Lower Kittanning coal. The "C" wells were completed with a 3 m open interval accessing strata beneath the Clarion coal seam. The "D" wells were left as open uncased boreholes extending down to units slightly below the Lower Kittanning coal with only 3 m of casing extending from the surface through the soil horizon. A few well nests had "E" wells installed in sandstones open at a 1.5 m interval located between the "A" and "B" wells.

Fracture Frequency Analysis

A borehole video camera was used in the uncased boreholes to record and analyze characteristics of the fractures intersected. Fracture frequency, distribution, orientation, and ground-water-yielding fractures were determined from these borehole surveys. Previous research has shown that fracture density decreases with depth (Ferguson and Hamel, 1981; Borchers and Wyrick, 1981; Kipp and Dinger, 1987; Wright, 1987). Studies on the rate of fracturing decrease or fracture orientation are rare. Merin (1989) recorded decreasing bedding-plane fracture frequency and a decrease in hydraulic conductivity with depth in a hard siltstone. Decreasing fracture frequency was reflected by decreases in permeability and significant water-quality differences.

Video tapes of the uncased boreholes were reviewed to determine the fracture orientation and the fracture frequency. The fracture counts were grouped in 10 foot intervals and normalized into fractures per unit length.[1] The first interval started at 10 feet below the surface because casing was installed for hole stability. Fractures were categorized by orientation: 1) vertical/near vertical (90°+/-10°), 2) horizontal/near horizontal, mainly bedding plane separations (0°+/-10°), and 3) oblique, all fractures between vertical/near vertical and horizontal/near horizontal.

The highest fracture frequency intersected by the borehole was observed for horizontal fractures. The relatively high number of horizontal fractures was somewhat unexpected, based on previous research by Ferguson and Hamel (1981) that indicated that bedding-plane separations were more common to the valley bottoms than the ridge tops. Trainer (1983) observed that "bedding-surface openings" are common in horizontally bedded rocks in West Virginia, Maryland, Virginia, and Tennessee. The frequency of horizontal fractures (fig. 1) decreases substantially with depth. The average number of horizontal fractures intersected by the borehole decreases from 0.7 and 0.93 fractures per foot for the 10-20' and 20-30' intervals, respectively, to less than 0.1 fractures per foot at the 100-110' level.

Vertical fractures exhibited the second highest frequency of the three types (see fig. 2). However, given the mode of frequency determination, the number of vertical fractures are probably under-represented spatially in the ridge top. The fractures and boreholes are parallel or nearly so. Therefore, it is anticipated that the number of vertical fractures observed is less than the actual frequency in the strata. Measurements at freshly-exposed highwalls indicate that this assessment is correct. Vertical fractures

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[1]As an artifact of the borehole videography, the fracture frequency analysis will be reported in english units to maintain consistancy with figures 1 and 2.

frequency significantly decreased at increasing depth. The average number of vertical fractures ranged from 0.77 fractures per foot at the 10-20' interval to none being recorded at the 100-110' interval. Although no fractures were encountered at the 100-110' interval, measurements of hydraulic properties indicate that fracturing extends beyond 100'.

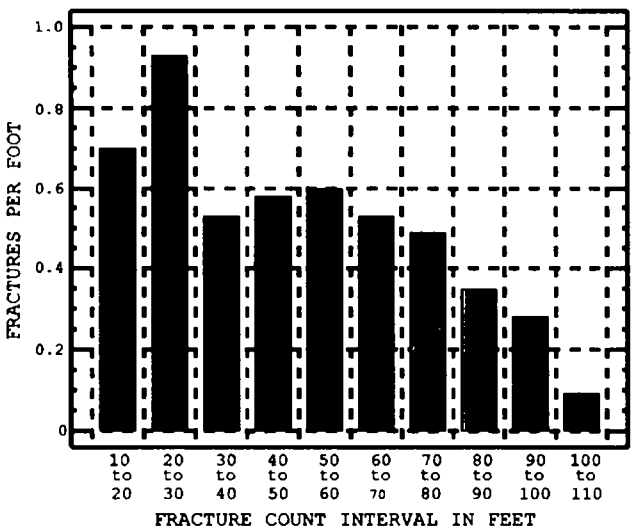

Figure 1. Frequency of horizontal fractures.

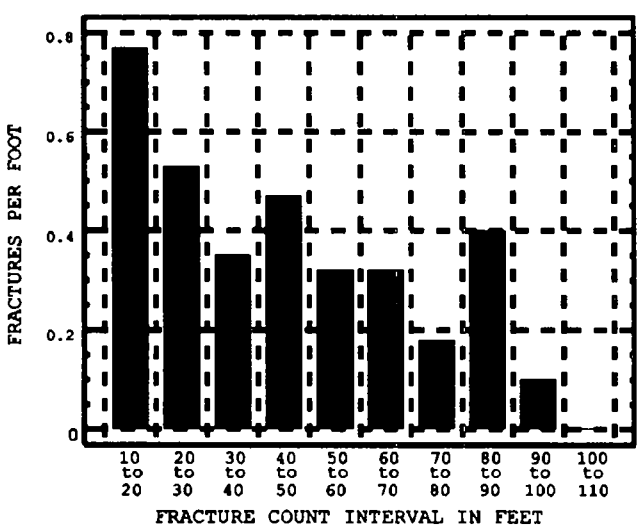

Figure 2. Frequency of vertical fractures.

Oblique fractures exhibited the lowest frequency of the three types. No trends with regard to depth were noted. The lack of frequency changes with depth may be related to the genesis of oblique fractures. The stress-relief origins of most of the horizontal and vertical fractures in the ridge-top wells are directly related to tensile stresses from rock mass removal. Oblique fracture orientation is consistent with those created by compressional forces from past tectonic activity. Compression generated fractures are usually less than 90° and generally 30° on either side of the direction of the compressive force (Billings, 1972). Given the relatively shallow nature of the strata studied (<35 m) and the forces creating oblique fractures, a change in frequency with depth was not expected because tectonic forces are not limited to shallow depths.

Aquifer Properties

Decreasing stress-relief fractures with increasing depth is reflected by lower hydraulic conductivities recorded in the uncased boreholes as the water level decreased. The hydraulic conductivity of several of the uncased boreholes was determined using slug injection tests. The wells were retested several times with differing water levels (saturated thicknesses). The uncased boreholes exhibited a wide range in water levels with fluctuations commonly exceeding 10 m.

Changes in hydraulic conductivity that spanned two orders of magnitude were recorded with water-level fluctuations.[2] With a water-level drop (saturated thickness decrease) of 8 m, from 16.8 m to 24.8 m below the surface, well W6D exhibited a hydraulic conductivity decrease from 3.6×10^{-6} to 3.3×10^{-8} m/s. A similar change was observed in that borehole with a 7.5 m reduction of water level. Well W5D exhibited a decrease in hydraulic conductivity of nearly two orders of magnitude, from 3.1×10^{-7} to 6.0×10^{-9} m/s, with a 8 m water-level drop, 10.8 m to 18.6 m. Well W22D exhibited a change in hydraulic conductivity from 4.6×10^{-6} to 1.3×10^{-7} m/s with a water level decrease of 3.8 m. Testing of well W7D showed a substantial and systematic reduction in hydraulic conductivity with minor water-level drops. With a water level of 21.55 m,

[2]Complete hydraulic conductivity and transmissivity data for this site are available upon request.

22.24 m, and 22.61 m below the surface in W7D, the estimated hydraulic conductivity was 2.3 X 10^{7}, 1.3 x 10^{8}, and 5.1 x 10^{9} m/s, respectively. Merin (1989) observed similar hydraulic conductivity decreases with depth in a siltstone aquifer of the northern Appalachian Plateau.

The remaining uncased boreholes (W2D, W3D, and W4D) exhibited relatively small water-level changes (<3 m) when they were tested. Hydraulic conductivity changes for these wells and wells W5D, W6D and W22D, with minor water level changes, were small (<10X).

In contrast to the uncased boreholes, aquifer testing of the piezometers by both slug injection and constant discharge yielded no distinct trends with regard to increasing depth. The hydraulic conductivity ranged from 10^{6} to 10^{7} m/s. The lack of trends may be due to the relatively narrow open interval (1.5 m or 3 m). The piezometers access the deepest levels or below the levels of the uncased boreholes. Fractures at depth are of a hit-or-miss nature. Short intervals (<5 m) with very few or no visible fractures were commonly observed in the boreholes. This was especially true for the deeper rock units. Therefore, the hydraulic conductivity of the piezometers may exhibit large deviations from the average conductivity, mainly toward lower values.

Well Hydrographs

The substantial hydraulic-conductivity decreases observed with depth create a temporary shallow perched system following significant precipitation events. Hydrographs for the uncased boreholes and water-level measurements in the piezometers indicate that precipitation events create a bimodal flow system in the uppermost unconfined aquifer. A shallow perched flow system develops above the water table, causing water levels in the uncased boreholes to rapidly rise. The deeper water table continues to exist in the "core" of the hill above the underclay of the Lower Kittanning coal. The piezometers did not exhibit similar rapid water-level changes. Tighter strata within the deeper system cause ground-water flow to be much slower, with longer retention times than the shallow zone. Recharge to this lower zone has a longer lag time than the shallow zone.

With precipitation, water rapidly infiltrates into near-surface fractures. Ground water flows vertically for several meters (roughly 7 to 17 m) and then flows laterally toward the hillsides. Depth to which the water will percolate until it begins to flow laterally depends on the hydraulic conductivity, which is directly dependent on fracture aperture size and frequency. Aquifer testing illustrates that large changes in hydraulic conductivity (>100X) can occur within a short distance (8 m) vertically. This magnitude of hydraulic conductivity change is often defined as the distinction between an aquifer and an aquitard. These hydraulic conductivity changes with depth cause the formation of a temporary shallow perched water table. Water from this shallow system rapidly enters the uncased boreholes and fills them to the level of the upper saturated surface (see figs 3) and 4). The rate of inflowing water from shallow depths is greater than the rate the water is reintroduced into the lower and less permeable strata. The subsequent return of water levels to pre-rainfall levels are regulated by the hydraulic conductivity of the lower units and the low inflow rate that continues for several days to a few weeks after a precipitation event. Similar temporary perched ground-water systems have been observed at numerous other sites in the northern Appalachian Plateau (*e.g.*, Garrett Co., Maryland; Westmoreland Co., Pennsylvania) (Hawkins, unpublished data).

The boreholes exhibited water-level rises exceeding 12 meters from the onset of rain. Commonly, the initial water-level rise was observed 5 to 20 hours after the start of rain, whereas the peak usually occurred in less than 30 hours for well W5D and 48 hours for well W6D. This compares to 25 hours for the initial rise and 120 hours for peak rise for a cropline spring located on the site (Abate, 1993). The longer response time for cropline springs is caused by the longer vertical and lateral flow distance. Figures 3 and 4 are examples of water-level responses to rainfall events of 2.92 cm over 12.5 hour and 7.44 cm over 22 hour periods, respectively. These rapid water-level increases, if no additional rainfall

occurs, are followed by a much slower return to pre-rainfall levels (Fig. 4). The protracted return to pre-rainfall levels is regulated by the permeability of the lower sections of the boreholes. Piezometers accessing the same units as the lower sections of the open boreholes do not exhibit the same rapid water-level rise as the boreholes. The water levels in the boreholes were over exceeding 15 meters above levels measured concurrently in comparable piezometers. A much longer time interval is required with recharge to this lower zone.

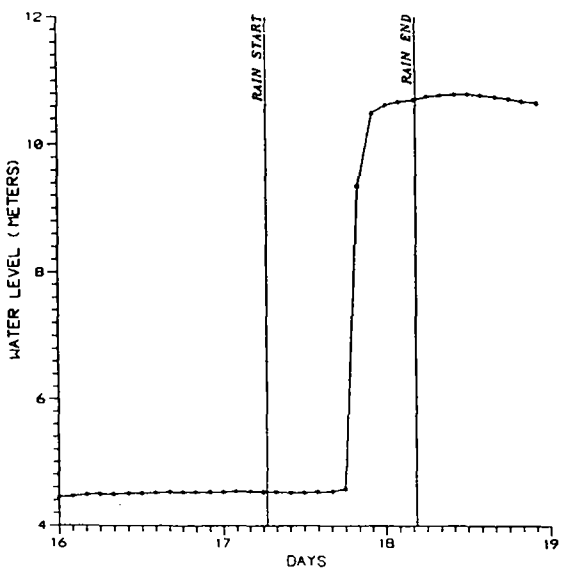

Figure 3. Hydrograph of well W6D from August 16 to 19, 1994.

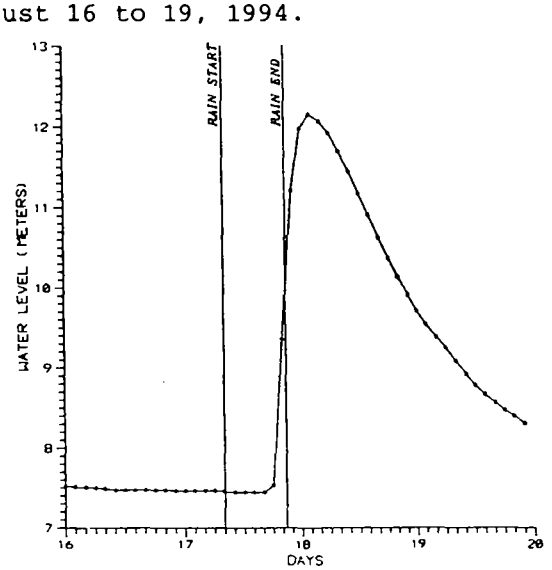

Figure 4. Hydrograph of well W5D from September 16 to 20, 1994.

Most of the ground water in the shallow perched zone, upon reaching the less fractured and unweathered zone, will rapidly flow laterally along a path nearly parallel to the surface. When a unit is encountered with poorly developed fractures, (*e.g.*, coal underclay), much of this water will emanate as a cropline spring. Figure 5 is a schematic cross-section, conceptually illustrating the shallow ground water flow. This water has a relatively short in-ground residence time and flows primarily through highly-leached strata. Therefore, the spring water exhibits a very low ionic strength, similar to rainwater. The chemical quality aspects of this water are discussed by Brady and others (1996).

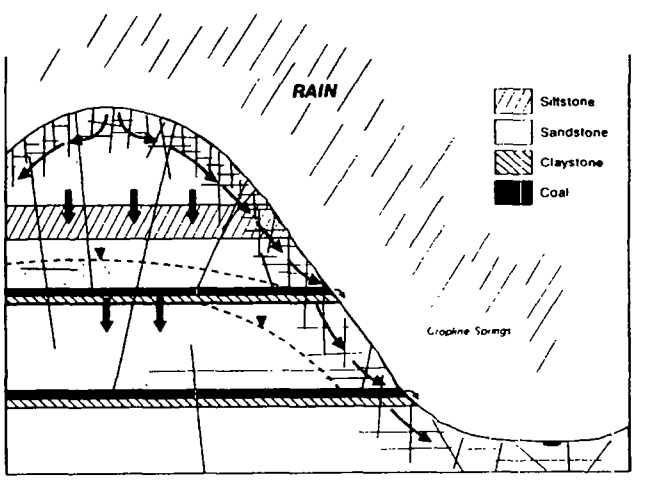

Figure 5. Schematic cross-section with conceptual ground-water flow paths.

The vertical hydraulic conductivity changes observed are not as abrupt as those associated with lithologic changes; they occur over a moderate distance. Therefore, a small amount of ground water flow continues in the shallow zone for several weeks after precipitation. A low-rate seepage was commonly observed into the open boreholes (at this site as well as several other sites in the northern Appalachian Plateau) several weeks following the last rainfall. This small amount of ground-water flow is what maintains the flow at the springs

through dry periods. The gradual reduction of vertical and horizontal permeability in the shallow fractured zone permits continued ground-water flow long after the last rainfall event. Small aquitards or less permeable areas within the shallow fractured zone may also be a source of this slow-release water. Spring-water quality during dry periods indicates that the majority of the water originates in the shallow fractured zone, rather than in the deeper ground water system (Brady and others 1996).

Conductivity Logs

The short residence time of the shallow perched system and the longer residence time of the deeper unconfined system is reflected in the water quality observed in the uncased boreholes. Logging recorded increases in specific conductance with increasing depth in the uncased boreholes.

Some of the boreholes or portions of boreholes exhibited a relatively gradual increase with increasing depth. Figure 6 illustrates a gradual specific-conductance increase from 11 to 18 m and 19 to 30 m below the surface. Other boreholes exhibited substantial conductivity rises in a short distance (0.3-1.0 m). Figure 6 exhibits one of these rapid rises between the 18 and 19 m levels. A series of these conductivity jumps are exhibited by figure 7 at 20, 28, 31, and 32 m below the surface. The video logs show that these conductivity increases correspond to the locations of prominent fractures intersected by the borehole.

Not all fractures observed in the boreholes caused conductivity increases. There were numerous prominent fractures intersected by the boreholes that did not correspond to conductivity increases. However, the conductivity increases were always associated with fractured zones. This indicates that not all fractures contribute to the ground-water flow system. Booth (1988) observed that individual fractures may represent discrete aquifer zones that may have a distinctly different piezometric surfaces. Rasmuson and Neretnieks (1986) estimated that 5-20 percent of the fracture plane carries 90 percent of the water. Ground water flows through "channels" within fracture planes. Where fractures do not have corresponding conductance increases, ground water in these fractures may be of similar quality to water in fractures located above, or this may be where water from the borehole re-enters the aquifer. The latter is especially true in the lower portions of the hole, but may also occur in higher fractures during substantial recharge periods. When the head in the borehole exceeds the pressure head of a fracture, inflow from the borehole to the aquifer will occur.

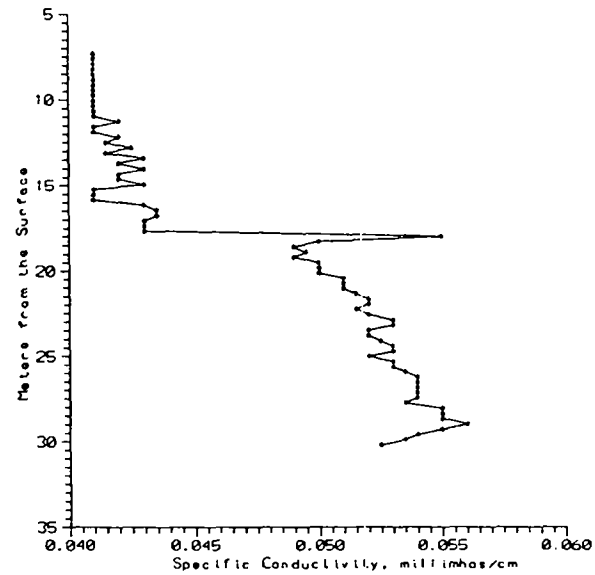

Figure 6. Specific conductance log for well W4D.

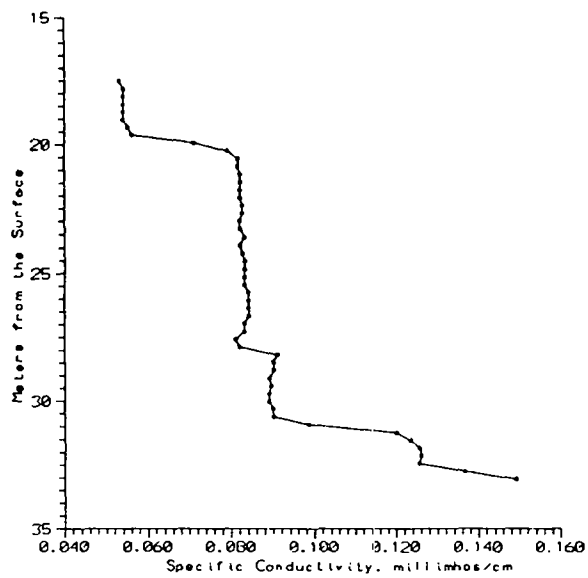

Figure 7. Specific conductance log for well W6D.

Piezometer Data

Head levels measured in the piezometers indicate that there is a downward flow component for ground water across the site. Head potential decreases between piezometers within a nest ranged from 0.23 to 1.0 m for each meter of increasing depth. Decreasing water levels indicate that the ridge top is a recharge area. The water levels measured in the "D" wells represent a composite of the different head levels of the rock units accessed.

Water levels exhibited by the piezometers indicate that a series of confined or semi-confined aquifers exist below the unconfined aquifer above the Lower Kittanning coal. Piezometric surfaces exhibited by "C" wells were above the level accessed by the "B" wells, but below their piezometric level. Abate (1993) suggested that "non-uniform pressure distribution" and substantial vertical head reduction below coal underclay units indicated that a series of perched aquifers exist near the outcrop. These aquifers above the regional aquifer probably exhibit both confined and unconfined conditions depending on the location monitored within the site (*e.g.*, unconfined near the outcrop). Specific portions of an aquifer may also exhibit confined and unconfined conditions at different time periods.

Discussion and Conclusions

Based on the data and information presented, a refined conceptual ground-water model has been created. An extensive review of the published literature indicates that this model is applicable for fractured sedimentary strata throughout most of the Appalachian Plateau. Minor modifications may be required in some regions.

A highly fractured and weathered zone up to 20 meters thick blankets the hilltops and hillsides in the Appalachian Plateau. This zone is highly transmissive and is underlain by progressively less transmissive fractured units. This less transmissive zone facilitates a temporary perched system from rainfall events. Rainwater quickly infiltrates into near-surface fractures flowing vertically for a several meters and then flows laterally toward the hillsides. This water has a short residence time in this shallow, highly transmissive zone. Much of the laterally flowing near-surface ground water emanates at cropline springs. These springs occur at the level of the coal seams. Fractures in the coal underclay tend to be poorly transmissive. Some shallow flowing water passes through the underclay and continues down slope in the weathered/highly fractured zone. Some of this shallow-flowing ground water recharges underlying aquifers via deeper fractures and emanates from coal seams at lower elevations. This conceptual ground-water flow system is illustrated by figure 5.

The short residence time (days to a week) for most of the shallow-flowing ground water is reflected by the low ionic strength of water at cropline springs. Ground-water quality is reported in detail in the second part of this study (Brady and others, 1996).

Data collected from the piezometers indicates that a series of confined or semi-confined aquifers exist beneath the water-table aquifer underlying a coal underclay. Decreasing head levels with depth indicate that there is a downward ground-water flow component. Aquifer tests performed on piezometers in the deeper confined aquifers illustrate that they have low transmissive properties and moderate head pressure; therefore, ground water movement into them and through them is slow. Decreases in the fracture frequency with depth account for the reduction in hydraulic conductivity with depth in the unconfined aquifer and the low hydraulic conductivity of the underlying confined units.

Determination of ground-water flow in the Appalachian Plateau is complex. Changes in transmissive properties caused by decreasing fracture development with increasing depth must be considered when designing a monitoring and water-quality characterization plan. Water quality at cropline springs is representative of ground water flowing through a near-surface, highly fractured and weathered zone. Monitoring wells installed into the core of the hill access ground water that reflects the deeper and slower moving aquifer systems. Deep ground-water quality is indicative of

the geochemical quality of the deeper strata. Below the unconfined zone, a series of confined aquifers exist with a continued downward ground water flow component.

A comprehensive literature review and the widespread experience of the authors indicate that the conceptual ground-water model developed in this paper is applicable to fractured strata aquifers throughout the Appalachian Plateau. However, in certain regions, minor adjustments may be required to completely characterize the system.

Acknowledgments

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SHALLOW GROUNDWATER FLOW IN UNMINED REGIONS OF THE NORTHERN APPALACHIAN PLATEAU: PART 2. GEOCHEMICAL CHARACTERISTICS[1]

Keith B.C. Brady, Arthur W. Rose, Jay W. Hawkins,
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ABSTRACT. Surface mines in Pennsylvania occur in the shallow (< 60 m depth) groundwater flow system and typically are located in groundwater recharge areas. Two shallow flow systems are usually present in unmined areas: an upper weathered-rock zone and a deeper unweathered-rock zone. The weathered-rock zone, although variable, is commonly 6 to 12 m thick. Groundwater in the weathered-rock flow system, as evidenced by shallow wells and cropline springs, has low dissolved solids (specific conductance < 50 μS/cm). The deeper unweathered-rock flow system, as evidenced by water from wells deeper than 10 m, has higher concentrations of dissolved constituents. The differences in water quality are due to previous intensive leaching of minerals within the weathered-rock zone and to much higher flow rates in the more porous and fractured weathered zone. In particular, calcareous minerals are absent or negligible in the weathered-rock zone, but can be appreciable in the unweathered-rock zone. This distribution of calcareous minerals, groundwater contact time with rock, and flow path influence the groundwater composition and concentrations of dissolved ions. A positive relationship exists between the presence and abundance of calcareous minerals and associated groundwater alkalinity.

Our observations are probably applicable to much of the Appalachian Plateau. Groundwater alkalinity can help determine the presence and distribution of calcareous minerals within coal overburden. Coal-cropline springs should not be depended upon for showing groundwater quality associated with the coal seam; they typically only reflect shallow flow through weathered rock. Deeper wells are required to determine the chemical characteristics of water in the unweathered rock zone.

Additional Key Words: groundwater chemistry, acid-base accounting, weathering, groundwater flow

Introduction

The companion paper (Hawkins et al., 1996, this proceedings) provides a review of groundwater hydrology studies on the plateau. This paper addresses chemical evidence in support of that model. Our model defines two separate shallow groundwater flow systems in unmined areas within the unglaciated portions of the Appalachian Plateau of western Pennsylvania. Numerous studies throughout the plateau, from Kentucky to New York have examined shallow groundwater hydrologic characteristics, but few have related water chemistry to the groundwater hydrology.

The three areas discussed in this paper lie within the Allegheny Mountain section of the Appalachian Plateau physiographic province. This portion of the plateau has moderate to high relief (90 to 300 m), marked by deep, V-shaped valleys (Berg et al., 1989). Stratigraphically the sites occur within the Pennsylvanian Period middle Allegheny Group (Sites A and B) and upper Allegheny Group/lower Glenshaw Formation (Site C). Structurally the rocks are horizontal to gently dipping (generally < 6°).

Natural water quality in shallow groundwater flow systems of the Appalachian Plateau results primarily from the influence of three factors: (1) the chemistry/mineralogy of the rock that the water contacts, (2) flow path, and (3) water and rock contact time. Although many solutes are present, we concentrate on bicarbonate alkalinity, because it can be compared to overburden neutralization potential (NP) (an estimation of calcareous mineral content) and because it can be used as a mine drainage quality prediction tool. Other water quality parameters will be addressed where appropriate.

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Shallow groundwater is associated with two distinct flow-systems. The one system, represented by coal-cropline springs, is recharged from the near-surface zone of weathered rock up slope from the spring. This zone consists of soil, colluvium (in some areas), and weathered and highly fractured rock. This weathering is both chemical and physical. The weathering is enhanced by stress-relief fracturing on the hill slopes and by bedding plane separations, which promote intensified chemical weathering and resulting porosity increases (Hawkins et al., 1996). The weathered zone, as seen in strip mines and drill logs, is typically 6 to 12 m thick, but can extend deeper along fractures. The weathered zone is typically devoid of calcareous minerals and pyrite (Singh et al., 1982; Brady et al., 1988). It is characterized by a yellow-red-brown color resulting from iron oxidation. The lower boundary of the weathered zone is an irregular surface. The irregularity is caused by variations in fractures, topography and lithology.

The other flow-system is below the weathered rock within the zone of largely unweathered bedrock. The unweathered bedrock zone is much less fractured, having fewer bedding plane separations and stress-relief fractures. The bedrock zone retains calcareous minerals and pyrite (if originally present) and the rocks are typically gray, with iron oxidation restricted to some fractures and bedding plane separations.

Most water flowing through the weathered zone has a relatively short residence time (days to weeks) because of the shallow nature of this material and higher permeability which is induced by the factors cited above. Some storage is present, however, because the springs continue to flow during dry periods. The absence of readily leachable or oxidizable minerals and the short residence time results in spring-water quality which typically has low concentrations of dissolved ions.

The water flowing through the unweathered-rock zone has longer residence time (months to years) because of lower permeabilities (Hawkins et al., 1996) and the slower flow rate allows longer contact with soluble minerals. Groundwater in unweathered rock typically has greater concentrations of dissolved solids than water emanating from weathered regolith. Water chemistry from unmined areas can provide information concerning the groundwater flow system.

Methods

The three study areas discussed in this paper were selected because they had: (1) water chemistry data from coal-cropline springs, (2) water quality from wells completed down to the same coal seam as the cropline springs, and (3) acid-base accounting (ABA) data.

ABA is defined by two parameters, neutralization potential (NP), which is an approximation of calcareous mineral content (reported in terms of parts per thousand $CaCO_3$; traditionally reported as tons $CaCO_3$/1000 tons of material), and maximum potential acidity (MPA), which is an approximation of pyrite content. MPA is converted to the same units as NP by multiplication of the weight percent total sulfur by the stoichiometric equivalence factor of 31.25 (Cravotta et al., 1990). Average NP's and MPA's were determined for each overburden drill-hole by using thickness weighting and assuming each hole represented a column of constant diameter, following the methods described in Smith and Brady (1990).

All water quality sample locations were represented by multiple samples. Medians were determined for pH. Other water quality parameters discussed in this paper, unless specified otherwise, are mean concentrations.

Mine Sites

To illustrate the relationship between rock chemistry and water chemistry, several specific examples will be given. All sites discussed are isolated hill tops where the only recharge is from direct precipitation. In all cases the coal outcrops along the sides of the hill, and cropline-springs and bedrock monitoring wells are present.

MINE A: Boggs Township, Clearfield County

Mine A is the Kauffman mine where various research studies have been and are being conducted (see Abate, 1993; Evans, 1994; Rose et al., 1995; Hawkins et al., 1996, this volume). Mine-site topography, locations of drill holes and water-sample points for this mine are shown on Figure 1. This map also shows alkalinity and specific conductivity of wells and springs associated with the lower Kittanning (LK) coal seam, plus a few springs flowing from the Clarion #2 cropline. All the wells are completed as 10-cm-diameter piezometers that are open for 1.5 m at the interval of the LK coal. Figure 1 also

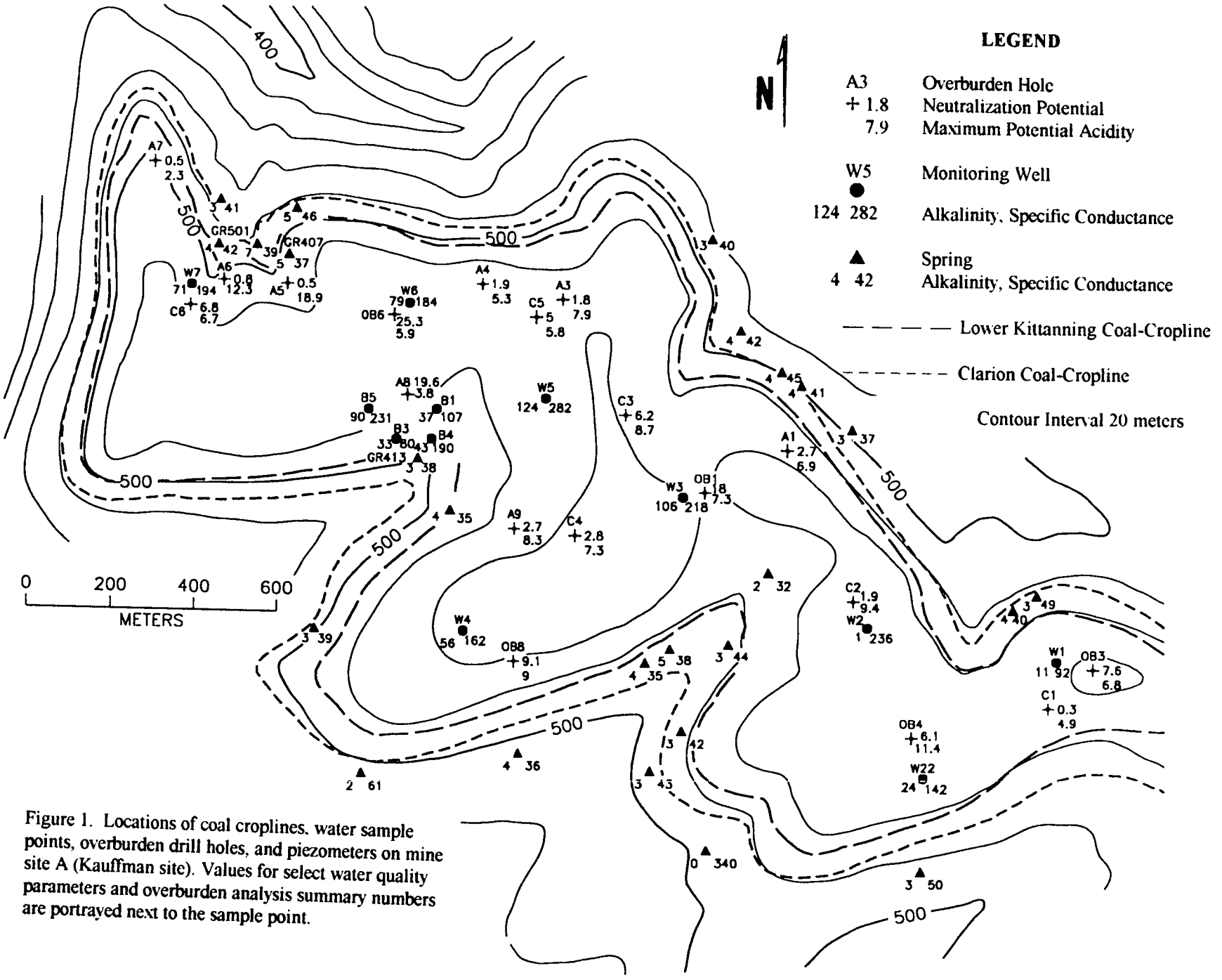

Figure 1. Locations of coal croplines, water sample points, overburden drill holes, and piezometers on mine site A (Kauffman site). Values for select water quality parameters and overburden analysis summary numbers are portrayed next to the sample point.

shows summaries of acid-base accounting data for overburden drill holes.

Figure 1 clearly indicates that spring-water quality is significantly lower in alkalinity and specific conductance than well water. Alkalinity of springs is typically less than 5 mg/L and conductivity less than 50 µS/cm. In contrast, wells typically have much higher alkalinity (typically >70 mg/L) and specific conductivity (~200 µS/cm). Piezometers completed within the hill of the Kauffman site indicate a steep downward flow gradient at the horizon of the LK coal. Between the lower permeability of the unweathered-rock zone and the downward component of much of the flow, little water from the unweathered-rock zone is contributed to the springs. It is also apparent that there is a regional distribution of well water alkalinity. Wells on the eastern portion of the area have lower alkalinity than those on the western portion. This pattern is consistent with the higher NP's reflected in the overburden data on the western portion of the site (Fig. 1).

The geologic controls on distribution of calcareous units are complex on this site. Observations of highwalls show that the calcareous material occurs as calcite and minor siderite as cement between sandstone grains in trough bottoms of some channel sandstone (Skema, 1995, personal communication). These calcareous trough deposits are laterally discontinuous and only occur in the basal portion (bottom 3 to 5 m) of a thick (18 to 24 m) channel-sandstone unit. Observations of drill core also suggest that some of the calcareous cement in sandstone is associated with vertical fractures and bedding plane separations. However, calcareous minerals are not found in most fractures.

Overburden holes were paired with wells or springs within 140 m for comparisons of alkalinity of groundwater with overburden NP. NP and alkalinity show a strong positive relationship (Figure 2), with alkalinity (mg/L) being four times the NP value (tons/1000 tons). A similar relationship is seen between specific conductance and NP (correlation coefficient, r = 0.82). As would be expected, specific conductance and alkalinity show a strong positive relationship (r = 0.94). These relationships imply that dissolution of calcareous minerals, where present, has a significant impact on groundwater chemistry.

Sulfate, a conservative aqueous ion, is a weathering product of pyrite. Therefore, a comparison was made between maximum potential acidity (MPA) and sulfate. MPA shows no relation to sulfate in nearby groundwater and has, if anything, an inverse relation to specific conductance. Sulfate concentrations were generally less than 35 mg/L.

Detailed Study of Water Quality at the GR-413 Spring. Zhradnik (1994) investigated the flow patterns in a small area at the Kauffman mine. Five air rotary drill holes (B1 through B-5, Fig. 1) were drilled within the area that apparently feeds groundwater to the GR-413 spring. The drill holes were completed with 2.5-cm-diameter piezometers in the LK coal horizon. The piezometric surface as measured in these five holes is just above the coal elevation, and indicates that water in the coal has a component of flow toward the cropline. Figure 3 is a cross-section showing the stratigraphic and hydrologic relationships up gradient from spring GR 413. The flow of the spring in conjunction with estimated evapotranspiration values indicates that the spring must drain an area of about 140,000 m^2, including the area of holes B-1 through B-5 (Abate, 1993).

Based on four years of quarterly sampling, plus Zahradnik's data, the average specific conductivity of the spring water is 42.9 µS/cm, with a standard deviation of 10.8 (Fig. 4). In contrast, water from the B-series wells ranges from 74 to 270, averaging 171 µS/cm (s.d. 72), or about four times as high. Comparison of Ca, Mg, K, and SO_4 values indicates similar ratios, as indicated in Table 1. During periods of high flow and during many periods of low flow, SO_4 is 7-12 mg/L, but during some periods of low flow, SO_4 increases to 15-25 mg/L, or even higher (Fig. 5). Most of these high SO_4 values are during late winter and early spring. This pattern indicates that most of the year, the spring is fed mainly by dilute water that does not penetrate as deeply as the LK coal horizon. Instead, the spring flow must pass through shallower horizons, very likely in the zone of highly fractured rock described by Hawkins et al. (1996). Hydrographs of this spring (Fig. 6) indicate that rainfall is accompanied almost immediately by a sharp peak in flow resulting from surface runoff (observed at the site), then a few days later by a broader peak that is interpreted to result from flow through the shallow weathered zone. Appreciable storage must exist in this shallow zone to account for the year-round flow of this spring.

MINE B: Wharton Township, Fayette County

The principal coal seam mined at site B is the upper Kittanning (UK). Field pH and specific conductance were measured prior to mining in several uncased drill holes which were completed down to the UK seam. Additionally, numerous UK cropline springs

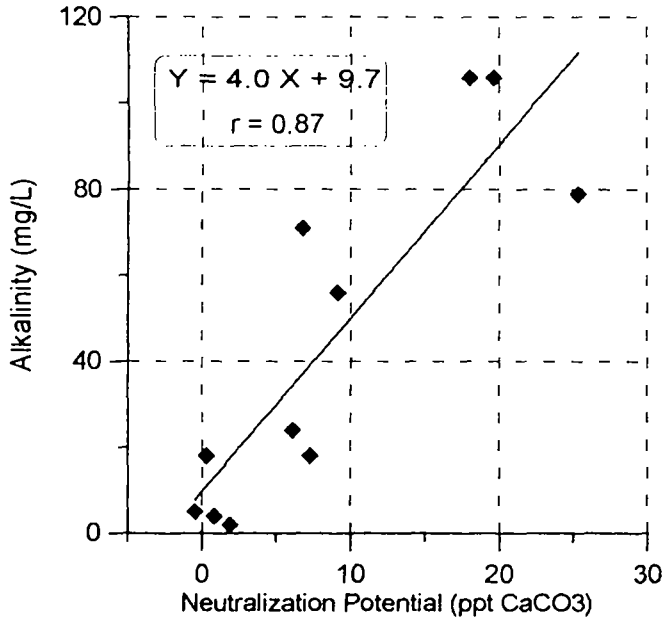

Figure 2. Alkalinity of groundwater associated with the lower Kittanning coal seam as a function of NP at mine site A.

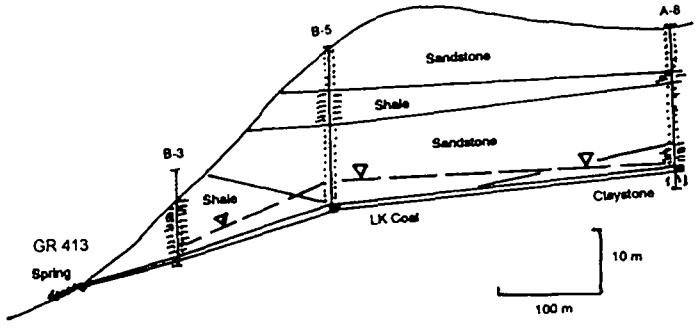

Figure 3. Cross-section through the area of GR-413.

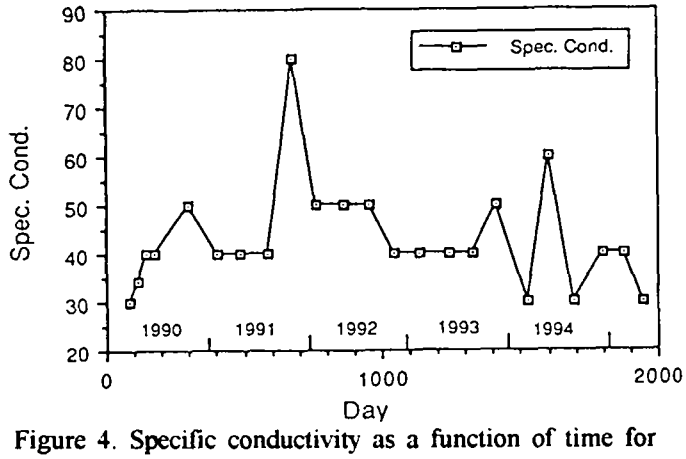

Figure 4. Specific conductivity as a function of time for spring GR-413.

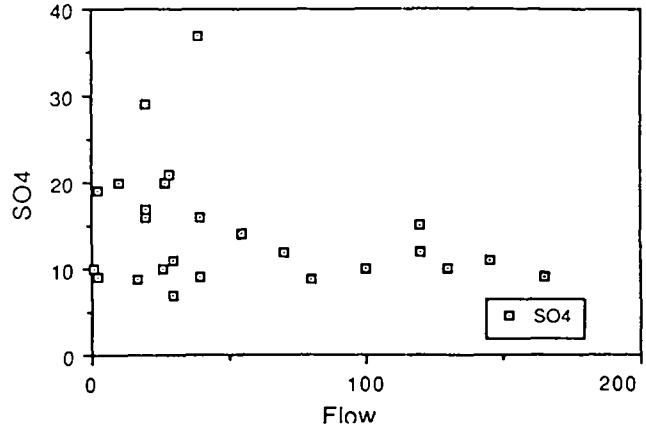

Figure 5. Sulfate as a function of flow. Note higher sulfate values during low flows.

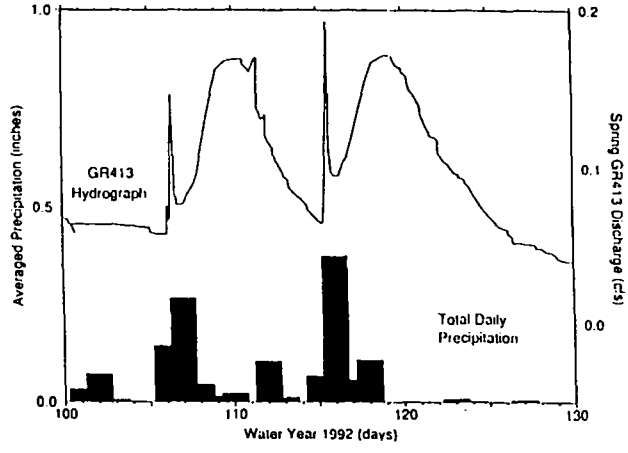

Figure 6. Spring GR-413 response to individual storm events. The initial spikes are from surface runoff (from Abate, 1993).

Table 1. Representative chemical data for the GR-413 spring area (Zahradnik, 1994).

Constituent	B-1	B-2	B-3	B-5	GR-413
SiO_2 (mg/L)	4.7	6.3	4.9	4.2	2.2
Ca (mg/L)	7.9	11.5	4.5	26.7	1.7
Mg (mg/L)	5.2	8.9	3.4	4.8	1.9
Na (mg/L)	4.5	4.5	2.8	1.3	.3
K (mg/L)	3.2	3.1	3.6	3.3	1.2
HCO_3 (mg/L)	6.1	0.0	4.9	14.6	2.4
Cl (mg/L)	3.1	2.3	2.2	2.2	0.7
SO_4 (mg/L)	28.1	97	15.7	14.1	8.9
pH	6.5	3.9	6.5	6.9	5.6
Sp. Cond. (µS/cm)	120	240	80	180	40
Temp. (°C)	11	10	11	9	14

Date of Sampling: Dec. 5, 1993. Oct. 28, 1995 data shows similar relations.

were sampled during collection of background data. Figure 7 is a map showing the Upper Kittanning coal cropline and water sample points.

The specific conductivity of the crop springs and shallow (near crop) wells have lower values (38-62 μS/cm) than the drill holes located toward the middle of the hill where depth to the coal is the greatest (158-221 μS/cm). The pH of the springs is also lower than the wells. NP is negligible in areas with less than 9-12 m of cover, evidently because of weathering (Fig. 8). The higher pH and conductance increase are both coincident with an increase in NP (Fig. 8 and 9). The water in deeper drill holes exhibits higher dissolved solids than in cropline springs and holes with shallow cover, due to the increasing abundance of calcareous minerals with increasing overburden thickness. The spring water is very similar in quality to that of the shallowest drill holes. The high NP strata are freshwater limestones and calcareous shales. The geology of this area is addressed in Brady et al. (1988), where the mine site is refered to as area "C".

Cropline springs have alkalinity between 1 and 9 mg/l. Unfortunately, alkalinity was not sampled in the drill holes, but it is a near certainty that the alkalinities were higher, judging from elevated pH and conductance. This hilltop has been mined and reclaimed and is producing drainage with an average alkalinity of 380 mg/L.

MINE SITE C: Lower Turkeyfoot Township, Somerset County

Mine Site C (Fig. 10) is an example of a site lacking significant calcareous overburden rock within the proposed mine area. This deficiency is shown by the low NP's of the two overburden holes (OB-A and OB-C), and by the low alkalinity of cropline springs, small country bank mines, and water collected from the two overburden holes, which were uncased down to the coal. Near the actual area mined, the highest alkalinity is 11 mg/L from cropline spring SP-14. Two of the springs below the coal cropline have alkalinity as high as 31 mg/L. Apparently calcareous strata exists below the coal.

The two overburden holes encountered predominately sandstone, with coal at 16.4 m and 22.4 m for OB-A and OB-C respectively. The highest neutralization potential in OB-A is 19 tons/1000 tons, the highest in OB-C is 15 tons/1000 tons. The highest percent sulfur (excluding the coal) in OB-A was a 0.3 m binder within the coal bed that had 2.0 percent sulfur (% S). A 1.2 m-thick shale overlying the coal contained 0.5 % S. The highest percent sulfur in OB-C was the shale below the coal, which had 1.7 % S. The next highest sulfur in the overburden is only 0.2 %. The overburden shows the presence of acid-producing strata, but lacks alkalinity producing strata.

Table 3 compares rock chemistry and water chemistry in the two overburden drill holes. The water quality is consistent with the overburden data in indicating a lack of calcareous rocks. Because of the lack of naturally occurring calcareous rocks, an average of 45 tons $CaCO_3$/acre was brought to the site. Additionally, the material with percent sulfur greater than 0.5 % was selectively placed in "pods" that were located such that they would be above the postmining water table. Twenty tons/acre of the alkaline material was placed on the pit floor, with most of the remainder mixed into the high-sulfur spoil pods. Analyses of water from sample point SP-14, within a year after backfilling was completed, show degradation due to mining (Table 3). Evidently the amount of alkaline material added was inadequate to prevent acidic drainage.

Figure 10 includes water quality for some springs, up slope and to the north of the mine site, that have higher alkalinity (35 to 52 mg/L) than cropline springs. The recharge of these springs is stratigraphically and topographically higher than the upper Freeport coal. The Glenshaw Formation, which is the unit from which these springs arise, is known to contain several calcareous marine zones (Flint, 1973). These higher alkalinities may indicate that there is some calcareous material preserved within the shallow groundwater flow system and/or the springs may be receiving some water from a deeper source. Although no deeper well data exists for the Glenshaw Fm. in the immediate area of the mine site, a 40 m deep well 3 km north-northeast of the site, which penetrates the lower portion of the Glenshaw Formation, has alkalinity of 106 mg/L, conductance of 260 μmhos/cm, and sulfate of 9 mg/L (Tom McElroy, 1995, personal communication). Although these data are limited, it appears that wells within the Glenshaw Formation also have higher alkalinity than springs associated with the same strata.

Comparison with Other Parts of the Appalachian Basin

Climate in the Appalachian Basin from Pennsylvania to Alabama is fairly similar. Average annual temperature in the north is ~10° C and in the south >15° C. Precipitation also increases from north to

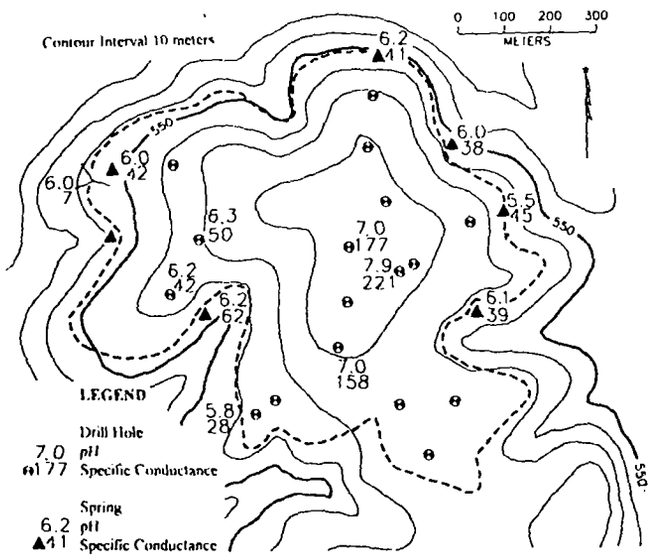

Figure 7. Location of cropline, drill holes and water sample points, and water quality data at mine site B.

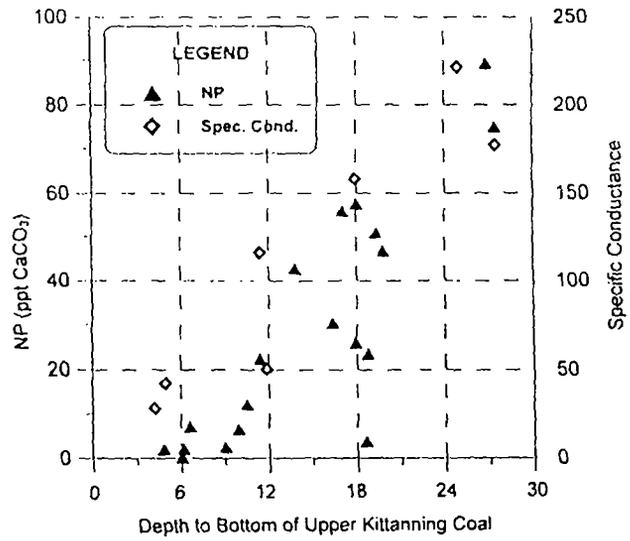

Figure 9. NP and specific conductance (μS/cm) as a function of depth (meters) from the surface to the bottom of the upper Kittanning coal seam for overburden holes (NP), and drill holes and springs (alkalinity).

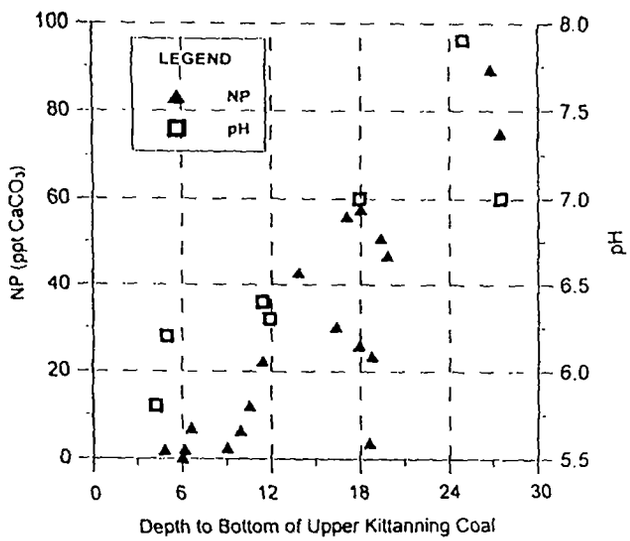

Figure 8. NP and pH as a function of depth (meters) from the surface to the bottom of the upper Kittanning coal seam for overburden holes (NP), and drill holes and springs (alkalinity).

Table 2. Overburden and water quality comparisons for drill holes OBA and OB B on Mine Site C.

Drill Hole	Parameter	NP (ppt)	Alkal. (mg/L)	MPA (ppt)	SO_4 (mg/L)
OB A	ABA (1)	3.5		4.4	
	ABA (2)	0.0		2.4	
	water		5.5		34
OB B	ABA (1)	4.5		2.8	
	ABA (2)	0.0		1.0	
	water		6.5		10

Notes: NP, neutralization potential; MPA, maximum potential acidity. ABA (1) is the traditional method of ABA computation, which includes all determined NP and MPA values. ABA (2) computations were made such that all NP's < 30 and % S's < 0.5 are assigned a value of zero. Methods of computation from Smith and Brady (1990).

Table 3. Postmining water quality at SP-14.

Date	pH	Acidity (mg/L)	Mn (mg/L)	Al (mg/L)	SO_4 (mg/L)
1/12/94	3.7	22	33	35	930
4/21/94	3.8	434	54	54	434

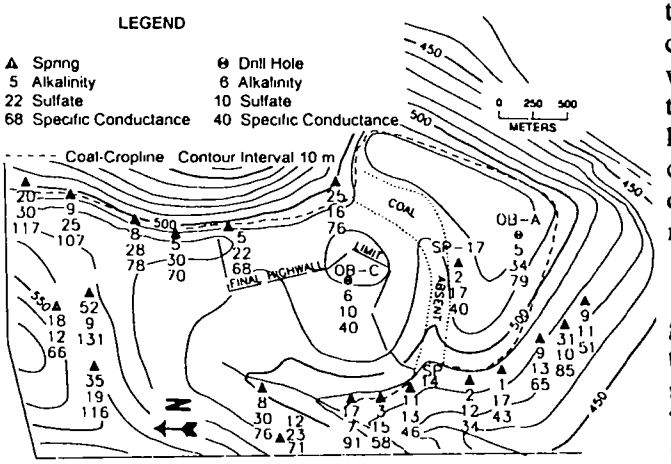

Figure 10. Locations of coal cropline, water sample points, select water quality parameters and overburden holes for mine site C.

south, with rainfall averaging 90 cm in northern Pennsylvania to more than 140 cm in Alabama (Weeks et al., 1968). Annually the infiltration rate of water is sufficient that groundwater is continually flushed through the rock strata from recharge to discharge points. This flow tends to leach out the more soluble minerals. Therefore, shallow groundwater on the Appalachian Plateau has low dissolved solids compared with arid regions.

Geology throughout the Appalachian Plateau is also similar in many respects. The rocks are predominately flat-lying, consisting of sandstone, conglomerate, siltstone, shale, claystone, limestone and coal (Miller et al., 1968). Hills have been subjected to stress-relief forces which have intensified fracturing and subsequently weathering of the slopes, and resulted in higher permeabilities within the weathered zone (Hawkins, et al., 1996). These geologic and climatologic factors result in similar groundwater hydrologic characteristics throughout the plateau.

Numerous studies have investigated the groundwater hydrologic characteristics on the Appalachian Plateau, but very few studies have related the hydrology, groundwater chemistry and rock chemistry. Powell and Larson (1985) investigated water quality in an unmined watershed in a coal-producing region of the Appalachian Plateau of southwestern Virginia. The rock strata are in the Pennsylvanian Norton Formation. The most common carbonate present was siderite, following by calcite and dolomite. Minor amounts of pyrite were generally associated with coal and adjacent strata. They observed that water from springs typically had lower concentrations of alkalinity and dissolved solids than water from dug and drilled wells (Fig. 11). The springs typically have lower alkalinity than dug wells, which have lower alkalinity than drilled wells. Sulfate concentrations are similar for springs, dug wells and drilled wells (with one exception, SO_4 was less than 40 mg/L).

Powell and Larson (1985) envisioned two groundwater flow systems in their study area. One of these "moves under and through the weathered rock or soil layer along the surface of the consolidated rock." The other system "flows through rock fractures and provides the main source of domestic supply" (most domestic supplies were drilled wells). Water in the shallow flow system flows relatively rapidly through rocks thoroughly leached of calcareous minerals. A few of the springs, and many of the deeper waters have high alkalinity (>100 mg/L) (Fig. 11). The high alkalinity springs probably issue from deeper flow systems; however these are the exception rather than the rule.

Singh et al. (1982) in a study of the Mahoning Sandstone in Preston Co. WV, found a weathered zone of about 6 m deep which was low in sulfur and "exchangeable bases" (e.g., Ca and Mg). This indicates that both pyrite and calcareous minerals were removed by weathering within this zone. Brady et al. (1988), in a study that included the area of Mine C of this study also documented the loss of calcareous minerals within ~6 m of the surface. These findings on surface rock-weathering are consistent with the current study.

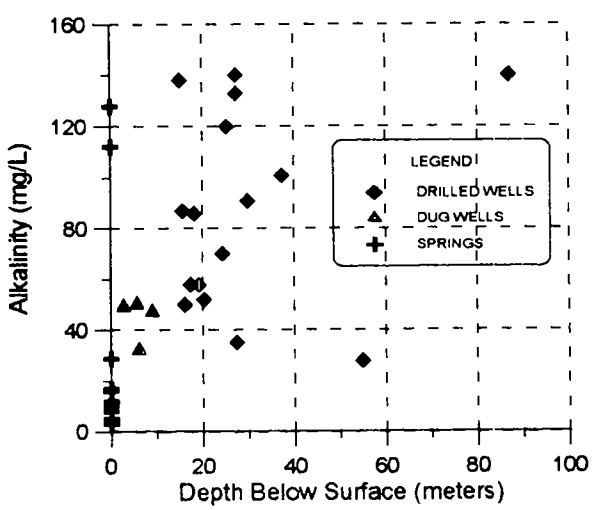

Figure 11. Sulfate as a function of depth of groundwater sample souce in Buchanan Co., VA. Springs are shown at zero depth. Data are from Powell and Larson (1985).

The combined observations of Powell and Larson (1985), Singh et al. (1982), Brady et al. (1988), and this study show evidence for a shallow chemically and physically weathered zone, and two separate shallow flow systems. Figure 5 in the companion paper (Hawkins et al., 1996) is our conceptual model of groundwater flow on the Appalachian Plateau. The literature review in Hawkins et al. (1996) demonstrates the consistency of groundwater hydrologic properties throughout the Appalachian Plateau. Although less research has been done on rock and water chemistry, that which has been done shows results similar to those in PA. It appears that the principals observed in Pennsylvania are applicable to much of the Appalachian Plateau.

Discussion

Surface mining in Pennsylvania mainly occurs within 30 m of the surface and most mines are in groundwater recharge areas (e.g., hilltops). A common misconception has been that water quality of cropline springs in unmined areas is typical of water associated with the coal seam. As illustrated at Mine Sites A and B, water associated with coal- cropline springs can be much more dilute than water from the same coal seam under deeper overburden cover. The cropline springs and shallow wells represent water flowing through the near-surface weathered-rock zone. This weathered-rock zone is quite permeable due to chemical and mechanical weathering. Typically calcareous minerals are absent or negligible within this weathered zone. The rapid flow-through time for the water along with the leached nature of the weathered-rock zone results in water that lacks alkalinity and contains low dissolved solids.

Wells penetrating deeper overburden are completed in unweathered or less weathered materials with lower permeability. Groundwater flow is primarily along fractures and bedding planes. The combination of calcareous minerals, and longer residence time for the groundwater, results in significant dissolution of calcareous minerals forming bicarbonate alkalinity. Downward flow and substantially lower hydraulic conductivities probably result in little of this water reaching the cropline springs.

Mine Site C is an example of a site that lacks significant calcareous rocks. This situation persists from shallow cover to deep cover (maximum overburden that would be disturbed by mining). The lack of calcareous rocks is confirmed by the chemistry of the two overburden test holes and water quality associated with springs, country bank mines, and the overburden drill holes.

From Mine Sites A and B there appears to be a direct relationship between the amount of calcareous material preserved in the overburden and the alkalinity, conductivity and pH of the groundwater. No such relationship exists when comparing MPA (i.e., percent sulfur) with sulfate, probably because of the very limited oxidation of pyrite under saturated conditions. Calcareous minerals are rather soluble in groundwater, whereas pyrite is not. The acid in acid mine drainage is not produced by a simple dissolution process, but by an oxidation process. Pyrite in unmined areas remains largely in an unoxidized state. Premining water quality in deeper drill holes provides a second confirmatory tool, along with acid-base accounting NP, to determine the relative presence or absence of calcareous rock and its distribution within the proposed mine area.

Conclusions and Implications

All of the Pennsylvania sites studied were isolated hilltops within groundwater recharge areas. In all cases the coal cropped out on the sides of the hill. The dominant flow systems for all mines are relatively shallow, with the deepest monitoring wells (completed to the coal seam) on the order of 35 m deep. Where conditions are similar to those given in the examples above, groundwater alkalinity reflects the presence or absence, and abundance, of calcareous rock.

Water quality is directly related to the flow path, the dissolution of material contacted by the groundwater, and the contact time of the water with the rock.

Cropline springs and shallow wells (6 to 9 m deep) that have low or no alkalinity are indicative of shallow leached/weathered overburden. No significant NP's are likely to occur within this zone. Where calcareous rocks are present, such as some deeper cover situations, the calcareous minerals will dissolve in the water and can be measured as alkalinity. Low alkalinity in well or spring water indicates the absence of calcareous strata within the groundwater flow path for that well or spring. It might be expected that sulfate would reflect the amount of pyrite that is present, but there is no relationship between the amount of pyrite (in terms of MPA) and sulfate concentration, thus indicating that pyrite oxidation prior to mining is negligible.

The findings of this study have several important implications:

(1) Coal cropline springs typically reflect very shallow flow through a weathered rock zone and do not necessarily reflect water quality under deep groundwater conditions. Wells are needed to ascertain water quality in the deeper groundwater systems.

(2) Groundwater chemistry, when used with groundwater hydrologic data, can help in better defining groundwater flow systems.

(3) There is a relationship between overburden neutralization potential and groundwater chemistry. If the site is hydrologically isolated such that the only recharge to the site is precipitation, and if alkalinity in wells is high (> 50 mg/L), calcareous minerals are within the flow system and probably near the water sampling point. Where alkalinity is low in wells or springs (< 15 mg/L), the recharge area lacks appreciable calcareous rock.

(5) Groundwater alkalinity can be used to help determine whether overburden sampling for NP has been representative. If overburden analysis does not indicate significant calcareous rocks to be present, but water from wells into the same units are alkaline, the overburden sampling may not be representative of site conditions and additional overburden drilling and testing would be warranted. The combination of groundwater and overburden chemistry can be used together to better define the extent of calcareous overburden.

(6) Overburden sampling and water sampling must represent both shallow and deep overburden cover to adequately represent the entire mine-site hydrology and overburden chemistry. Holes drilled at greater than the maximum cover to be mined may overestimate NP in the overburden that will be disturbed by mining.

(7) There is no relationship between MPA in the overburden and sulfate in the groundwater, or for that matter, between MPA and any other measured parameter. Sulfate in groundwater from unmined watersheds is typically less than 40 mg/L, regardless of flow system.

(8) The above conclusions are probably applicable to a large portion of the Appalachian Plateau. However, the applicability to areas with different climate, physiography and/or geology is unknown.

Acknowledgments

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MINE SEEPAGE PROBLEMS IN DRIFT MINE OPERATIONS

By

Clyde DeRossett, David E. Johnson, and David B. Bradshaw

Abstract. Extensive mining in the Eastern Kentucky Coal Region has occurred in coal deposits located above valley floors. Underground mines present unique stability problems resulting from the creation of mine pools in abandoned works. "Blowouts" occur when hydrostatic pressures result in the cataclysmic failure of an outcrop- barrier. Additionally, seepage from flooded works results in saturation of colluvium, which may ultimately mobilize as landslides. Several case studies of both landslides and blowouts illustrate that considerations should be taken into account to control or prevent these problems.

Underground mine maps and seepage conditions at the individual sites were examined to determine the mine layouts, outcrop-barrier widths, and structure of the mine floors. Discharge monitoring points were established in and near the landslides. These studies depict how mine layout, operation, and geology influence drainage conditions.

The authors suggest that mine designs should incorporate drainage control to insure long-term stability and limit liability. The goal of the post-mining drainage plan is control of the mine drainage, which will reduce the size of mine pools and lower the hydrostatic pressure. Recommendations are made as to several methods that may be useful in controlling mine drainage.

Additional Key Words: mine seepage, landslides, mine blowouts

Introduction

Extensive mining in the Eastern Kentucky Coal Region has occurred in coal deposits located above valley floors. These drift mines present unique stability problems resulting from the creation of mine pools. These pools can cause cataclysmic failure of outcrop-barriers, referred to as "blowouts". Another condition caused by mine pools is sudden increases in seepage from the mine works causing saturation of the colluvium, resulting in landslides. A recent blowout and two landslides were selected for case studies. These studies illustrate seepage problems encountered by drift mine operations, demonstrating the need for planned mine drainage. Recommendations are made as to several methods that may be useful in controlling mine drainage.

The groundwater flow of an undisturbed mountain in Eastern Kentucky is primarily through fractures and colluvium on the exterior portion of the mountain. This flow will recharge perched aquifers during wet seasons. The water stored in the aquifers will provide base flow to the

[1]Paper presented at the 1996 National Meeting of the American Society for Surface Mining and Reclamation, May 18-23, 1996.

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mountain's exterior during the dry season. Wide distribution of stress-relief joints on the mountain, and the existence of perched aquifers help to keep the groundwater flow diffused. For additional study of groundwater flow in the area see Kipp and Dinger (1987) and Harlow and LeCain (1991).

The creation of a mine void will tend to concentrate groundwater, causing a change in groundwater flow patterns . This process is enhanced through fracturing caused by high-extraction techniques. Fractures created by mining provide better conduits into the mine void, draining overlying perched aquifers. Even installation of roof bolts may provide increased flow into the mine, if the drill holes penetrate a low hydraulic conductivity stratum in order to anchor into an overlying unit with a higher hydraulic conductivity. At the elevation of the mine works, groundwater flow changes from fracture-dominated to dip-controlled flow. Drainage into the mine will then be directed towards the mine pools, allowing a build-up of hydrostatic head and creating potential seepage areas when these pools intersect the joint system.

Concentration of Groundwater Flow

Case Study 1

Examination of a mine in Floyd County, Kentucky, is a good example of how a mine void concentrates diffuse groundwater flow. This study was based on an investigation of a slide that occurred in a small hillside drain in Southeastern Floyd County. At this site, two items were studied: a comparison of the volume of discharge from the slide area with the volume of measured rainfall, and a comparison of the discharge from the slide with flow from an undisturbed watershed (Figure 1).

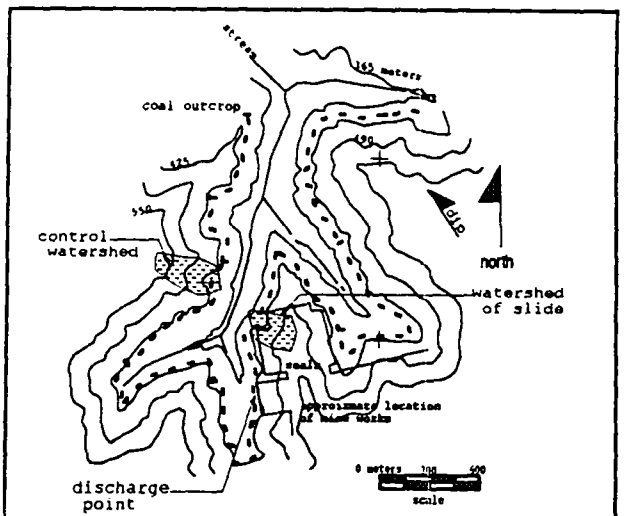

Figure 1. Area of watersheds at case study no. 1

Both the watershed of the slide and the control watershed have similar size and topography, and both were forested with little surface disturbance. The slide's watershed is 4.5 ha. (11 acres) and the control watershed is 5.5 ha. (13.5 acres). No known mining has occurred under the control watershed, but underground mining has occurred in the Amburgy coal seam to the ridge line behind the watershed. The underground works are approximately 305 meters west of the outcrop, and the nearest pillaring is 490 meters to the east. Underground mining in the Amburgy coal seam has occurred under the watershed of the slide. Mine works exist to within 60 meters of the crop line and the nearest pillaring is 80 meters from the crop line.

Monitoring points were established and flow rates were monitored from August, 1994, to May, 1995. In the mine-affected watershed, three monitoring points were established near the base of the slide in order to measure the total discharge leaving the slide area. One monitoring point, that should have intercepted most of the drainage, was established in the control watershed.

During this study, the control watershed seldom flowed, with the only measured flow occurring on May 15,

1995. After establishing the monitoring point, a small point-discharge from the colluvium was discovered below the monitoring point. This point flowed for a few days at an estimated peak flow of 11 liters-per-minute during wet periods of the year, when the colluvium was saturated. Discharge from the point was observed only three times after heavy rainfall events in January, March, and May.

The affected watershed flowed constantly throughout the study, with flow rates ranging from 17.4 to 435.3 liters-per-minute. After one storm in January, 1995, the measured peak flow was 155.0 liters-per-minute, while the control watershed had a flow of less that 11.0 liters-per-minute. Four days after the storm, the flow rate in the affected watershed was 30.3 liters-per-minute, while the control watershed had stopped flowing.

The second item studied was a comparison of the volume of rainfall with the flow measured in the mine-affected watershed. Flow rates and on-site rain-gauge measurements were recorded from September, 1994, to January, 1995. A Belfort rain gauge (Model 5-780) recorded the amount and time of precipitation at the site. Flow was measured intermittently. Results of the volumetric calculations indicated that approximately 63 percent of the measured rainfall volume that fell in the watershed was measured at the monitoring points. This percentage of precipitation as flow is considered low, because the discrete monitoring probably did not always measure the peak flows. The monitoring points only measured a portion of the water coming from the mine void, with an unknown amount being discharged as subsurface flows.

The amount of groundwater coming from the slide area is considered high when compared to studies done by others. Analysis of stream flow hydrographs done in the watershed of Russell Fork near the Kentucky-Virginia border, estimated approximately 18 percent of precipitation was available for groundwater discharge. It was estimated that approximately 50 percent of the rainfall was lost to evapotranspriation (Larson and Powell 1986). This is in rough agreement to earlier estimates done at other large drainage basins in Eastern Kentucky showing approximately 67 percent of the rainfall lost to evapotranspriation (Price 1962). While direct comparison with these large watersheds may not be applicable, they do support the view that seepage seen in the study watershed was more than would be expected from the watershed.

Results from short-term monitoring of the two watersheds clearly showed that mining had affected groundwater flow in the slide area. The mine-affected watershed showed a constant flow that could vary greatly and produce large flow rates not noticed by the landowners before the mining operations. The mine pool feeding the seepage area had good storage, as evidenced by the constant flow rate and a total discharge volume of 222,202 cubic meters (18 acre-feet), from August 1994 to May 1995. High specific conductance measurements, often associated with mine drainage would also support the view that much of the water is coming from the mine. Comparison of the specific conductance taken in the control watershed would support this conclusion (Figure 2).

Mine Blowouts

The most drastic result of improper drainage control is the occurrence of mine blowouts from drift mines. These events can result in the sudden release of force that can endanger the public and cause substantial environmental damage in a few hours. From the spring of 1993 to the spring of 1994, five mine blowouts were studied. All the sites had one

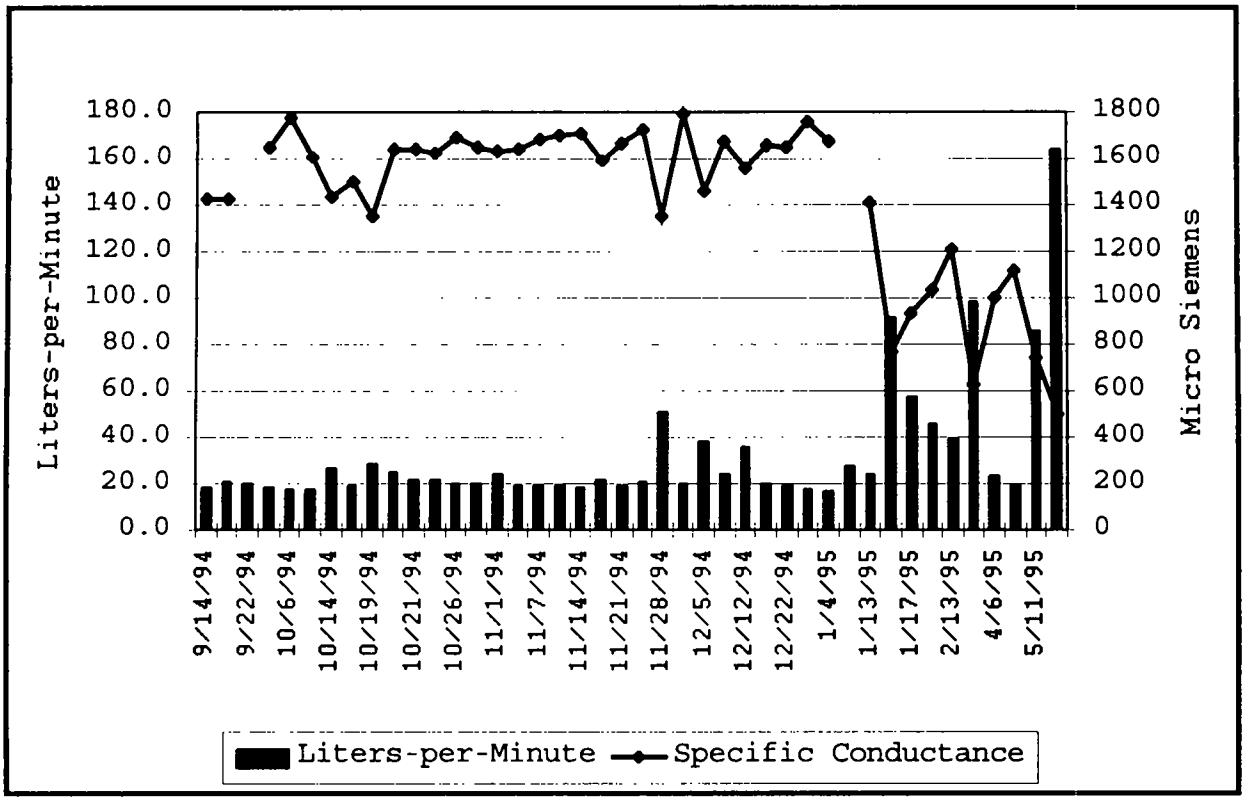

Figure 2. Discharge and Specific Conductance measurements for monitoring point No. 3, Case Study 1.

item in common: the events occurred at excessively weakened outcrop-barriers. These were either at extremely narrow outcrop-barriers or at intersections with small unknown mine works.

Case Study 2

At one study site, the blowout occurred where the outcrop-barrier was approximately 5 meters wide. The resulting blowout formed a gully in less than 3 hours and moved at least 3,600 cubic meters of colluvium. The effects of the sudden discharge blocked a stream, damaged several small bridges downstream, and blocked the road into a community over one mile away. The mine works associated with the blowout covered approximately 113 ha (280 acres) and had been active for approximately 3 years. The mining was completed in 1986, eight years before the blowout.

In addition to the blow out, there were two other small surface discharges located in the area. These were small discharges, one site had only diffused seepage in the area of the portals (no measurements taken). The second seep had a flow rate measured from 19 to 38 liters-per-minute, based on infrequent sampling done over the period from March, 1994, to March, 1996. The layout of the mine was such that almost all of the mine drained to the area of the blowout. The difference in elevation between the high and low points in the mine was almost 15 meters. The low point was located in the area of the seep (Figure 3) with the blowout occurring approximately 9 meters below the highest point of the mine. The two surface discharges were located close to each other and along with the configuration of the mine floor helped to reduce the water level in the mine pool during periods of low inflow, reducing the hydrostatic pressure in the area of blowout.

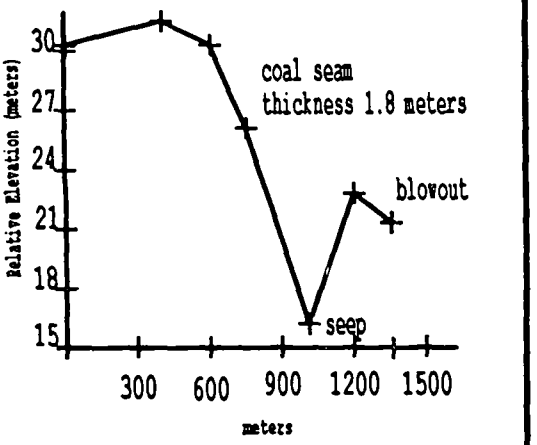

Figure no. 3. Profile of mine floor along drainage path in mine.

Measurements taken after the blowout showed a trend of reduced flow. While no attempt was made to account for the various factors influencing the flow from the blowout area, the measurements do indicate that a considable amount of water was stored in mine voids (Figure 4). Measurements over the period from February, 1994, to September, 1995, showed a downward trend indicating a reduction of storage in the mine. The main lessons to be learned from the study of this site were that control of the mine's outcrop-barrier and the size of the collection basin is necessary in order to lessen the chance and severity of the mine blowout.

Study of the dataset would also indicate that none of the mine blowouts occurred in areas where modest amounts of outcrop-barrier were maintained to resist the pressure. None of the cases examined indicated the Ashley or Mine Inspector Formula was invalid. This would support the use of the formula for the design of outcrop-barrier widths. However, the Ashley Formula only indicates the needed coal-pillar width (Chekan 1985). The coal seam may not be the structurally weakest member containing the mine pool, due to the topography of the hillside.

Mine Seepage Problems

The second stability problem associated with mine drainage is the saturation of outslopes resulting from seepage through outcrop-barriers. This problem is much harder to control than mine blowouts. Seepage from mine pools will result in saturation of colluvium on the outslope, sometimes resulting in landslides. Flow from mine pools to seepage areas is controlled by hydraulic conductivity along the outslopes. The greatest hydraulic conductivity is associated with the stress-relief joint-system. Studies have shown that hydraulic conductivity decreases greatly toward the center of mountains. Studies done in Virginia (Harlow and LeCain, 1991) have shown a large reduction of hydraulic conductivity within 31 meters for most rock strata, except coal, from the outcrop. Stress-relief joints are more open near the surface and close rapidly with increased overburden.

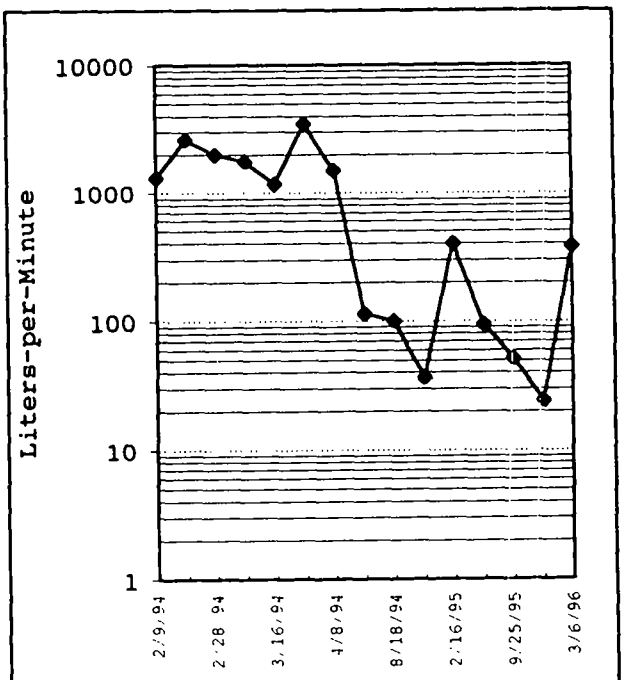

Figure 4. Flow Measurements From Blowout in Case Study No. 2.

A narrow outcrop-barrier will result in a better connection between the joint system and the mine pool. A chart showing a cumulative percentage curve of the width of the outcrop-barrier at the areas of the slides studied is shown in Figure 5. In these studies, the distance from the slide to the mine works has a mean of 45 meters, with a standard deviation of 8.5 meters. The study indicates that outcrop-barrier width is very important in the prevention of slides. However, maintenance of an outcrop-barrier alone will not always guarantee prevention of seepage problems. In case study number one, examination of the mine map indicates that a barrier of 60 meters was maintianed around the whole panel and mine pool. As suggested by Harlow and LeCain, 1991, hydraulic conductivity of the coal seam would decrease with increased barrier width due to the increased overburden depth. The decrease in hydraulic conductivity may allow a build up of the hydrostatic head, thereby increasing the flow through fractures that intersect the mine pool. Though increasing the outcrop-barrier will increase the head loss to the surface-seepage areas, resulting in a reduced flow, their use will not always assure prevention of seepage problems. If that area of the mine has enough inflow, the increase in depth may still provide enough hydrostatic head to cause seepage problems. While the use of increasing outcrop-barrier widths at sensitive areas may prove to be an effective tool, the main objective should be to control the size of the mine pools.

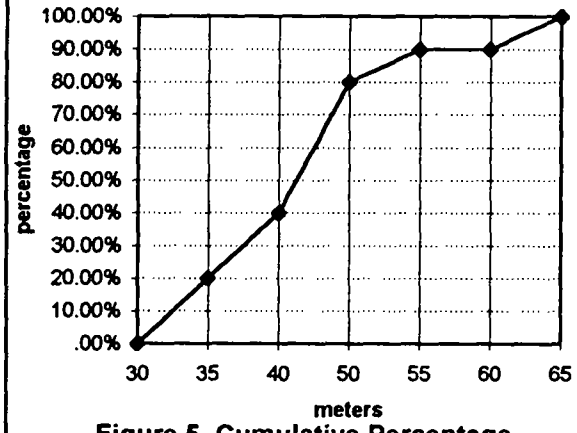

Figure 5. Cumulative Percentage Curve of Barrier Width at Landslides. Data set = 10.

From the case studies, it is clear that control of mine seepage must address both the maintenance of adequate outcrop-barriers width and control of the sizes of the mine pools. The control of the size of the mine pool can be either by planned drainage of the mine or by limiting the size of the drainage area, preventing most of the drainage from collecting into one large pool. Restricting the size of the mine drainage area should be a neccessary part of mine planning. The neglect of this has lead to serious seepage problems.

One method that can be used to control drainage is by planning the layout of barrier pillars in ways that divert drainage to various sections of the mine or to discharge points. While these pillars may allow considerable seepage, their use to divert flow to other areas is considered useful. Mine seals may also play an important part of the mine drainage plan. However, consideration needs to be given to ensure the long-term stability of these structures. Placement of the barrier pillars and mine seals along with the local dip of the coal seam could greatly aid in the control of mine drainage. A good example of this is demonstrated in Case Study 3, where seepage was observed on both sides of the of a small hollow.

Case Study 3

In Case Study 3 the mine map indicates two mine pools. One pool shows a possible collection basin of 40 ha (100 acres) and the other pool is

basically limted to one panel, approximately 8 ha (20 acres), under the ridge. This panel is isolated by the barrier pillars and seals separating the sub-main from this panel. Sampling at several monitoring points was done to investigate seepage conditions. On the side of the large drainage area, the mine pool resulted in a slide and later a blowout. The seepage on the other side of the hollow, while not planned, has not resulted in a slope stability problem. Study of mine maps and monitoring points give some indications as to the differences in the problems seen at the two sides of the mountain.

Sampling was initially started at three sample points; however, one point was removed during stabilization of the slope. Two monitoring points remained allowing continued monitoring of the two mine pools. The most significant finding from the sampling was in the difference in the seasonal fluctuations of the two monitoring points and the reflection of the change of the flow rate at monitoring point 2 (Figure 6), which resulted from reduced hydraulic head caused by a blowout. Monitoring point 2, at the larger mine pool, showed less fluctuation of flow rate, except for one surge in flow due to precipitation. This surge in the flow showed large drops in specific conductance (Figure 7). This observation gives a good indication of how near surface groundwater (with lower specific conductance) affects the two seepage areas. Monitoring point 3 at the small drainage area shows a large downward trend in the flowrate during the dry time of the year. Results of the trends in the amount of seasonal fluctuation of the monitoring points indicated that the volumes of the mine pools are different as suggested by the mine map. The mine pool feeding monitoring point 2 showed less change in the hydraulic head as indicated by a more consistent flow rate, measured at the discharge point. Monitoring at sampling point 3 indicates a larger change in the hydrostatic head. The mine pool at monitoring point 3 is fluctuating more, presumably as a result of less storage capacity and/or lower inflow rates. In April, 1994, a blowout occurred in the area of monitoring point 2 reducing the flow measurements (Figure 6).

The Correlation Coefficient (r)

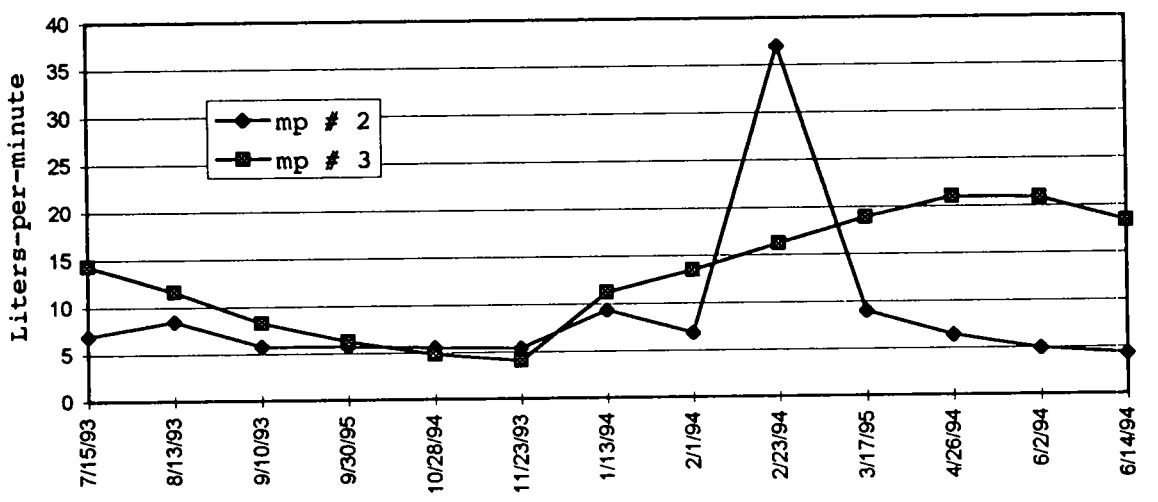

Figure 6. Comparison of Monitoring Points 2 and 3 in Case Study No. 3

was also calculated for the flow rates (the sample set on February 23, 1994, was not used because of the influence of the storm event). The r value for the data set before the blowout had a value of 0.75. The r value for all the data set had a value of 0.09. This change in the correlation of the two monitoring points would indicate that the blowout had affected a change in the hydrostatic head of the mine pool at monitoring point No.2. While the data set is limited and details of the actual drainage conditions and mine pool size are not available, this case study still demonstrates how control of mine drainage areas are important and that planning of the mine layout can reduce the size of the mine pools.

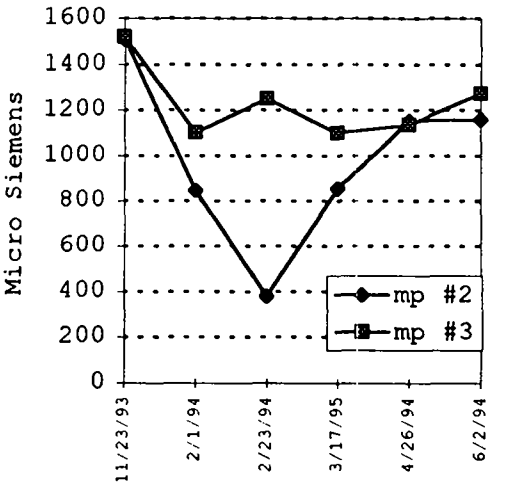

Figure 7. Specific Conductance Values for Case Study No. 3.

Conclusions and Recommendations

These case studies are not detailed investigations of mine drainage control; however the studies show the need to control mine drainage. The study indicates the need to have flexible systems that can take advanage of structural geology, outcrop-barrier width, and drainage points. Planning for drainage control must be flexiable enough to take advantage of the unique conditions that are present in each mine. In planning mine drainage the main concern is in limiting the hydrostatic pressure of the mine pool. Several tools are avialable for control of drainage both into and out of the mine pool. Specifically, the study has found:

*The mining operations will concentrate groundwater flow to specific areas of the mine. Failure to plan for the change in the groundwater flow can result in dangerous and costly environmental problems.

*Outcrop-barrier widths are important in the control of mine seepage. The outcrop-barrier widths seem useful in reducing the hydraulic gradient, causing less flow to specific areas at the outcrop. In areas where the mine pools develop close to the stress-relief fracture system seepage problems will increase. However, the use of a minimum barrier width will not assure prevention of landslides.

*Discharge points provide a method of reducing the volume and hydrostatic pressure at the mine pools. Possible discharge points should be considered where feasible to reduce the amount of water stored in the mine area. These discharge points can provide improvements to the landuse area when the water quality is adequate.

*Study of the mine maps and the flow measurements can give some indications of the mine pool conditions. While this data may be limited in detail on exact drainage areas and area and depth of the impoundment, useful information of the changing conditions of the mine pools can still be obtained. This information may be useful in planning of abandoned mine projects and assessment of potential hazards to the public, or when investigating the potential undermining of mine void.

*The use of barrier pillars and seals needs to be investigated as structures

to reduce the size of the collection basin draining into the mine pool.

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EFFECTS OF SPOIL GROUNDWATER ON WATER-QUALITY IN A RECEIVING LAKE

by

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Abstract. A groundwater and surface-water study was conducted in order to determine the source(s) which are adversely affecting a 35-acre lake in southeastern Ohio. The lake has a pH of 3 and is flanked on the north by about 100 acres of recently reclaimed spoil, and on the south and west by an area of similar size that has not been reclaimed since surface mining ceased over thirty years ago. Inflows into the lake from the north include three major seeps and the outlet of a limestone drain installed early in the recent reclamation. These inflows contribute only about 5 to 10 percent of the total inflow into the lake but have very poor water-quality, with acidities approaching 1800 mg/L as $CaCO_3$. The other major inflows originate to the south and west of the lake and include three streams and Spring U, which account for nearly 70 percent of the total inflow to the lake, but which have lower acidities ranging from -15 to 250 mg/L depending on the time of year. The remaining inflows include groundwater that seeps into the lake. Although the lake receives inflows from other sources, we believe that the groundwater contribution is a significant factor in overall lake-water quality. Coal-seam-elevation data, monitoring-well surveys and estimations from old topographic maps indicate that the dominant groundwater flow is from old underground mines and auger mining north of the site.

Additional Key Words: Spoil, Highwall, Coal Mine, Acid Mine Drainage

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Introduction

Acid mine drainage may result from a variety of mining methods performed in a watershed - underground mining, strip mining, and auger mining. The mining process exposes iron sulfide (pyrite) in the sandstone overburden and exposes faces of unremoved coal to air and water. The oxidizing conditions result in a lowering of the pH, increase in acidity, and increase in dissolved metals, leading to an overall degradation of water-quality and the inability to support aquatic life.

The degradation of Howard Williams Lake is the result of surface and underground mining of coal. The pH of the lake varies from 2.9 to 3.5 depending on the time of year and has no signs of any aquatic life. The objectives of our work include determination of the sources of acid water and engineering approaches to improve lake-water quality.

Background and Objectives

Howard Williams Lake is a 0.14 square kilometer (35 acre) reservoir located in Perry County, in southeastern Ohio. It was built before 1950 to provide water for a coal-washing plant farther down the valley. The lake, created by damming the valley, collects water from a 1554 square kilometer (6 square miles) watershed. The lake has had a low pH, around 3.0, for at least two decades, with acidity ranging from 200 to 300 mg/L as $CaCO_3$.

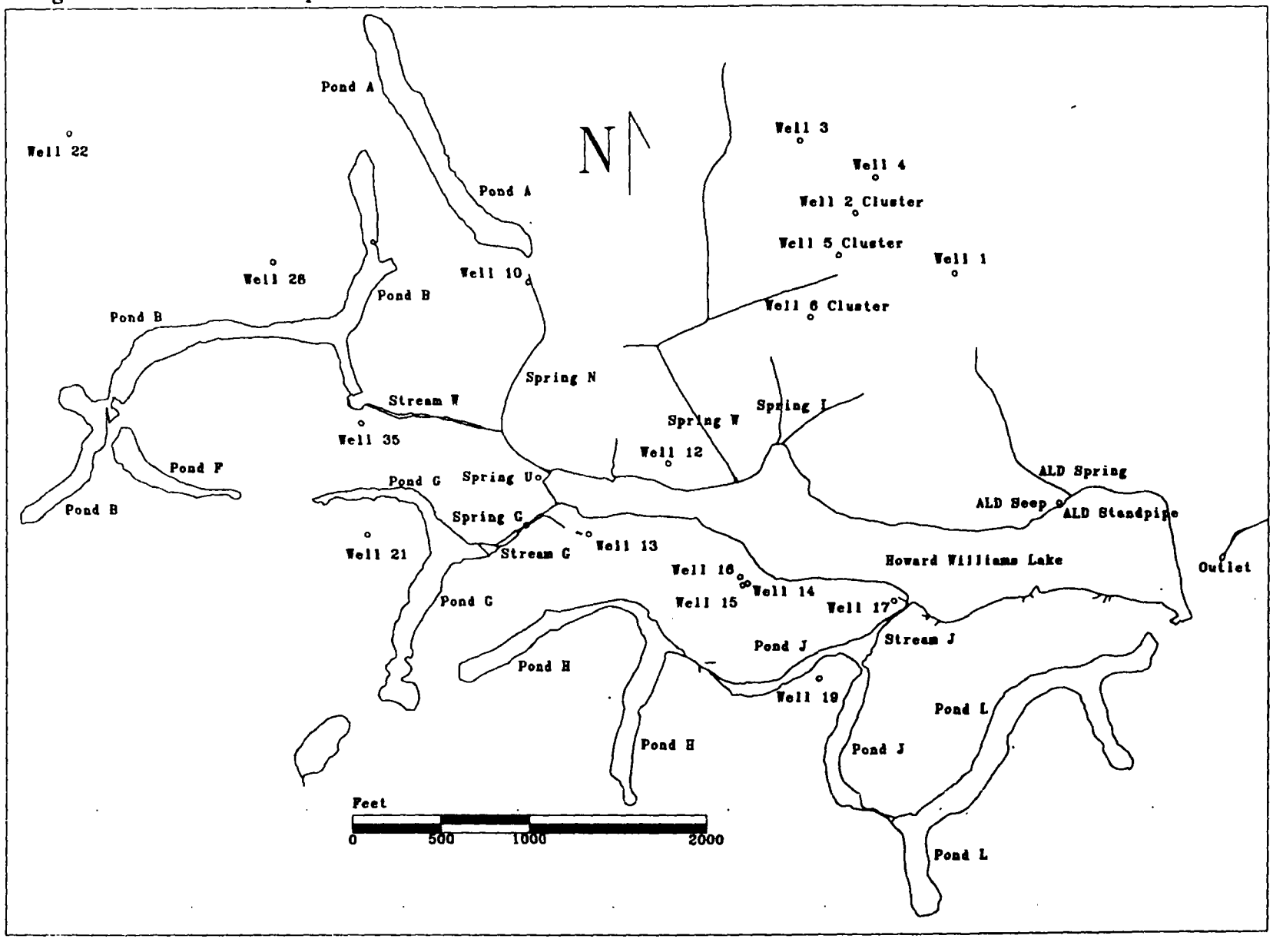
The economic coal seam in this region is the Middle Kittanning (No. 6), which averages about 1.4 meters thick (4.5 feet), with local variations. The outcrop was stripmined by dragline, and auger methods were used at some locations north and south of the lake. The auger system used the "push-button-miner", or PBM, which created rectangular borings about 12-ft in width, rather than the round boreholes common with other augering machines. Coal was also recovered by underground room-and-pillar methods until the late 1960's.

No reclamation of the surface disturbances was done until 1990. At that time the State of Ohio reclaimed about 100 acres north of the lake, primarily in areas where old township roads terminated at the tops of highwalls of about 30.5 meters (100 feet) in height. The spoil was regraded to a rolling topography from the old highwalls to the lake. Compost additions, papermill sludge additions, and revegetation seeding resulted in a current good ground cover of grasses in this reclaimed area.

Acidic waters enter this lake from (1) seeps and surface drainage from the reclaimed area north of the lake, (2) seeps from unreclaimed spoil piles and surface drainage from water ponded in old strip pits south of the lake, and (3) surface drainage from old ponded pits west of the lake. The overburden of the No. 6 seam, and the coal seam itself, contain from a few tenths percent up to several percent sulfur, as pyrite, with very little alkaline material.

The basic mechanism that produces acid mine drainage is the oxidation of pyrite. Pyrite dissolves to form ferrous iron, acidity, and sulfate in equation (1). (Donovan *et al*., 1994; Lapakko, 1994) Both reactions take place in the presence of oxygen and water and are catalyzed by the bacteria *Thiobacillus ferrooxidans*. (White *et al*., 1994)

Figure 1. Site Map.



$$FeS_2 + 7/2O_2 + H_2O \Rightarrow 2SO_4^{2-} + Fe^{2+} + 2H^+ \quad (1)$$

Ferrous iron is then oxidized to ferric iron in the presence of oxygen and it consumes acidity.

$$Fe^{2+} + 1/4\ O_2 + H^+ \Rightarrow Fe^{3+} + 1/2\ H_2O \quad (2)$$

Equation (3) shows that ferric iron in solution will form iron hydroxides and produce acidity, but this is a very slow reaction in waters with a pH below 3. At higher pH values this reaction is very fast, resulting in noticeable quantities of iron hydroxides precipitating out of solution.

$$Fe^{+3} + 3H_2O \Rightarrow Fe(OH)_3 + 3H^+ \quad (3)$$

Methods

Mine maps were obtained from the U. S. Bureau of Mines showing the extent of underground mining in the regions surrounding the lake. A digitized map, derived from aerial photos taken in 1994, of the area around the lake was obtained from the Ohio Department of Natural Resources. Other aerial photos taken at various stages of the surface mining operation were also examined. These, and other anecdotal information, described the deposition of a large volume of coal-washing wastes in a pit along a highwall north of the lake. This waste deposit was covered during the 1990 reclamation; estimates of the volume of this material are about 245,000 cubic meters (320,000 cubic yards).

Study of old maps, photographs, and intensive walk-through of the mined land, highwall exposures, and spoil piles provided definition of the surface waters flowing into the lake. It was expected that surface mining intercepted old deep-mine tunnels, but the bases of all current highwalls are covered with slumped material and water in abandoned strip pits. This prevented clear determination of possible hydraulic connections between underground mined areas and surface water in strip ponds.

Mine maps provided data on the extent of mine voids underneath the rolling hills of the region, underclay elevations, coal-seam thickness, and localized geologic sections. Figure 1 shows the major features of the area around the lake, with individual strip-pit ponds identified.

In early 1994, twenty-four groundwater monitoring wells were installed with the assistance of the Ohio Department of Natural Resources' equipment and staff. Figure 1 shows the locations of these wells. Well clusters were installed at No.2, No.5, and No. 14 to provide opportunities to perform medium-scale hydraulic tests of the minespoil. All wells are either 2-in or 4-in diameter and are screened either 1.5 or 3.0 meters (5 or 10-ft) above the underclay. Well Nos. 4, 19, 21, and 28, were located in attempts to intercept underground mine rooms or auger-mine voids. This was in order to determine water-quality, flow direction, and the extent to which these subsurface voids are flooded. An underground mine room was found at well 21 and an auger opening was found at well 4. Both well Nos. 22 and 28 were drilled into unmined coal.

Well Nos. 1, 2, 3, 5, 6, 10, and 12, are located in recently reclaimed spoil north of the lake. Well Nos. 13, 14, 15, 17, and 35, are located in unreclaimed spoil south of the lake. Wells 1, 2, and 3, are located in the coal-cleaning waste buried in a pit north of the lake.

Surface-water inflows to the lake have been sampled for water-

quality analyses and flow rates since fall, 1994. Chemical analyses were conducted by a commercial laboratory using standard methods. Flow rates were determined by diverting flows through a 6-in polyvinyl chloride (PVC) pipe and measuring amounts captured in a bucket during a time measured by stopwatch. Flow measurements were made at least in triplicate and the results averaged. The October 2, 1995, data is shown in Table 1 and represents typical water quality data for the surface inflows.

Results

Determining whether or not the coal seam is inundated is a very important aspect of the site. The mine floor in well 21 is at an elevation of 281.16 meters (922.45 feet), the water table is at approximately 283.51 meters (930.15 feet), and with a relatively flat coal seam and a thickness of 1.22 meters, it can be assumed that the underground mine is completely flooded. The direction of flow is towards Pond G, located at the base of a highwall 15 to 23 meters (50 to 75) feet north of the well (Figure 1). Pond G has an average water elevation of 281.7 meters (924.1 feet). So, there is about 1.8 meters (6 feet) of head driving the water from the old underground mine voids towards Pond G.

Mine maps showed possible auger mining northeast of well 2 and adjacent to Pond J. Well drilling penetrated an auger opening in the highwall above well 2, and a 2-inch observation well was placed there (well 4). The well is 32.6 meters (107 feet) deep and rests on top of the underclay. The auger-mining void was completely flooded. The water level was 286.6 meters (940.4 feet) and the water level in well 2 was 284.0 meters (932.0 feet), which indicates that the auger-mining voids are a source of recharge for the coal-cleaning waste pit.

Several attempts were made to intercept auger openings in the highwall above Pond J. Every attempt resulted in hitting unmined coal. One well, well 19, was left in and surveyed to obtain water-level

Table 1. Water-Quality and Flowrates for Flows into and out of Howard Williams Lake for October 2, 1995.

Location	pH	Total Acidity (mg/L as CaCO3)	Sulfate (mg/L)	Total Iron (mg/L)	Total Manganese (mg/L)	Aluminum (mg/L)	Flowrate (gpm)	Influx of Acidity (kg/day)
ALD Spring	2.93	1672	3860	600	79	60	6.5	53.9
ALD Stream	2.95	1394	3081	460	60	69	2.5	17.2
Standpipe	5.61	-22	3068	520	60	1	1.7	-0.2
Spring W	3.00	368	2140	106	44	15	4.9	8.9
Spring I	3.95	790	3002	417	64	23	1.2	4.6
Spring U	4.15	227	1288	83	25	13	83.5	94.7
Stream W	7.60	-106	230	<1	<1	<1	33.7	-17.8
Spring N	3.00	1093	2800	235	37	48	<.5	0
Stream G	6.02	-19	453	12	2	2	59.3	-5.6
Spring G	3.15	250	1255	76	19	18	18.4	22.9
Stream J	2.78	418	1473	54	31	17	21.4	44.7
Outlet	3.00	305	1236	24	23	13	234	357

elevations. The water level in well 19 is around 279.3 meters (916.2 feet) and Pond J is 278.1 meters (912.4 feet). So the strip ponds to the south of Howard Williams Lake are receiving recharge from the unmined coal seam and auger openings. This connection was verified with a qualitative tracer test using fluorescein dye. A positive result was obtained in Pond J within a week of dye injection into well 19.

Two more wells were installed in areas to the northwest of Howard Williams Lake that are thought to be affected by underground mining. Wells 22 and 28 are located above Pond A and B. Water level elevations for well 22 and 28 are 285.9 and 284.1 meters (938.0 and 932.0 feet), respectively. Both of these wells have about 16 feet of water above the underclay, thus the coal seem is also completely submerged in this area. No underground mine rooms were intercepted with these wells, but rooms do exist in the area according to the mine maps.

The remainder of the wells were placed in the strip mine areas around the lake. Wells 1, 2, 2A, 2C, and 3 are all in an old strip pit occupied by coal-cleaning waste. The mine waste has a thickness of about 18 meters (60 feet), with the bottom 5 to 6 meters (15 to 20 feet) being submerged. According to water-level elevations, the water in the pit tends to move to the southeast towards Howard Williams Lake. It may be possible that the water in this pit is seeping out at the seeps near the anoxic limestone drain (ALD).

Wells 13, 14, 15, 16, 17, and 35, are all in the unreclaimed spoil areas to the south and west of Howard Williams Lake. All of these wells are very shallow and are located near the base of the spoil piles. Wells 10 and 12 are also shallow wells, but have been placed in the reclaimed spoil areas. Well 10 is near Pond A and well 12 is within 100 feet of the north shore of Howard Williams Lake. Wells 5, 5A, 5B, and 6A are located to the southwest of the coal-cleaning waste pit in the reclaimed spoil.

Water Budget

There are several surface inflows into Howard Williams Lake. The quality and quantity of these inflows vary considerably. The lake discharges through a drop-inlet in the southeastern side of the lake. The drop-inlet then drains into a 122 centimeter (4 feet) diameter concrete pipe which passes through the dam and drains to the east of the lake. Flowrate measurements for the outlet of the lake have been done using Manning's equation, since normal flow exists in the pipe (Chow, 1959). By using Manning's equation, the only field measurement needed is the width of flow. The pipe has a slope of 0.026 m/m, diameter of 122 centimeters (48 inches), and the Manning's roughness coefficient has been estimated at 0.018 for this pipe.

All of the surface inflows to the lake were low enough to be measured using a bucket and a stopwatch. At least three trials were done on each inflow to get an average value. The location of the inflows are shown in Figure 1 and the average inflow rates are shown in Table 2. Stream J discharges water from Ponds L, J, and H, and enters the lake through a small culvert underneath an old haul road on the south side of the lake. Seep G is a seep that converges with Stream G and enters the lake though another culvert that runs underneath the haul road on the south side of the lake. Spring U is a seep about 0.3 meters (1 foot) in diameter that is located about 5 meters west of the west edge of the lake and is between the haul

Table 2. Average Flowrates.

Location	Flowrate (gpm)
ALD Spring	4.96
ALD Stream	1.87
Standpipe	1.50
Spring I	2.24
Spring W	5.08
Spring N	0.55
Stream W	27.66
Spring U	76.72
Spring G	19.30
Stream G	79.80
Stream J	27.79
Outlet	295.00

road and the lake. Stream NW has two forks - the west fork is runoff from Pond B, Spring N is fed by a seep coming out of the reclaimed spoil created by the reclamation to the north. Both forks flow through a limestone-lined channel installed during reclamation. Spring W, Spring I, Spring N, ALD Stream drain seeps originating in reclaimed spoil north of the lake and flow through limestone-lined channels installed during the reclamation. ALD Spring and ALD Stream, which drain seeps, are adjacent to an anoxic limestone drain that drains out a standpipe located near the shore of the lake. The streams are all located in swales created by the reclamation. The limestone channels are heavily covered with iron oxides. The channels also carry very little surface runoff and have not shown any scouring effects during storm events.

Water-Quality

Water sampling began in the fall of 1994 and has continued through 1995. Initially only the surface inflows to, and outflows from, Howard Williams Lake were sampled. This was expanded to include all of the monitoring wells as they were installed throughout the site. The goal of the water-quality analysis is to determine the areas around Howard Williams Lake that are contributing to the poor water-quality conditions found in the lake, so that an economical and effective restoration plan can be designed.

All inflows and outflows have been sampled several times over the last year and a half. Attempts were made to sample during wet and dry periods to see what effect rainfall had on water-quality and flow. A few sample sets were taken after rainfall events, but most of the sample sets were primarily taken during low flow periods. After rainfall events, flowrates of streams connected to ponds increased while their acidity decreased. Springs had no noticeable change in flowrate or quality.

Discussion

From Table 1 it can be seen that the major sources of acid mine drainage are ALD Spring and Spring U. These two sources of water account for only 32% percent of the surface flow entering the lake, but contribute 63% of the acidity.

ALD Spring is believed to be the result of recharge entering the spoil and flowing through zones of high pyritic content. A gob pit to the northwest of ALD Spring and underneath wells 1, 2, and 3, has been proven to be in direct connection to ALD Spring by using particle tracking in Visual Modflow (Guiguer and Franz, 1996).

The source of water for Spring U is Pond G, which is believed to be connected to an underground mine opening. Pond G is at an elevation of 281.7 meters (927.69 feet), while Howard Williams Lake is at 278.5 meters (913.8 feet). Pond G is separated from Howard Williams Lake by a spoil pile. Stream G carries water from Pond G over a series of

beaver dams and into Howard Williams Lake through a culvert. The beaver G and the lake. The large difference in head between these two bodies of water is believed to be the driving force behind two seeps, Seep G and Spring U, that exit the spoil pile at or near the elevation of the lake. The seeps are believed to be the result of pseudokarstic conditions typical of mine spoil (Aljoe, 1994). Table 3 shows how the water-quality changes as it goes from the underground mine rooms through the highwall into Pond G and eventually into the lake via Stream G and Spring U.

As can be seen from Table 3, the water that seeps through the spoil pile deteriorates substantially before it exits the spoil pile. Acid-base accounting data for the spoil adjacent to Pond G is not known, but core samples from well 21 give a good indication as to the quality of the overburden (Table 4). Well 21 is located about 30 meters from the edge of the highwall that is adjacent to Pond G. Based on stratigraphy from drilling records and tables, well 21 has a height-averaged pyritic sulfur content of 0.24 percent, with a potential acidity of 7.55 g/kg, and a neutralization potential of 0.43 g/kg ($CaCO_3$). These values are representative of the spoil adjacent to Pond G. Sobek *et al* . (1978), defined geologic material as being potentially toxic if the pH is less than 4.0 or the material has a net $CaCO_3$ deficiency of 5 grams or more dams have created a head difference of 5 meters (13.9 feet) between Pond per kilogram of material. For well 21, the only material that is considered potentially toxic is the Freeport Coal, but this seam is very shallow and often non-existent in the highwalls. The gray sandstone, which makes up a good portion of the overburden, can be considered potentially toxic due to its $CaCO_3$ deficiency, but not its pH. Pyrite contents are not particularly high, but result in toxicity due to a general lack of alkalinity in the overburden.

Stream J is another major source of acid mine drainage. Pond H and L overflow into Pond J, which drains out Stream J to Howard Williams Lake. There are two sources of recharge to these ponds: spoil groundwater and the highwalls. A connection exists between the highwall and Pond J, as proven recently with a tracer test. However, adjacent spoil is the major cause of poor water-quality in Pond H, J, and L. By comparing water-quality data for Stream J, well 17 (in spoil) near Stream J, and well 19 (highwall), the water-quality of Pond J is related to the water-quality in the spoil. The percentage of water that each source contributes to Pond H, J, and L is not clear at this time. The quality of the overburden was obtained from acid-base accounting data for well 21. The height-averaged pyritic sulfur content is 0.24 percent with a potential acidity of 7.55 and

Table 3. Effects of Spoil on Typical Water Quality from Well 21A to Spring U

Location	Head (ft)	pH	Total Acidity (mg/L as CaCO3)	Sulfate (mg/L)	Total Iron (mg/L)	Manganese (mg/L)	Aluminum (mg/L)
Well 21A	929.7	6.18	-4.67	266	19	2.02	0.69
Pond G	927.7	6.66	-48.50	95	1	0.90	0.10
Stream G	922.1	4.94	32	492	15	4.43	3.08
Spring G	915.2	3.27	308	1401	79	25	21
Spring U	915.2	3.98	419	1438	82	28	15

Table 4. Acid Base Accounting Data

Location	Depth (m BGS)	pH	Material	Neutralization Potential ton/kton as CaCO3	Total Sulfur (%)	Pyritic Sulfur (%)	Potential Acidity total sulfur*	Potential Acidity pyritic sulfur*	CaCO3 Deficiency total sulfur*	CaCO3 Deficiency pyritic sulfur*
Well 3	4.6	2.87	Gob	-2.35	6.60	4.94	206.3	154.4	208.6	156.7
Well 3	9.1	3.39	Gob	-1.05	6.00	4.82	187.5	150.6	188.6	151.7
Well 3	10.7	4.27	Gob	3.67	4.81	3.90	150.3	121.9	146.6	118.2
Well 3	13.7	4.35	Gob	4.40	10.75	8.61	335.9	269.1	331.5	264.7
Well 3	18.3	4.06	Gob	-2.87	4.72	3.10	147.5	96.9	150.4	99.7
Well 3	22.3	3.41	Gob	-0.20	1.37	1.03	42.8	32.2	43.0	32.4
Well 5	3.0	4.10	Spoil	0.50	0.24	0.06	7.5	1.9	7.0	1.4
Well 5	6.1	4.23	Spoil	-0.25	0.50	0.36	15.6	11.3	15.9	11.5
Well 5	12.2	4.35	Spoil	-0.25	0.29	0.24	9.1	7.5	9.3	7.8
Well 5B	5.5	3.61	Spoil	-2.00	0.40	0.03	12.5	0.9	14.5	2.9
Well 6	9.1	5.38	Spoil	9.99	0.60	0.53	18.8	16.6	8.8	6.6
Well 6	12.2	4.82	Spoil	10.70	0.86	0.80	26.9	25.0	16.2	14.3
Well 6	18.3	2.10	Spoil	-75.70	20.00	11.50	625.0	359.4	700.7	435.1
Well 6	18.6	2.67	Spoil	-9.74	3.26	2.07	101.9	64.7	111.6	74.4
Well 6	19.5	5.10	Spoil	0.50	1.02	0.98	31.9	30.6	31.4	30.1
Well 10	6.1	4.91	Spoil	2.25	0.30	0.07	9.4	2.2	7.1	-0.1
Well 10	9.1	3.43	Spoil	-75.70	2.00	1.23	62.5	38.4	138.2	114.1
Well 19	13.1	5.99	WSS	4.40	0.01	0.01	0.3	0.3	-4.1	-4.1
Well 19	22.3	6.85	WSS	6.10	0.34	0.33	10.6	10.3	4.5	4.2
Well 19	25.3	6.59	USS	24.20	0.27	0.26	8.4	8.1	-15.8	-16.1
Well 19	28.3	5.68	USS	19.50	1.29	0.57	40.3	17.8	20.8	-1.7
Well 19	31.4	5.39	Shale	12.90	1.87	1.32	58.4	41.3	45.5	28.4
Well 19	32.9	6.50	Shale	11.00	1.48	0.80	46.3	25.0	35.3	14.0
Well 19	34.7	3.52	Coal (K)	-3.75	4.00	2.18	125.0	68.1	128.8	71.9
Well 21	7.6	5.21	Shale	22.00	0.36	0.36	11.3	11.3	-10.8	-10.8
Well 21	8.8	3.10	Coal (F)	-27.50	8.36	6.31	261.3	197.2	288.8	224.7
Well 21	11.0	5.59	USS	0.80	0.30	0.30	9.4	9.4	8.6	8.6
Well 21	22.3	6.16	WSS	0.57	0.01	0.01	0.3	0.3	-0.3	-0.3
Well 21	25.3	5.75	USS	-0.27	0.57	0.57	17.8	17.8	18.1	18.1

BGS = Below Ground Surface
WSS = Weathered Sandstone
USS = Unweathered Sandstone
F = Upper Freeport No. 7 Coal
K = Middle Kittanning No. 6 Coal
* units = ton / 1000 tons as CaCO3

neutralization potential of 0.43 g/kg ($CaCO_3$). That well 19 (thus the highwalls) has good water while spoil groundwater is bad indicates oxidizing conditions in the spoil.

Using the MODFLOW computer program (McDonald *et al* ., 1988), the recharge for the spoil surrounding Pond J, H, and L that is required to match the seasonal flowrate recorded at Stream J (1.6 L/s), is 0.60 cm per year, which is relatively low. This may result, in part, from the steep slopes of the spoil piles and the lack of good vegetation. The highwall was not considered a source of water; it was modeled as a no-flow boundary. The haul road between the lake and the ponds was also considered a no-flow boundary because it is less than 3 meters above the elevation of the underclay and greatly compacted.

One of the most interesting aspects of this site is the water-quality in strip ponds to the west of Pond G (Table 5). All of the water to the west and northwest of Pond G is of good quality water. The strip mining done in these areas was done at the same time as the mining around Pond G, H, J, and L. As you go to the west of Spring U the water-quality improves. One possible reason is that there is a lithologie change from west to east. We have been unable to determine if limestone is present in the overburden to the west of Howard Williams Lake. Well logs from wells 22 and 28 show no evidence of any alkaline material. Another possible reason is that Pond A, B, and F are much deeper than Pond H, J, and L. On average, Pond A, B, F, and G contain 4 to 9 meters (15 to 30 feet) of water above the underclay. Pond H, J, and L are much shallower with depths of only 4 to 10 feet. The deep ponds are able to submerge the pyritic roof shale of the exposed highwall, while the water level in the shallow ponds fluctuate within the roof-shale zone, resulting in greater acid production in the shallow ponds.

The topography of the reclaimed spoil on the north side of the lake is substantially different than the undisturbed spoil to the south and west. The most noticeable difference is the lack of any significant inflows into the lake. The reclamation eliminated all of the strip ponds, thus it also eliminated the possibility that beavers would dam up a flow, creating a situation similar to Pond G, where increased head increased the amount of water flowing through potentially toxic spoil. The north side only contributes about 25 percent of the flow that enters Howard Williams Lake, but contributes a more significant amount of chemical load due to much higher concentrations of sulfate, iron, manganese, aluminum, and various other metals.

The effects that the coal-cleaning waste pit has on lake-water quality has not been clearly defined. Wells 1, 2, 2A, 2C, and 3 are all screened within the pit and are shown on the map in Figure 1. The water-quality data for these wells are

Table 5. Typical Water Quality of Ponds West of Howard Williams Lake

Location	pH	Total Acidity (mg/L as CaCO3)	Sulfate (mg/L)	Total Iron (mg/L)	Manganese (mg/L)	Aluminum (mg/L)
Pond A	6.51	5.0	240	0.47	2.12	0.38
Pond B	7.10	-34	223	0.27	0.55	0.30
Pond F	6.83	-17.5	198	0.50	0.47	0.49

Table 6. Typical Water Quality of Wells Located in Gob Pit.

Location	pH (field)	Total Acidity (mg/L as CaCO3)	Sulfate (mg/L)	Total Iron (mg/L)	Manganese (mg/L)	Aluminum (mg/L)	Total Hardness (mg/L)
Well 1	3.55	437	1317	244	16	4.1	1144
Well 2	4.91	345	1310	215	11	0.6	1025
Well 2A	4.52	856	2800	1160	22	21.6	1913
Well 2C	5.13	334	3382	299	12	4.6	1949
Well 3	3.56	197	791	112	4	7.0	493

shown in Table 6. Though the field pH is high in wells 2 and 2C, upon oxidation the pH drops to 3.0 resulting in the hot peroxide acidities shown in the table. Acid-base accounting data for well 3, shown in Table 4, show very high pyritic sulfur contents and $CaCO_3$ deficiencies. From the water-quality data, the mine waste appears to be very heterogeneous, with some areas producing more acidity than others.

Wells 5, 6, 10, and 12 are located in reclaimed spoil north of Howard Williams Lake. Water-quality data, shown in Table 7, show acidic conditions in wells 5, 6A, 10, and 12, with varying degrees of acidity. This is another example of heterogeneous conditions within a spoil pile. Acid-base accounting information for wells 5, 5B, 6A, and 10, also show very heterogeneous conditions, with some samples being marginally toxic, while other samples

Table 7. Well Water Quality Data for October 2, 1995.

Location	pH	Total Acidity (mg/L as CaCO3)	Sulfate (mg/L)	Total Iron (mg/L)	Manganese (mg/L)	Aluminum (mg/L)	Hardness (mg/L)
Well A8	3.21	1709	3800	720	83	90	2316
Well 2	5.20	-12	1129	214	9.3	0.6	872
Well 2C	5.85	-55	3671	310	10.8	0.1	1716
Well 3	3.15	176	727	120	4.5	29	120
Well 4	3.19	124	851	97	4.2	0.5	589
Well 5	3.09	523	2245	267	51	20	1768
Well 5B	4.51	237	2335	196	30	0.1	2126
Well 6A	5.10	521	1800	303	32	0.2	1410
Well 10	5.43	187	1893	262	52	0.2	1916
Well 12	3.68	227	3071	296	107	22	2516
Well 13	3.18	1783	4987	276	78	280	2947
Well 14	3.88	178	665	89	20	2	599
Well 17	3.05	970	3323	280	68	84	2253
Well 19	6.68	-127	482	0.81	2.5	0.1	453
Well 21A	6.35	-91	356	27	1.02	0.2	473
Well 22	5.30	-14	43	0.54	0.53	0.2	326
Well 28	7.02	-197	119	0.91	0.27	0.1	316
Well 35	5.92	-105	897	40	44	0.5	1168

are extremely toxic.

To this point, direct groundwater contributions to the lake have not been discussed. According to mine maps, the elevation of the underclay is above the lake surface. This make sense due to the number of seeps that exit the spoil several feet above the lake surface. Several water budget calculations have been made and are shown in Table 7. On average, surface inlfows contribute a high percentage of the lake outflow, the differences being subsurface flows and evaporation from the lake surface.

The anoxic limestone drain (ALD), located between ALD Spring and ALD Stream has been effective in treating groundwater that would eventually enter the lake. The water-quality characteristics of the water going into and exiting the anoxic limestone drain are shown in Table 9. The drain was built in 1990 and has a flowrate of about 2 gpm. There is no stabilization pond collecting the water that drains from the ALD, the water drains directly into Howard Williams Lake. It is interesting to note the amount of aluminum being taken out by the ALD, therefore at some point in the future it may show signs of clogging up.

Conclusions

The degradation of water-quality in Howard Williams Lake is caused by movement of groundwater through toxic spoil and possibly also by drainage of underground mine voids which are not fully flooded. The source water is recharge from precipitation percolating through spoil, and head differences created by ponds in old pits surrounding the lake. The water-quality in mines which are completely inundated is good, as a result of anoxic conditions. If the water table in the inundated mines is lowered, introduction of air could cause an increase in acid mine drainage

Table 8. Water Budget for Howard Williams Lake

Date:	9/22/94	10/8/94	10/15/94	10/27/94	8/28/95	10/2/95
Inflows (gpm)						
Stream J	27	30	36	28	14	21
Stream G	78	66	73	67	94	59
Spring G	25	19	20	20	19	18
Spring N	0	0	0	0	1	1
Stream W	0	0	0	0	53	34
Spring U	90	69	72	71	84	83
Iron stone seep (east)	0	0	0	0	0	0
Spring I	4	4	4	4	2	1
Spring W	0	0	0	0	7	5
Standpipe	2	0	1	2	2	2
ALD Stream	0	0	0	0	3	2
ALD Spring	3	4	4	4	7	6
Total Inflows	229	194	210	196	284	234
Outflow (gpm)						
Drop-Inlet	295	143	163	143	295	234

Table 9. Performance of Anoxic Limestone Drain.

Anoxic Limestone Drain					
date sampled:	8-Oct-94	5-Nov-94	22-May-95	28-Aug-95	2-Oct-95
	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)
In					
pH			3	3	3
Total Acidity			905	1414	1709
Aluminum			87	87	90
Out					
pH	5	6	6	6	6
Total Acidity	746	791	882	618	0
Aluminum	2	2	2	1	1

production. Since the No. 6 coal seam averages several percent pyritic sulfur, the coal remaining as pillars and between auger openings may provide a significant source for acid to be produced.

The effect that the coal-cleaning waste pit has on lake Water table elevations show a slight gradient toward the lake, but whether or not this water is directly affecting the water-quality of Howard Williams Lake is not known. Tracer tests and geophysical investigations may be able to define what role the coal-cleaning waste has on lake-water-quality. Likewise, the actual contribution to lake acidity from spoil piles surrounding the lake in not yet well defined. Water-quality data and acid-base accounting data show the heterogeneous conditions throughout the spoil area.

Data clearly show regional differences in water-quality surrounding the lake. Local sportsmen routinely fish in ponds west of the lake; unfortunately these high quality ponds contribute very little flow to the lake. Complete surveys of pond depths, correlated with adjacent coal-seam elevations should confirm the extent to which mined-out subsurface areas are flooded, and to what depths. Underclay elevation data suggest very localized variations in topography which could provide preferential flow paths in partially flooded underground mine rooms. Ponds as much as 25-feet deep may be creating anoxic conditions in both the mined-out subsurface as well as several feet upward into roof shales and pyritic sandstones. Other studies in Appalachian mining regions have shown that coal-seam roof rock may be more pyrite-rich than super-jacent or subjacent strata, and thus a greater source of acidity.

Our investigations ultimately will provide data upon which a sound and effective engineering design to re-route waters affecting the lake can be based. This design may well include some classical treatment options; but control of flows within the entire watershed appears also to have a high probability of effectiveness. A future use of this lake is proposed for recreation purposes, so that consideration of all related waters in the region is highly desirable.

Acknowledgements

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GROUND- AND SURFACE-WATER INTERACTIONS INVOLVING AN ABANDONED UNDERGROUND COAL MINE IN PIKE COUNTY, INDIANA[1]

by

Denver Harper[2], Greg A. Olyphant[3], and Donald R. Sjogren[4]

Abstract. Several highwall pits of an abandoned surface mine in the Springfield Coal Member (Pennsylvanian) are currently occupied by ponds with a total area of approximately 2.3 x 10^4 m^2. These ponds are adjacent to an abandoned underground mine (Patoka Valley Coal and Coke Company No. 1 Mine) in the same coalbed. The mine underlies about 0.3 km^2 and contains approximately 4 x 10^5 m^3 of flooded voids. Monitoring of water levels in wells that are screened in the mine and of the levels of adjacent ponds reveal that average hourly levels vary in unison across a range of less than one meter. The mean potentiometric level of the mine-aquifer, the neighboring ponds, and an artesian spring that issues through the outcrop of the coalbed, are at elevations of about 163 m above sea level. Long-term monitoring and a field experiment that involved pumping of a pond indicated that the mine was connected to two of the ponds and served to recharge, rather than discharge, the ponds. The monitoring and field experiment also allowed determination of the mine aquifer's barometric efficiency (0.3) and its storativity (2 x 10^{-3}). A water-balance calculation indicates that the average recharge rate of the mine is about 0.1 mm/day.

Introduction

The presence of flooded underground workings has profound effects on the hydrology of coal-mining districts because they represent large reservoirs of free-flowing, chemically altered water that may percolate vertically and exfiltrate laterally. In areas that have been subjected to both surface and underground mining, the potential for interactions between surface waters and groundwater is greater than in areas undisturbed by mining: (1) surface-mine highwall pits may abut workings of abandoned underground mines, (2) overburden above underground mines may be fractured, thereby increasing vertical percolation and recharge of voids, and (3) springs which are fed by flooded underground workings may discharge through the coalbed's crop or through mine entrances. Estimates are needed of the hydraulic properties of *mine aquifers*, which we can conceptualize as being comprised of fully flooded underground workings and associated strata in close hydrologic connection. These associated strata would include interior mine pillars, porous and (or) fractured overburden, and surrounding unmined coalbed to an indeterminate distance; in the case of most coalbeds of the Illinois Basin, the presence of relatively impermeable underclays may preclude downward percolation.

As part of background investigations at the Midwestern Reclamation Site in southwestern Indiana, we undertook a program of monitoring and experimentation, in an effort to characterize surface and groundwater interactions. Such characterization is needed in order to identify possible pathways of contaminant movement and other unanticipated hydrologic effects associated with reclamation.

Study Site And Monitoring Installations

The study area is located in Section 22, T. 2S., R. 7W., Pike County, Indiana (fig. 1). It is an upland area situated near the drainage divide between the watersheds of the Patoka River and

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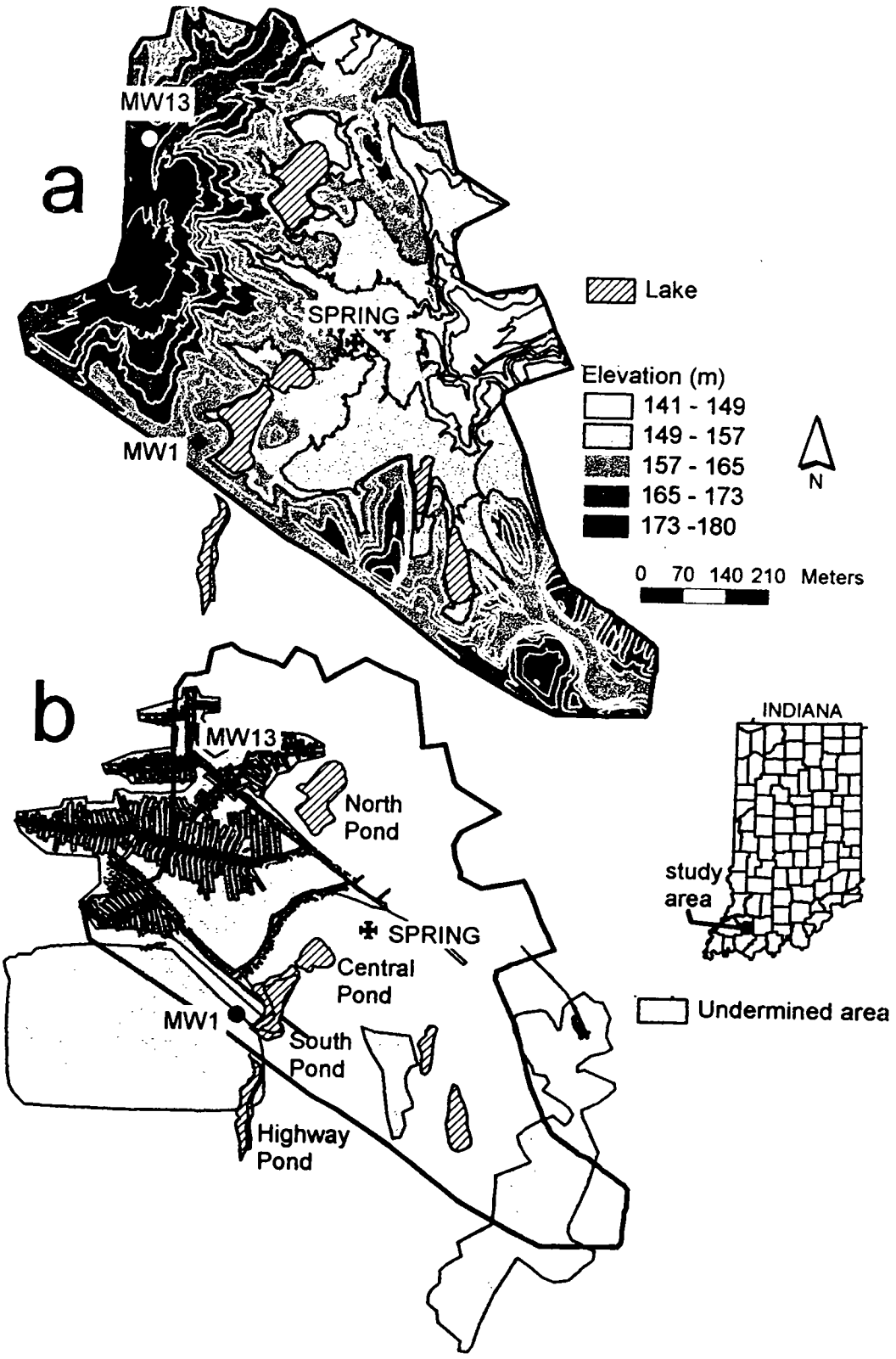

Figure 1. Maps showing (a) surface topography and (b) extent of underground mines and locations of ponds. Locations of monitoring wells and a spring are also shown.

the South Fork of the Patoka River. Much of the surrounding area has been disturbed by surface mining. Three small underground mines underlie parts of the study area at depths of about 30 m or less. As many as eight slopes and shafts formerly existed. In places, the surface mines encountered these abandoned underground workings. Mining was conducted in the Springfield Coal Member (Pennsylvanian).

Figure 1a shows the topography of the reclamation site and Figure 1b shows the portion of the reclamation site and adjacent areas that are underlain by abandoned underground workings. The abandoned Patoka Valley Coal and Coke Co. No. 1 Mine underlies the western part of the reclamation site. There is a map that shows details of the workings in the northern part of this mine (see fig. 1b), but no such detailed map exists for the southern part; the two parts may be separated by a long barrier pillar.

Four surface-mine highwall pits, now filled with water and referred to as the *North, Central, South,* and *Highway Ponds,* are situated along the eastern margin of the Patoka Valley No. 1 Mine and are in close proximity to the mine's workings (fig. 1b). According to available maps, the South Pond intersects a slope entry, while the Highway Pond intersects the southeastern corner of the mine. Continuous electronic monitoring of water levels was conducted at the North Pond (August 15 to October 12, 1995), the Central Pond (May 25 to October 4, 1995), and the South Pond (October 4 to December 11, 1995). The lake levels were also periodically measured at all four ponds using stage staffs. Reclamation activity, which involves draining the ponds, commenced on October 10, 1995.

A monitoring well (MW13, fig. 1) is located within a small interior pillar within the mine workings, and another monitoring well (MW1) is located within the barrier pillar between the two parts of the mine. The water levels within MW13 and MW1 have been continuously monitored since May 25 and September 13, 1995, respectively. (Note: malfunctioning dataloggers resulted in a loss of data from MW13 between September 13 and October 4 and from MW1 between September 27 and October 4, 1995.)

A perennial spring that issued from an outcrop of the coalbed was the major source of streamflow that discharged from the site during dry periods. This spring was located only about 10 m from a small shallow impoundment whose water level stood about 2 m above the spring. Both the spring and the impoundment were in the vicinity of a former slope entry into the underground mine. As part of a field experiment (described below) the discharge of the spring was continuously monitored using an electronic pressure transducer installed in a stilling well; a stage-discharge rating curve was developed to translate the depth measurements into flows.

Results

Water Levels

At any given time, water levels in the wells and ponds differed from each other by as much as 0.4 m. Because monitoring was conducted from late spring through autumn, water levels exhibited an overall decline, falling from an elevation as high as 163.4 m (above mean sea level) at the beginning of the study to less than 161 m at the end of the period considered in this report (fig. 2). Both the wells and the ponds exhibited rapid water-level rises associated with rainstorms that occurred at the site. During a dry period in August, when there were few water-level perturbations caused by rainstorms, the Central Pond, North Pond, and MW13 declined at similar, yet measurably different, rates (1.45, 1.57, and 1.39 cm per day, respectively).

Figure 3 shows water-level changes in MW13 during a five-day period in mid-August. A daily cycle is superimposed upon the long-term decline that is evident in figure 2; this daily cycle is characterized by two maxima and two minima each day. We have previously observed a similar cycle in another flooded underground mine of southwestern Indiana (Harper and Olyphant, 1992), where we concluded that the cyclicity was caused by atmospheric pressure changes.

Barometric Efficiency

The ratio of changes in water level to changes of atmospheric pressure, when both are expressed in the same units, is

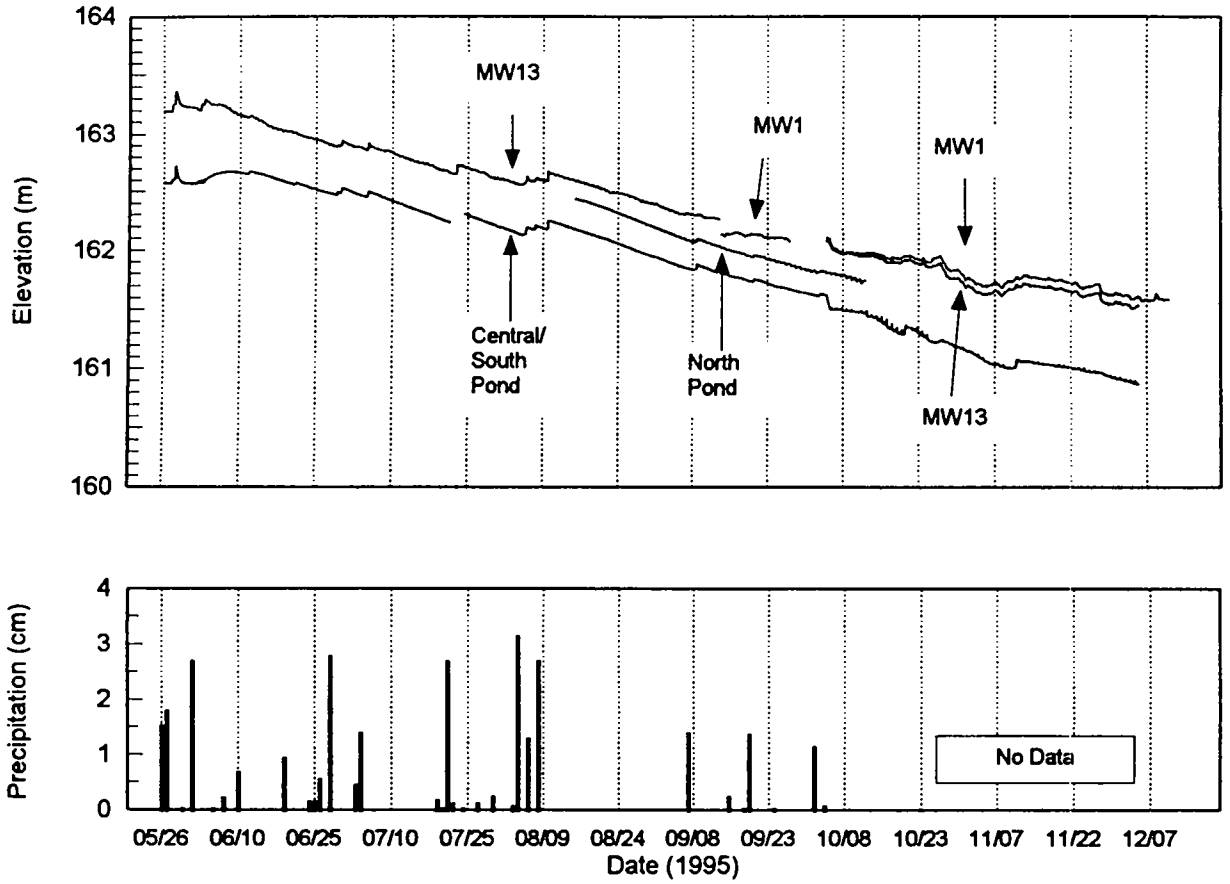

Figure 2. Graphs showing water levels and precipitation during the period of monitoring. Locations of monitoring sites are indicated in figure 1. Vertical grid lines indicate 00:00 AM.

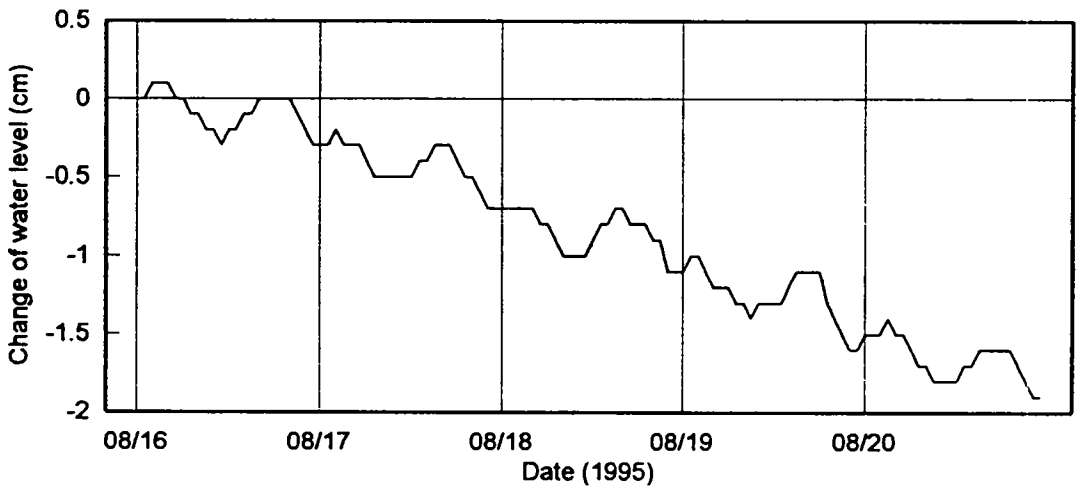

Figure 3. Graph showing daily cycles of water level observed in MW13 during a rainless period in August, 1995. The Y-axis shows the change in water level from its initial value. Vertical grid lines indicate 00:00 AM.

defined as the *barometric efficiency* of a confined aquifer and is related to the aquifer's storativity (Todd, 1959). In Figure 4, daily changes of water level in MW13 are plotted versus daily changes of atmospheric pressure. The overall trend is complicated by variations in water level associated with rainstorms. We employed multiple regression to calculate the partial effect of pressure variations on water-level changes. The statistical results, which are presented in Table 1, indicate that the barometric efficiency of the mine aquifer is 0.31.

Together with information on the thickness and porosity of the mine aquifer, this value of barometric efficiency can be used to estimate the mine aquifer's storativity, according to the following equation:

$$S=\frac{n\gamma b}{EB} \qquad (1)$$

where S is the storage coefficient, n is porosity, γ is the specific weight of water (9800 N/m^3), b is aquifer thickness, E is the bulk modulus of compression of water (approximately 2.07 x 10^9 N/m^2), and B is barometric efficiency. However, considerable uncertainties exist in the estimates of porosity and aquifer thickness in this setting. If we assume that the aquifer consists solely of the flooded mine workings (voids only, and excluding internal pillars), which average 1.4 m in thickness, then the porosity is 1.00 and the value of the storage coefficient is 2 x 10^{-5}. However, if we assume that the aquifer included the sandstone overburden up to the potentiometric level, as well as the mine workings, then the weighted value of porosity is 0.4, the aquifer thickness is 5.5 m, and the value of the storage coefficient is 3 x 10^{-5}. In either case -- given the value of barometric efficiency that we have determined -- the storativity of the mine aquifer lies at the low end of the range (5 x 10^{-3} to 5 x 10^{-5}) given by Freeze and Cherry (1979) for natural confined aquifers.

Field Experiment

On October 4 and 5, 1995, a field experiment was conducted in an effort to quantitatively evaluate interconnection between various ponds, the mine aquifer, and the spring. In order to remove the largest volume of water possible in the shortest period of time, it was decided to pump a lake rather than attempt to pump a well. Because the South Pond is believed to intersect an old slope entrance, it was chosen to be pumped. A volume of 1.5 x 10^6 l was withdrawn in a period of 19.5 hours. During this test, water levels in MW13, MW1, North Pond, South Pond, and the spring were electronically monitored at 15-minute intervals. Stage staff measurements were periodically made at the Central Pond, South Pond, and Highway Pond. Figure 5 shows water-level trends during the experiment, and Table 2 summarizes total water-level and volumetric changes associated with the drawdown.

As shown in figure 5, water levels in the mine (MW13) and the South Pond fell continuously during the period of pumping by about 3.7 and 12.2 cm, respectively, and did not recover in the period immediately following the cessation of pumping. In contrast, the stage of the spring's discharge increased slightly in the early phase of pumping, then declined abruptly about six hours into the experiment, and finally recovered to its preexisting value within a few hours following the end of the experiment. The decline of the spring's stage represents a decrease in its flow rate from 0.5 l/s to 0.1 l/s and a total outflow deficit of 4 m^3 over the period of the experiment.

The difference between the volume of water removed by pumping and the sum of the volumes of water associated with the drawdowns of the Central and Highway Ponds can be used as an estimate of the volume of water lost from the mine aquifer. This estimate, together with data regarding the area of the mine and the observed drawdown in the mine wells, can be used to calculate the storativity of the mine aquifer:

$$S=\frac{\Delta V}{A\Delta h} \qquad (2)$$

where ΔV is the estimate of the volume of water lost from the mine aquifer and A is the area of the mine. Using the data in Table 2, we calculate that the

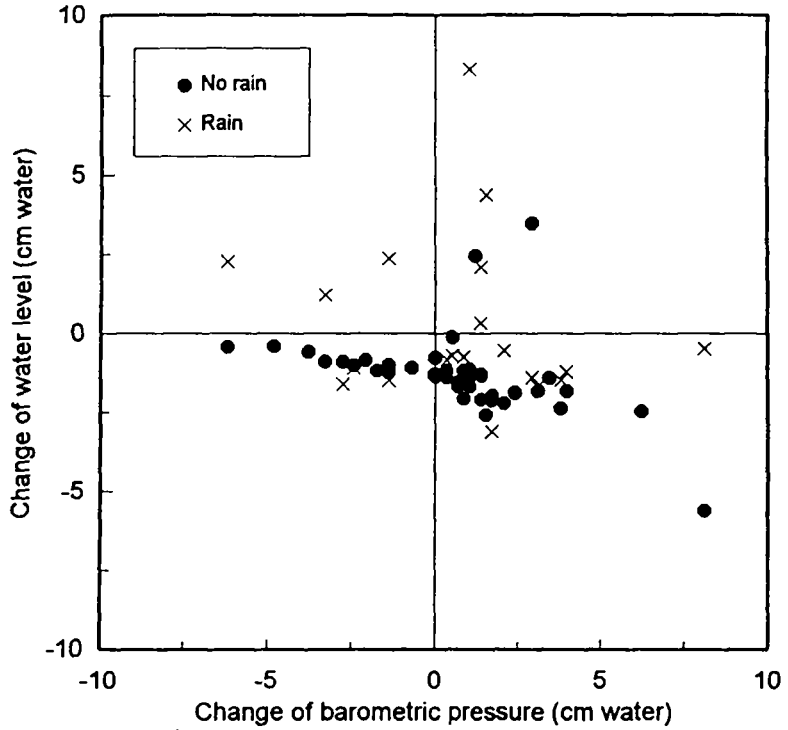

Figure 4. Plot of daily changes in water level of MW13 versus daily changes of atmospheric pressure.

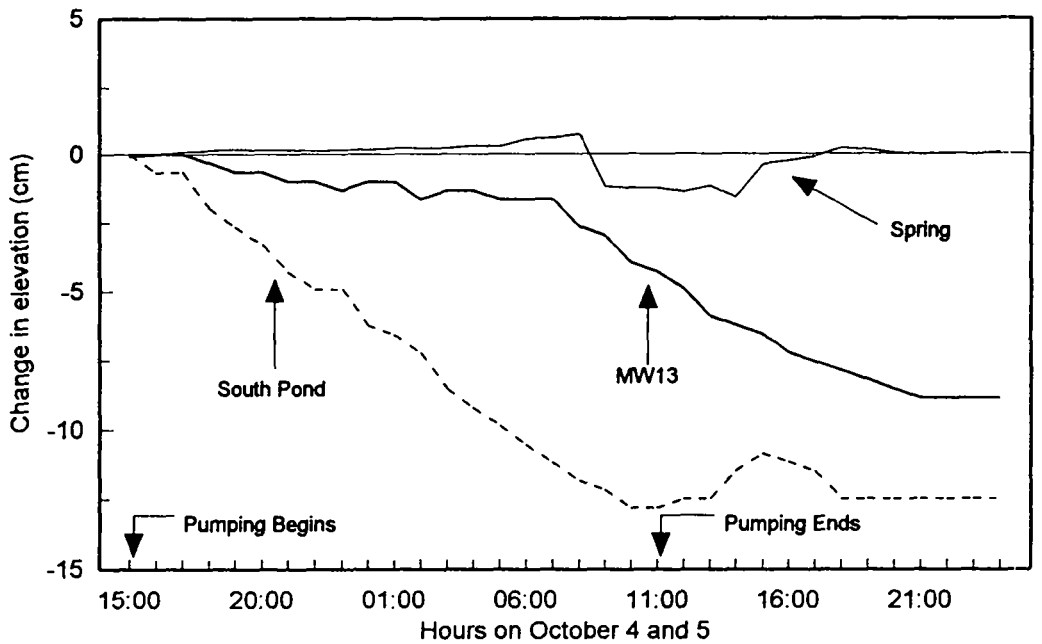

Figure 5. Response of water levels in MW13 and the South Pond during the pump test conducted on October 4 and 5, 1995. The spring's change of stage is also shown.

TABLE 1. Multiple regression analysis of the relationship between changes of the water level in MW13 (Δh), changes of atmospheric pressure (ΔB), and precipitation (P)[1,2,3].

Model: $\Delta h = b_0 + b_1\Delta B_t + b_2 P + (\rho e_{t-1} + v_t)$				
n[4]	b_0	b_1	b_2	ρ
67	-1.356* (0.144)	-0.306* (0.051)	2.050* (0.252)	-0.29

[1]Variables are expressed in centimeters of water.
[2]Regression parameters (b_0 to $_2$ b) and autocorrelation coefficient (ρ) were estimated using the procedure outlined in Kmenta (1971, p. 288).
[3]Values in parentheses are standard errors of regression parameters; asterisks (*) indicate that regression parameters are statistically different from zero at 95-percent confidence level.
[4]n = number of observations;

TABLE 2. Summary of total water-level and volumetric changes associated with pumping of the South Pond. The total volume that was pumped was 1,480 m^3.

Site	Water-level Change (cm)	Area (m^2)	Volume Change (m^3)
SURFACE WATERS			
South Pond	-12.2	6,575	802
Highway Pond	-12	5,460	655
Central Pond	0	not applicable	
North Pond	+0.1		
Spring	---	not applicable	-4[a]
UNDERGROUND MINE			
MW1	-3.7	310,400[b]	27[c]
MW13	-3.7		

[a] Difference between the integrated discharge that was observed (fig. 5) and the discharge that presumably would have been observed if the pump test had not been conducted.
[b] Area includes internal pillars of mine, as well as void spaces.
[c] The volume of water removed from the mine aquifer was determined by subtracting the sum of the volumetric changes given above for the ponds and the spring (1,453 m^3) from the total volume that was pumped (1,480 m^3).

storativity of the mine aquifer is approximately 2 x 10^{-3}. In contrast to our estimate based on barometric efficiency (see discussion above), this estimate lies at the upper end of the range cited by Freeze and Cherry (1979) for natural confined aquifers.

Discussion And Conclusions

Even though available maps indicate that the flooded highwall pits intersected the workings of the abandoned underground mine, the following observations indicate that resistances to flow -- both within the mine workings and between the workings and the ponds -- exist: (1) water levels at all of the sites that were surveyed are at somewhat different elevations at any given time, and (2) although the water levels at different sites show very similar long-term trends, their rates of change are slightly different. On the other hand, both the mine wells and the ponds showed rapid responses to all of the rainstorms that occurred. The higher elevation of the water levels in the mine wells during the period of study (a drying season) indicates that the mine serves as a source of recharge to the ponds and that the mine must be receiving rapid recharge from some other source.

The mine aquifer can be categorized as a *confined aquifer* because its potentiometric level sets high above the roof of the mine. Also, there is no evidence of any conduit connection between the mine and the adjacent ponds. This characterization of the mine aquifer is consistent with the observation of persistent daily cycles of water-level change in the mine and the statistically significant correlation that exists between daily water-level changes and daily atmospheric pressure changes (Table 1). The calculated barometric efficiency of the mine aquifer indicates that this aquifer's storativity is very small (compared with natural aquifers); the field experiment, however, indicates a contrary result. We have conducted similar monitoring (to estimate barometric efficiencies) and field experiments (to determine storativities) at two other sites in southwestern Indiana: at one site, the barometric efficiency was about 0.7 and the storativity was 3 x 10^{-4}, while at the other site, the barometric efficiency was about 0.2 and the storativity was 3 x 10^{-3} (Harper and Olyphant, 1992). These findings were consistent, in that the site with the lower barometric efficiency has the higher storativity. The values obtained in this study -- a low value of barometric efficiency (0.3) and a high value of storativity (2 x 10^{-3}) -- are consistent with the previous results at the other sites and indicate that the mine aquifer at the Midwestern Site is relatively poorly confined. Our results also indicate that values of the storativity of mine aquifers calculated from barometric efficiencies may not be reliable, because of the inherent difficulty of defining the porosity and thickness of such highly heterogeneous aquifers.

Our field experiment indicated the mine aquifer (as monitored in both MW1 and MW13) clearly responded to the drawdown of the South Pond (exhibiting a decline that was seven times greater than the normal daily decline), although the reponse was somewhat delayed and only 30 percent of the decline induced in the South and Highway Ponds. According to our calculation of storativity, only about 27 m^3 of water were lost from the mine aquifer.

The flow discontinuity that occurred near the end of the pump test (and that was coincident with a steepening of the rate of decline of water levels in the mine wells) may be additional evidence that the spring's discharge is derived in part from the mine aquifer. In our calculation of the volume of water lost from the mine, we assumed that the 4 m^3 decrease of spring's discharge associated with the observed reduction of flow was a credit to the mine aquifer's water budget. Even if we disregard the spring's discharge, however, the calculated storativity of the mine aquifer remains on the same order of magnitude.

The data derived from our monitoring program and field experiment have implications for the water budget of the site. For example, the average decline of the water level in MW13 of 1.4 cm/day indicates a net outflow of about 9 m^3/day. However, if we assume that the entire discharge of the spring (ca. 34 m^3/day) is derived from the mine and that other losses probably occur through the mine's connection to the ponds (by

evaporation or exfiltration into adjacent surface-mine spoil deposits) or through exfiltration into adjacent unmined coal, we are in a position to estimate the recharge rate to the mine. If discharge from the mine were solely through the spring, then the recharge rate of the mine would be about 0.1 mm/day; even by doubling the assumed discharge to account for other, unmeasured outflows, the mine's recharge rate must be substantially less than 1 mm/day. This low recharge rate is consistent with characterization of the mine aquifer as a confined aquifer, and indicates that the volumes of water associated with the response of the mine's water level to individual rainstorms is small (< 50 m^3 for rainstorms observed in this study).

Hydrologic conditions in areas where shallow underground workings are in close proximity to surface mines are inherently complicated. Continuous monitoring of water levels can provide information about the responsiveness of various system-components to external stresses and allow inferences about possible pathways of flow between those components. However, such data do not provide direct evidence of connections. Indeed, the variations in water levels that are observed may simply be similar responses to some unmeasured (exogenous) variable. Field experiments -- such as the one described in this report -- where stresses are artificially induced in mine-aquifer systems, provide a more direct basis for qualitative inferences about connections and flow directions, as well as for quantitative determinations of hydraulic properties (*e.g.*, storativity) that can be used in calculations of dynamic water-budgets.

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MODELING THE EFFECTS OF LONGWALL MINING ON THE GROUND WATER SYSTEM[1]

by

R. J. Matetic[2], J. Liu[3], and D. Elsworth[4]

Abstract: The objective of this U.S. Bureau of Mines hydrologic/ subsidence investigation was to evaluate the effects of longwall mining on the local ground water regime through field monitoring and numerical modeling. Field data were obtained from multiple-position borehole extensometers (MPBX's) that were used to measure subsurface displacements. Survey monuments were installed to measure mining-induced surface deformations. Numerous drawdown and recovery tests were performed to characterize hydrologic properties of the overburden strata. Coreholes were drilled above the study area to determine lithologic and strength characteristics of the overburden strata using the rock samples collected. Electronic recorders were installed on all monitoring wells to continuously monitor ground water levels in coordination with mining of the longwall panels. A combined finite element model of the deformation of overlying strata, and it's influence on ground water flow was used to define the change in local and regional water budgets. The predicted effects of the postmining ground water system determined by the model correlated well with field data collected from the field site. Without an infiltration rate added to the model, a static decrease of 3.0 m (10 ft) in water level would occur due to mining of both longwall panels and if an infiltration rate was inputted in the model, no predicted long-term effects would occur to the ground water system.

Introduction

Longwall mining is a method used to extract large blocks of coal. During extraction of the block, the immediate overburden is allowed to collapse, filling the void created by the excavation. Mining-induced strains and displacements are transferred throughout the overburden rock mass due to this collapse and the resultant stress redistribution creates changes to the ground surface and any water-bearing zones located above the mining area. Previous studies have been conducted to delineate the effects of longwall mining on the local ground water system (Booth 1992; Elsworth 1994; Johnson 1992; Leavitt 1992; Matetic 1990; 1991, 1992; Tieman 1992; and Trevits 1991)). Few of these studies, however, have used actual field data, in conjunction with numerical modeling, to determine and

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predict mining effects on the ground water regime. The U.S. Bureau of Mines (USBM) is studying the overburden rock mass and its response to high-extraction mining operations through a comprehensive program of field studies. The first part of the work involves the collection of mining, subsidence, overburden response, and hydrological data before, during, and subsequent to mining activity at numerous field sites. The second part of the program is examining methods of predicting the impact to local ground water supplies after mining activity occurs. One of these methods, is through the application of numerical modeling.

Model Approach

An intensive surface, subsurface, and ground water monitoring program was conducted at a mine site in southeastern Ohio. Data collected from this site served as input information for a Finite Element (FE) model. The two-dimensional FE model incorporates the deformation of overlying strata and its influence on ground water flow through applying a simple relationship between mining-induced strains and changes in hydraulic conductivity. The strain field that develops around a longwall panel as a result of mining is caused by material failure and self-weight. From this predicted strain field and from knowledge of the premining hydraulic properties of the overlying strata, the change in hydraulic conductivity that results from the strain field may be determined. With the modified conductivity field determined, the postmining hydrologic system may subsequently be defined through application of a ground water flow model. Again, this ground water model utilizes the finite element method to determine the postmining hydrologic system where the position of the piezometric surface indicates changes in well or aquifer yields. This methodology is used to evaluate the influence of mining on the local ground water regime in this study.

Theoretical Analysis of Modeling Approach

The following assumptions are made when operating the model: (1) the rock matrix is functionally impermeable in comparison with fractures; (2) fluid flow in fractures is defined on the basis of the parallel plate model; (3) changes in fracture conductivity result from changes in normal strains only; (4) strains are partitioned between fractures and matrix as defined by a modulus reduction factor, R_m; and (5) fracture spacing, s, does not change after mining activity (Liu 1994 and Ouyang 1993).

Correlation of Induced Strain and Hydraulic Conductivity

The equivalent porous medium conductivity, K_o, of a rock mass containing a parallel set of fractures can be defined as:

$$K_o = \frac{g}{12\nu_k}\frac{b^3}{s} \qquad (1)$$

where g is gravitational acceleration, ν_k is kinematic viscosity, b is the fracture aperture and s is spacing. R_m, the modulus reduction factor may be defined as:

$$R_m = \frac{E}{E_r} \qquad (2)$$

where E is the deformation modulus of the rock mass and E_r is the deformation modulus of a rock specimen. The modulus reduction factor, R_m, enables the closure across a fracture, Δu_j, to be determined from the difference between the strains in the rock mass and rock speciman as shown below:

$$\Delta u_i = [D + s\,(1 - R_m)]\,\Delta\epsilon \qquad (3)$$

where $\Delta\epsilon$ is the strain in the direction perpendicular to the fracture plane. $\Delta\epsilon$ is positive in extension and negative in compression.

Using the applied strain, $\Delta\epsilon$, from above, the revised conductivity of equation (1) may be defined as:

$$K = R_c K_o \tag{4}$$

where

$$R_c = \left[1 + \frac{b + s(1 - R_m) b^3}{b}\right] \Delta\epsilon \tag{5}$$

Directional conductivities, evaluated from initial conductivities, K_{ox} and K_{oy} (conductivities in the x- and y-directions, respectively) may then be determined from equation (4). With two sets of orthogonal fractures oriented in the x and y directions, the revised directional conductivities may be defined for a two-dimensional system as:

$$= K_{xo} \left[1 + \frac{b + s(1 - R_m)}{b} \Delta\epsilon \right. \tag{6}$$

and

$$= K_{yo} \left[1 + \frac{b + s(1 - R_m)}{b} \Delta\epsilon \right. \tag{7}$$

where K_x and K_y are postmining conductivities in the x-direction and the y-direction, K_{xo} and K_{yo} are the premining conductivities in the x-direction and the y-direction, and $\Delta\epsilon_x$ and $\Delta\epsilon_y$ are the induced strains in the x and y directions, respectively. R_m is a modulus reduction ratio (ratio of mass modulus to intact modulus) that apportions the changes in strain between the fracture and matrix material. When $R_m = 1$, the mass modulus and intact material modulus are identical and the strain is uniformly distributed between fractures and matrix. This results in the smallest possible change in conductivity. When $R_m = 0$, the extensional strain is applied entirely to the fracture system and precipitates the largest possible change in conductivity. These values bound the possible ranges in the behavior of the system in a natural and mechanistically defensible manner. This representation of conductivities is extremely useful, since the modulus-reduction factor, R_m, may be readily evaluated from rock mass classification systems defining structural behavior as a function of readily observable factors of rock structure (Voight 1970). This avoids the difficulty of defining conductivity enhancement in terms of the component moduli of fractures and matrix, parameters that are unlikely to be available in practice. The mining-induced conductivity changes can then be evaluated through equations 6 and 7, provided the mining-induced strain field is determined.

Determination of the Strain Field

The subsidence field that develops around a longwall panel may be determined directly from the FE model. The finite element model applies gravitational load, removes material excavated from the panel and allows the overburden material to fail and deform according to the mining-induced strains. The resulting subsidence field may use the modulus reduction parameter, R_m, to calibrate against field data for a particular site. The insensitivity of the resulting subsidence profile to the material properties of deformation modulus and rock strength parameters, originates from the overriding influence of geometric controls on deformation (Stoner 1983 and Walker 1986). Following mining, the panel span is sufficiently large that closure between panel floor and roof is unavoidable.

Consequently, the resulting strain field, ϵ_x and ϵ_y is defined purely as a function of geometry, as:

$$\epsilon_x, \epsilon_y = f\left[\frac{w}{t}, \frac{w}{h}\right] \qquad (8)$$

where: w represents the width of the panel, t is the thickness of the coalbed and h symbolizes the thickness of the overburden.

The assumption necessary in this evaluation is that strains are uniformly distributed at the scale of a single element. These assumptions seem reasonable where strains are moderate, but may be questionable where significant strain localization occurs.

Determination of the Postmining Ground Water Regime

With the modified conductivity distribution determined from an evaluation of the strain field, and equations (16) and (17), the influence on the postmining ground water regime may be evaluated. The finite element model may determine the influence of a continuously distributed conductivity field (evaluated from the calculated strain distribution) on the ground water budget and water table where boundary conditions are applied to the local system, to represent ground water and surface recharge. Therefore, the change in elevation of the phreatic surface may be determined for the postmining regime. This enables the influence of mining on well yields, aquifer yields, and flow patterns to be identified.

Input Parameters Used for the Modeling Approach

The primary parameters used as input to the model and the measurements obtained from the field are: (1) the initial hydraulic conductivity distribution of the local lithology as determined through field measurements; (2) modulus of elasticity and Poisson Ratio for the rock mass determined from field measurements; (3) the measured subsidence profile; (4) measured vertical displacements; (5) continuous fluid level fluctuations monitored at the site; and 6) flow rates entering the mine after excavation of the longwall panels, as recorded by the operator.

Site Description and Ground Water Monitoring Program

Site Description. The study site is located in southeastern Ohio (Vinton County). The study area overlies a portion of two contiguous longwall panels (Panel Nos. 1 and 2) measuring approximately 300 m (900 ft) wide and 2,950 m (9,000 ft) long (figure 1). The panels were separated by a five entry, four pillar system approximately 120 m (350 ft) wide. The mined coalbed, had an average thickness of 140 cm (55 inches) within the study area. However, the extraction thickness varied between 173 and 183 cm (68 and 72 inches). Overburden thickness was small and ranged from 65 to 85 m (214 to 280 ft). Overall, the strata were fairly level with a regional dip of about 1 degree towards the southeast. There were no major geologic structures and the topography consisted mainly of rolling hills with a maximum relief of approximately 49 m (160 ft).

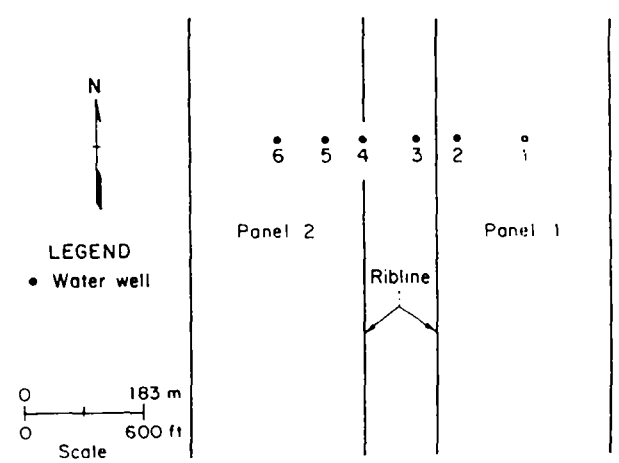

Figure 1. Sketch of study area.

Ground Water Monitoring Program. A total of seven 22-cm (8 5/8-in) diameter monitor wells were drilled for the study. Perforated Schedule 80 (15-cm diameter) (6-in) PVC casing was installed to the total depth in each well to ensure an open wellbore for the life of the study. The wells were strategically placed above both longwall panels as shown in figure 2. The wells were located along a line perpendicular to the trend of the longwall panels. This alignment permitted observations of effects during the mining of both longwall panels. Well Nos. 1 and 6 were located at the center of panel Nos. 1 and 2 respectively. Well Nos. 2 and 5 were located at quarter-panel width. Well No. 3 was located above the gate roads between the two panels and Well No. 4 was located above the edge of panel No. 2. Well No. 7, a control well, was located 427 m (1,400 ft) away from any mining activity.

Data was collected from all wells before, during, and after mining of both longwall panels. Various hydrologic parameters were determined and included specific capacity, transmissivity, hydraulic conductivity, and water level fluctuations. Initial data collection began three months prior to the undermining of Well No. 1 to establish baseline conditions. Drawdown and recovery pumping tests were performed on all wells before and after undermining to determine hydraulic conductivity parameters of the local, shallow geologic units.

Electronic recorders were also installed on all wells to continuously monitor water level fluctuations. The electronic data logger was programmed to record fluid positions every 4 hours.

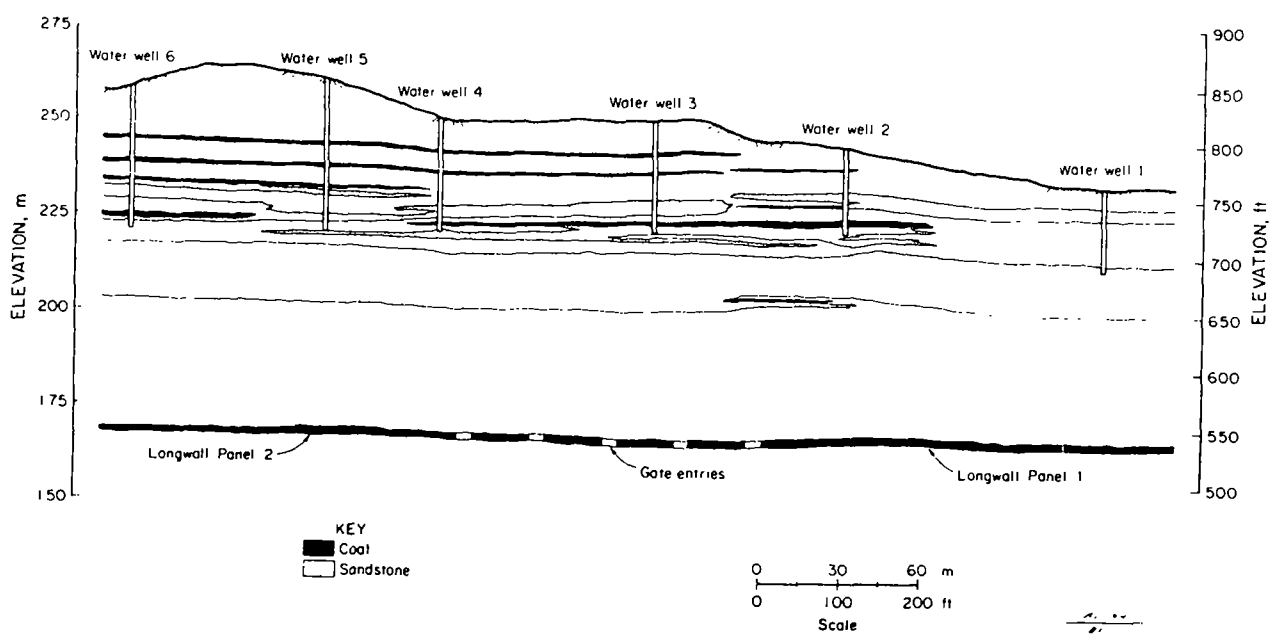

Figure 2. Generalized cross section of study area and Water Well locations

Geology

The geological setting of the study area is typical of that found in southeastern Ohio. The regional dip in this part of the state is to the southeast with the strata striking in a northeast-southwest direction. The average rate of dip is 6 m/km (30 ft/mi). Irregularities to this rate can be experienced by localized thinning or thickening of individual rock units (6). The rock is predominantly interbedded sandstones, shales, thin coal seams and claystones. The individual units are thin (less than 3 m (10 ft) thick) with one sandstone unit having an average thickness of 14 m (45 ft) and lying 43 m (140 ft) above the Clarion 4A Coalbed. To characterize overburden lithology prior to drilling the monitoring wells, six NX-sized coreholes were drilled in the study area to the Clarion 4A Coalbed. Generally, the overburden consisted of about 30% sandstone, 30% shale, 30% claystone, and 10% coal. A generalized cross-section of the study area with the monitor well locations is shown in figure 2.

Modulus of Elasticity and Poisson Ratio Values

The coreholes were drilled above the study area to characterize the local lithology of the overburden and to provide samples for determining geotechnical properties. For input to the model, tests for determining compressive strength, Modulus of Elasticity and Poisson Ratio were conducted on the rock cores collected. Results of tests performed on the core samples showed that the unconfined compressive strength of the major sandstone units are between 2.6×10^4 and 3.0×10^4 kg/m^2 (5,300 to 6,000 psi). Modulus of Elasticity and Poisson Ratio for these units are between 1.3×10^7 to 1.4×10^7 kg/m^2 (2.57 to 2.82×10^6 psi) and 0.30 to 0.32 respectively. A limestone unit, averaging 2.1 m (7 ft) in thickness, is situated about 1 m (3 ft) above the mined coalbed. Testing of core samples show that the unconfined compressive strength of this unit is 1.2×10^5 kg/m^2 (25,000 psi), with a Modulus of Elasticity of 5.1×10^7 kg/m^2 (10.5×10^6 psi), and a Poisson Ratio of 0.28.

Overburden Deformation Monitoring Program

To observe overburden displacement, six 219.1 mm (8-5/8 in) boreholes were drilled along a profile line extending across the two longwall panels. Boreholes 1 and 6 were located in the center of each panel, where the maximum amount of subsidence was expected to occur. Boreholes 2 and 5 were located 30 m (91 ft) from the ribline inside each panel. Borehole 4 was situated 3 m (10 ft) from the ribline (inside the panel) in the expected zone of maximum horizontal tension. Borehole 3 was located in a pillar in the gate entries between the panels to observe the lateral extent of overburden deformation.

Each borehole was outfitted with an eight-anchor multiple-position borehole extensometer (MPBX). Two of the eight anchors in borehole 3 were installed inside a coal pillar to monitor yielding of the pillar. The anchors are numbered 1 to 8, with anchor 1 being the closest to the surface and anchor 8 being the deepest. Figure 3 displays MPBX and anchor locations with respect to the longwall panels.

Subsidence Monitoring Program

To obtain the field subsidence profile, survey monuments were installed on the ground surface and were surveyed regularly to identify the dynamic characteristics of subsidence, the final subsidence profile, and to provide surface reference data for the MPBX units. The monuments were constructed of 1.3 m (4 ft) rebar and

were installed flush to the ground surface. The array of monuments consisted of a baseline (along the centerline) over each panel and a profile line trending perpendicular between the two baselines. The monuments were spaced 15 m (45 ft) apart along the baseline. The profile line was 375 m (1,136 ft) long with monuments spaced 7.5 m (23 ft) apart.

MODEL ANALYSIS

Finite-element Mesh Assemblage

The finite-element mesh construction was assembled utilizing figure 3. The effects of topography, geometry, and lithology were incorporated in the mesh. The mesh utilizes uniform spacing and was constructed of 2,066 nodes and 1,928 elements. The mesh assumes differing materials (overburden material and coal layer) for the determination of strain and displacement characteristics, and three materials (upper shale layer, sandstone layer and lower shale layer) for determination of the postmining flow characteristics within the system. The boundary conditions of the model for determining displacement and strain characteristics assume no horizontal movement on either side of the mesh and no vertical movement on the base. Boundary conditions associated with monitoring postmining ground water effects assume no flow on the bottom of the mesh and constant head conditions on the lateral sides of the mesh. The finite-element model is two-dimensional and determines the strain field with two displacement degrees of freedom applied to each node, and subsequently evaluates the revised flow system using a single degree of freedom. The analyses are coupled through the dependence of hydraulic conductivity on the induced strain field, as defined in equations 6 and 7.

Subsidence Profile

The subsidence profile (figure 4), generated by the model, was determined with the input parameters as shown in table 1.

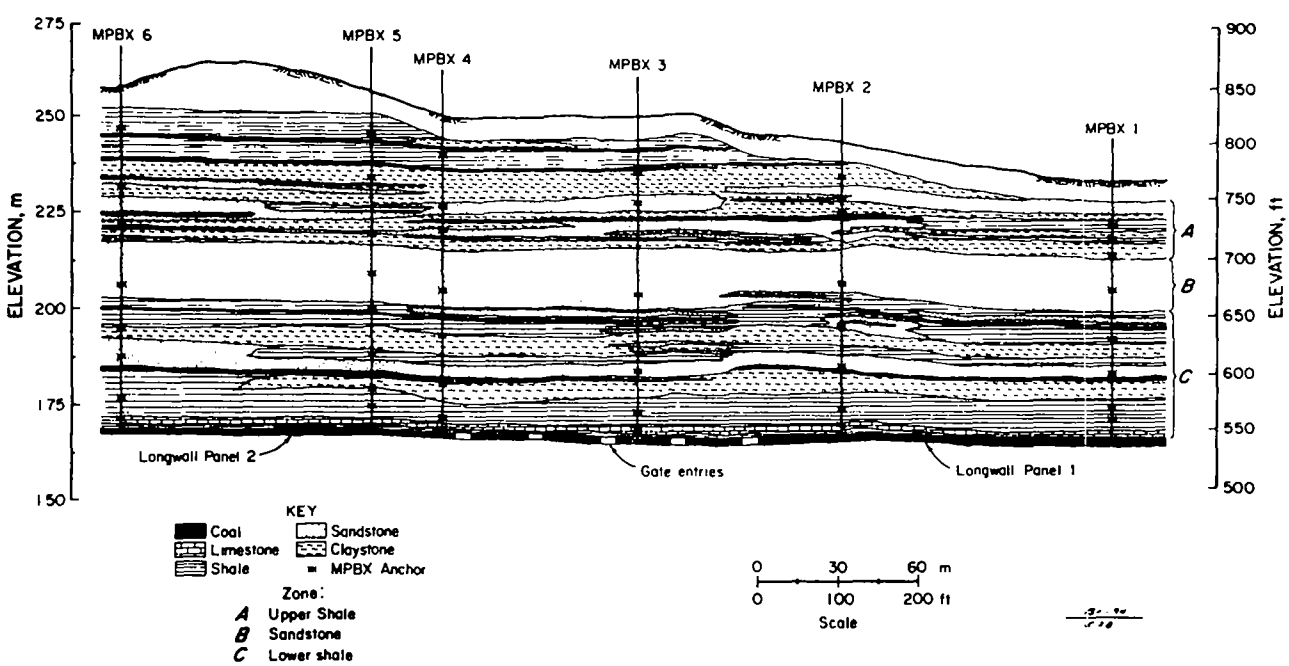

Figure 3. Generalized cross section of study area and MPBX locations

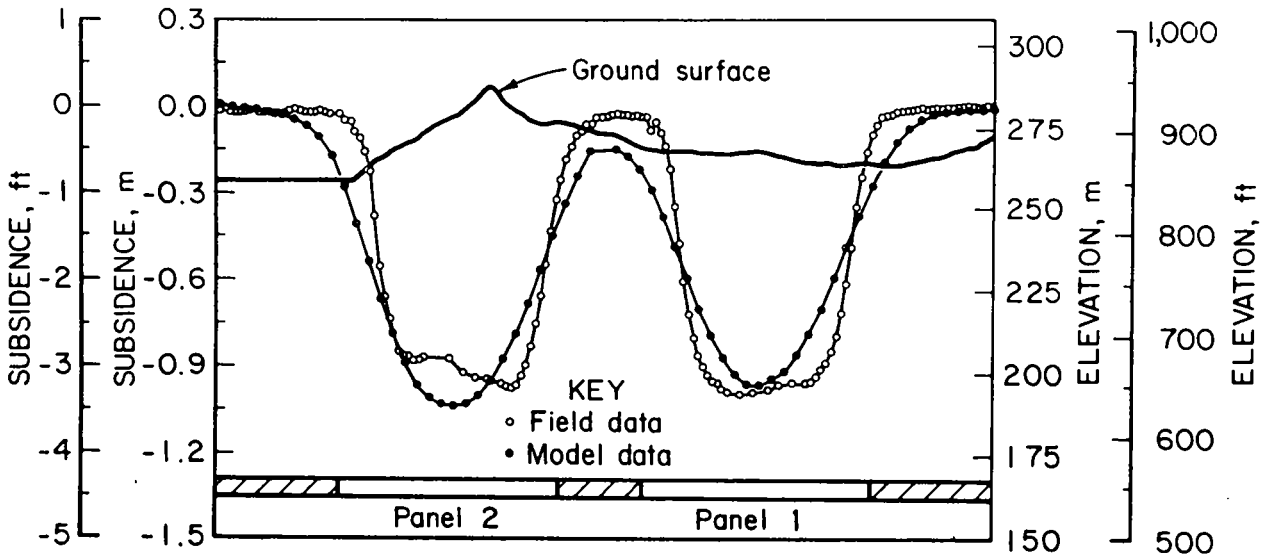

Figure 4. Comparison of field data and model subsidence profiles

Table 1.—Input parameters for generation of subsidence profile

Input Parameter	Overburden Material	Coal Layer
E, MPa (psi)	1,550 (2.2 x 10^5)	1.56 (2.2 x 104)
n	0.30	0.30
D, kg/m³ (lb/ft³) . .	2,400 (150)	390 (80)
E_{insitu}/E_{failed}	4.0	4.0
$n_{post\ failure}$	0.450	0.450

E = Modulus of Elasticity.
D = Density.
n = Poisson's ratio.

The values of the modulus of elasticity for the overburden were based on lab results obtained from core samples collected from the field site. To obtain the model inputs, an average of all field samples was calculated (3.0 x 10^9 kg/m² (6.4 x 10^8 lb/ft²)) and ratios of 1/20 for the overburden material and 1/200 for the coal material were applied to the average lab result. Prior research has shown that decreasing E_{lab} values by several orders of magnitude, results in a better representation of actual field conditions and accounts for rock mass effects (Voight 1970). Voight (1970), has also noted that lab results within the range of 4.88 x 10^6 to 4.88 x 10^8 kg/m² (1.0 x 10^6 and 1.0 x 10^8 psi) should be reduced by at least one order of magnitude. A value of 4.0 was inputted into the model for the postfailure ratio of $E_{in\ situ}/E_{failed}$. This ratio is simply a curve fitting parameter to match the field-measured maximum subsidence magnitude with that

derived from the model. In reality, the form of the subsidence profile predicted by the model is insensitive to the choice of elastic parameters. The excavation of coal is simulated using a bimodulus model. Initially, this material is assigned a very small modulus and a Poisson Ratio equal to zero, allowing the material to freely deform in the vertical direction. As the top and bottom of the panel contact each other, the modulus of elasticity value is increased to prevent interpenetration.

Figure 4 shows the subsidence profile determined by the model and the actual subsidence profile obtained from the collected field data. Although the two curves are not identical, the general trends of the curves are similar, with the maximum subsidence located above the two panels. This match is considered adequate for the subsequent hydrologic analyses.

Comparison of Vertical Displacements Determined by the Model and Field Data

Strain distributions are determined in the finite element model by solving the boundary value problem with appropriate constitutive relations. The subsidence profile is the surface manifestation of this continuous redistribution of strain surrounding the mined panel. Strains and displacements generated within the mesh, as a result of mining, may also be determined and analyzed. A comparison of vertical displacements, within the overburden, as generated by the model and the displacements measured at the field site are shown in tables 2-4.

Table 2.—V_{field} versus V_{model} (vertical displacements) for MPBX nos. 1 and 2

Borehole	Anchor	Height		V_{field}		V_{model}	
		m	ft	m	ft	m	ft
1	1	54.56	179	1.14	3.73	1.00	3.28
	2	50.60	166	1.16	3.79	1.01	3.30
	3	47.00	154	1.17	3.85	1.01	3.30
	4	38.10	125	1.30	4.26	1.01	3.30
	5	25.91	85	1.50	4.90	1.01	3.30
	6	16.15	53	F	F	1.00	3.28
	7	8.53	28	F	F	1.00	3.25
	8	6.10	20	F	F	1.00	3.25
2	1	64.62	212	0.57	1.87	0.50	1.63
	2	57.91	190	0.57	1.87	0.50	1.63
	3	55.17	181	0.60	1.98	0.50	1.63
	4	38.71	127	0.70	2.27	0.51	1.68
	5	27.74	91	1.64	5.37	0.51	1.68
	6	18.59	61	F	F	0.52	1.71
	7	9.45	31	F	F	0.53	1.73
	8	5.18	17	F	F	0.53	1.74

F - Anchor Failed
A - Anchor Number

Table 3—V_{field} versus V_{model} (vertical displacements) for MPBX nos. 3 and 4

Borehole	Anchor	Height		V_{field}		V_{model}	
		m	ft	m	ft	m	ft
3 . . .	1	70.10	230	0	0	0.09	0.29
	2	60.35	198	0	0	0.07	0.23
	3	36.90	121	0	0	0.07	0.22
	4	16.70	55	0	0	0.07	0.24
	5	7.01	23	0	0	0.07	0.24
	6	2.44	8	0	0	0.07	0.24
	7	0.91	3	0	0	0.07	0.24
	8	0.30	1	0	0	0.07	0.24
4 . . .	1	70.71	232	0	0	0.30	0.99
	2	58.52	192	0	0	0..27	0.88
	3	52.73	173	0	0	0.25	0.82
	4	36.88	121	0	0	0.21	0.70
	5	25.30	83	0	0	0.19	0.62
	6	13.41	44	0	0	0.16	0.52
	7	3.96	13	0	0	0.14	0.47
	8	1.83	6	0	0	0.12	0.41

F - Anchor Failed
A - Anchor Number

Table 4 —V_{field} versus V_{model} (vertical displacements) for MPBX nos. 5 and 6

Borehole	Anchor	Height		V_{field}		V_{model}	
		m	ft	m	ft	m	ft
5 . . .	1	83.21	273	1.21	3.97	0.56	1.85
	2	64.00	210	1.28	4.20	0.56	1.85
	3	50.30	165	1.43	4.70	0.56	1.84
	4	39.62	130	1.44	4.73	0.56	1.85
	5	31.10	102	1.53	5.01	0.56	1.85
	6	19.2	63	F	F	0.59	1.93
	7	10.36	34	F	F	0.60	1.96
	8	5.80	19	F	F	0.61	1.99
6 . . .	1	75.60	248	1.09	3.56	1.10	3.62
	2	62.79	206	F	F	1.11	3.63
	3	51.21	168	1.10	3.62	1.12	3.66
	4	38.10	125	F	F	1.12	3.66
	5	27.43	90	F	F	1.11	3.64
	6	19.50	64	1.14	3.73	1.11	3.61
	7	9.45	31	F	F	1.09	3.57
	8	3.96	13	F	F	1.08	3.53

F - Anchor Failed
A - Anchor Number

The predicted vertical displacement data obtained from the model for MPBXs Nos. 1, 2, 3, and 6 correlate well with the field data collected at the site (tables 2-4). The field displacement data for MPBX No. 4 (table 3) showed minimal vertical displacement, whereas, the predicted vertical displacements for these boreholes showed slightly higher magnitudes of displacement (.07 to .30 m) (0.22 to 0.99 ft). One contributing factor could be the difference between the predicted subsidence profile and the actual field data (figure 4). As displayed in the figure, the field data profile shows minimal amounts of subsidence occurring above the gate roads between the two panels. Whereas, the predicted subsidence profile shows a maximum of 0.15 m (0.5 ft) subsidence.

The anchors in MPBX No. 5 displayed larger magnitudes of vertical displacements as compared to those predicted by the model. Again, this is mainly attributed to the differing shapes of the subsidence profiles above the vicinity of the ribline over panel No. 2 (figure 4). The profile, developed from the model, shows less subsidence occurring above the ribline as compared to the profile determined from the field data. As shown in the figure, the maximum subsidence (field data) for panel No. 2 occurs near the ribline, whereas, the model predicted maximum subsidence to occur above the center of the panel. This nonclassical form of the subsidence profile is attributed to site-specific geological conditions that are undefined and therefore not incorporated into the homogeneously distributed material parameters used in the finite element evaluation of displacements.

Correlation of the Postmining Hydrologic Regime with Data Collected from Field Site

The postmining effects to the ground water system (figure 5), evaluated from the model, were determined though input of the parameters as shown in table 5.

Table 5.—Input parameters for determination of postmining ground water effects

Input Parameter	Overburden Upper Shale	Overburden Sandstone	Overburden Lower Shale
K_x, Horizontal Conductivity	7.01×10^{-8} m/s (2.3×10^{-7} ft/s)	7.01×10^{-5} m/s (2.3×10^{-4} ft/s)	7.01×10^{-9} m/s (2.3×10^{-8} ft/s)
K_y, Vertical Conductivity	7.01×10^{-8} m/s (2.3×10^{-7} ft/s)	7.01×10^{-5} m/s (2.3×10^{-4} ft/s)	7.01×10^{-9} m/s (2.3×10^{-8} ft/s)
S, Fracture Spacing	0.31 m (1 ft)	0.91 m (3 ft)	0.10 m (0.30 ft)
R_m, Modulus Reduction Factor	0.80	0.05	0.97
ν, Kinematic Viscosity	1.0×10^{-6} m²/s (1.0×10^{-5} ft²/s)	1.0×10^{-6} m²/s (1.0×10^{-5} ft²/s)	1.0×10^{-6} m²/s (1.0×10^{-5} ft²/s)
g, Acceleration of Gravity	9.81 m/s² (32.2 ft/s²)	9.81 m/s² (32.2 ft/s²)	9.81 m/s² (32.2 ft/s²)
I, Infiltration Rate	0.23 m/yr (0.75 ft/yr)	0	0

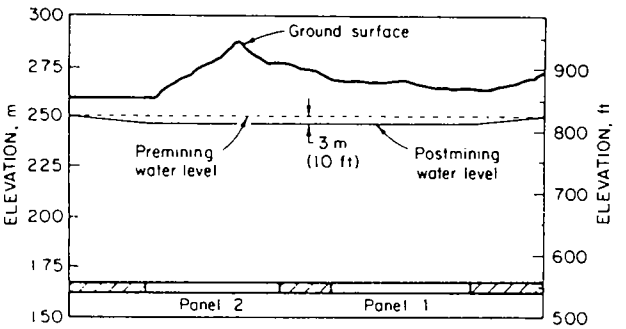

Figure 5. Postmining effects of ground water system - predicted by model. Flow rate - Q = 2,390 Lpm (630 gpm)

Numerous drawdown and recovery pumping tests were performed on all of the monitoring wells to determine hydraulic conductivity values. The hydraulic conductivity value of the 7.01×10^{-8} m/s (2.3×10^{-7} ft/s), upper overburden material (shale), for input to the model, was determined by averaging the hydraulic conductivity values measured at the site. The estimated hydraulic conductivity value, 7.01×10^{-5} m/s (2.3×10^{-4} ft/s) for the sandstone material was used as input into the model. This value was 3 orders of magnitude greater than the measured conductivity value obtained for the upper overburden material. This value of hydraulic conductivity was used because it is believed that the sandstone unit provides a much higher permeability than that of a shale unit. The hydraulic conductivity for the lower overburden material (shale) was one order of magnitude less than that of the upper material 7.01×10^{-9} m/s (2.3×10^{-8} ft/s). Prior USBM research has shown that fractures within the overburden rock mass commonly decrease in aperture and number with increasing depth (Walker 1986). Correspondingly, the rock mass at depth is assumed 'tighter' and consequently less conductive.

The fracture spacing values used are assumed, since no information regarding fracture spacings were determined at the field site. However, the selected values are consistent with those incorporated in other successful calibration studies (Elsworth 1994, Liu 1994, and Ouyang 1993). As mentioned earlier, the modulus reduction factor, R_m, reflects the partitioning of mining-induced strains between fractures and the porous matrix. In less stiff materials, the matrix accommodates proportionately more of the applied bulk strain than stiffer materials, where fracture closure dominates the mass response. Correspondingly, less competent materials, such as shales, return higher magnitudes of the modulus reduction factor. In the absence of field measured magnitudes, appropriate magnitudes of the modulus reduction factor are selected, reflecting these anticipated characteristics of behavior.

Flow rates entering the mine following the excavation of both panels were monitored by the mining company. Mine personnel estimated flow rates through the monitoring of a main sump located underground. Effects of surface area, lithology, and hydrogeology were incorporated into the analysis to determine flow rate estimates. This information provided a reasonable basis for the selection of several values for input into the model i.e., fracture spacing (s) and modulus reduction factor (R_m). The flow rate provided by the mining company was approximately 2,390 lpm (630 gpm). Values of R_m, determined through matching the flow rate, were 0.80 for material 1, 0.05 for material 2, and 0.97 for material 3. Choice of these parameters was clearly non-unique, but was predicted on the anticipated response of the lithologic units to straining and fracturing. The pre-existing fractures in the upper shale and the lower shale units have less effects on the postmining conductivities than those in the

sandstone layer unit. Actually, the shale material may be treated as a porous medium which is not sensitive to deformation. The difference between the two values is due to the effect of depth. The value of R_m is chosen as 0.05 for the sandstone unit because a large part of the extensional strain is applied to the fracture system and precipitates the largest possible change in conductivity.

The coarse finite element mesh, used to evaluate the subsidence profile and overburden displacements, was applied to determine the influence of mining-induced permeability changes on the ground water system. The predicted effects on the postmining ground water system determined by the model correlated well with the field data collected at the site as shown in figures 6-11. Figures 6-11 show water level fluctuations measured for Well Nos. 1-6, respectively. In addition, curves were added to the figures which show progression of the longwall face as a function of the overburden thickness. These values are expressed as the ratio of face position (FP) to overburden thickness (OB). For example, consider a longwall face that is moving towards a well but is 183 m (600 ft) away and the overburden at the site of the well is 61 m (200 ft). The FP/OB ratio is - 3. The negative value of FP/OB ratio indicates a premining position of the longwall face; zero indicates when the respective longwall face passed beneath the line of wells; and a positive value indicates postmining positions of the longwall face past the line of wells. If at another site a longwall face was approaching a well but was 366 m (1,200 ft) away and the thickness of the overburden at the site of the well was 122 m (400 ft), the FP/OB ratio is still -3. This curve allows one to compare well response at two different sites (conceivably in the same study area) without having too make complicated adjustments for differing overburden thicknesses. Under the applied boundary conditions, the model determined that a static decrease of 3.0 m (10 ft) in water level would occur due to the mining of both longwall panels. It should also be noted, that this decrease would occur without an infiltration rate inputted in the model. If an infiltration rate of 0.23 m/yr (0.75 ft/yr) is used, the regional model predicts that essentially no effects to the static phreatic surface would result. This infiltration rate is 25% of the total precipitation occurring at the site and correlates well with known infiltration rates of previous studies (Stoner 1983). Precipitation records provided by the mining company showed an average precipitation of 0.91 m (3.0 ft) which occurred at the site. Therefore, an infiltration rate of 0.23 m/yr (0.75 ft/yr) was used. Figure 5 shows the model prediction after excavation of both longwall panels.

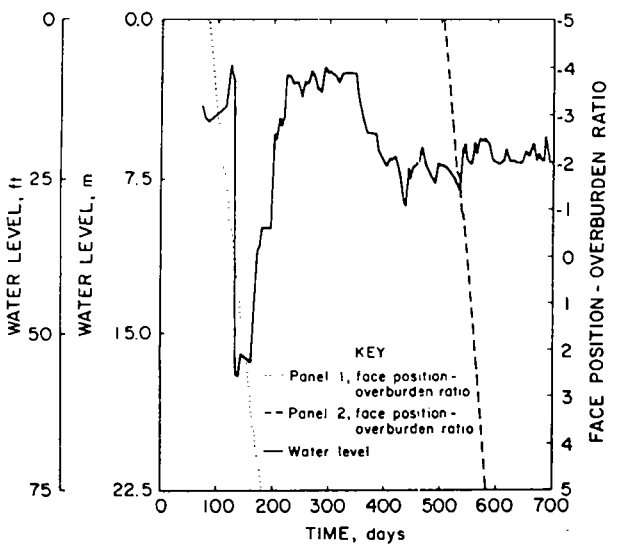

Figure 6. Comparison of water level change and face position to overburden ratio for Well 1

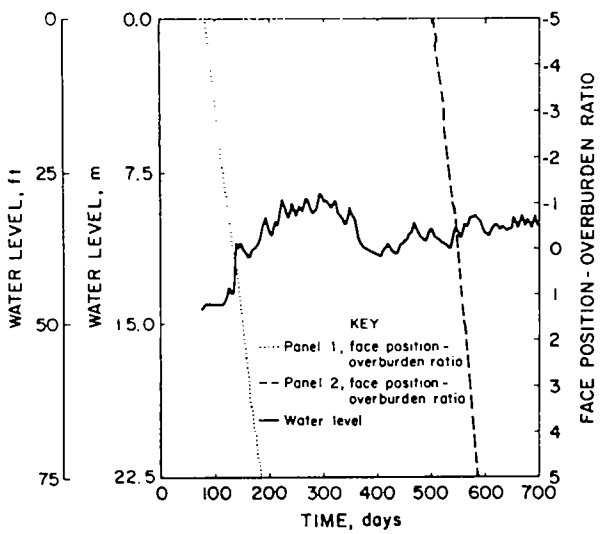

Figure 7. Comparison of water level change and face position to overburden ratio for Well 2

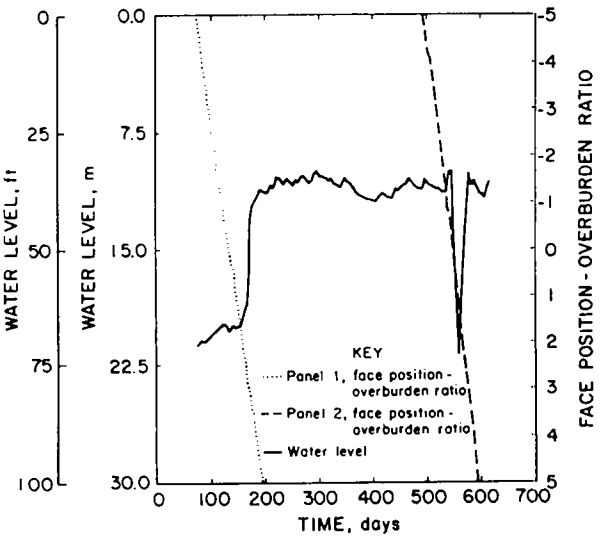

Figure 9. Comparison of water level change and face position to overburden ratio for Well 4

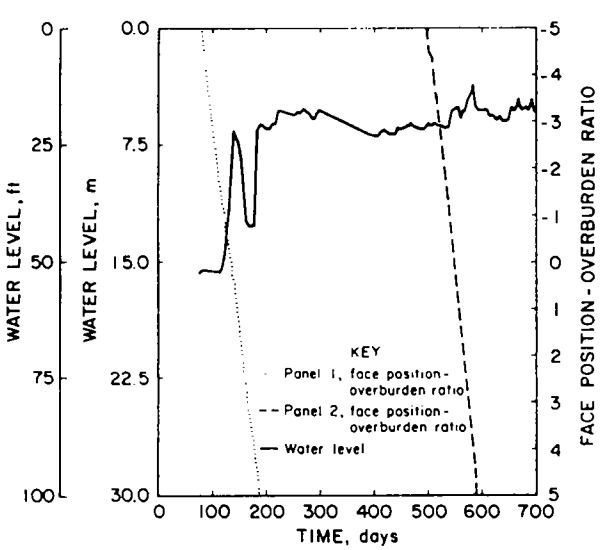

Figure 8. Comparison of water level change and face position to overburden ratio for Well 3

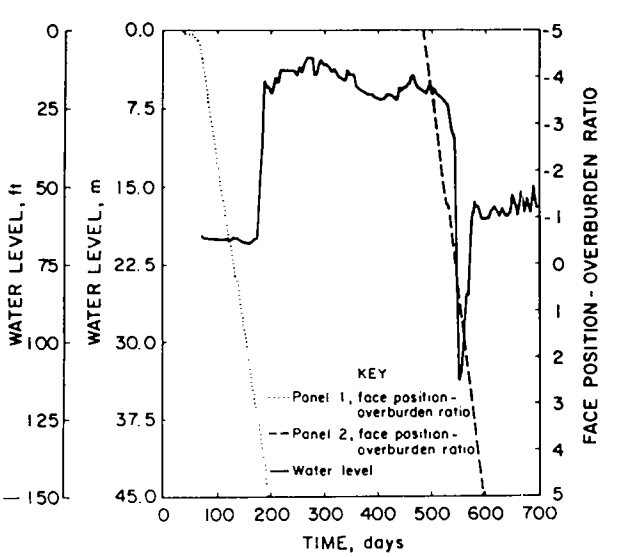

Figure 10. Comparison of water level change and face position to overburden ratio for Well 5

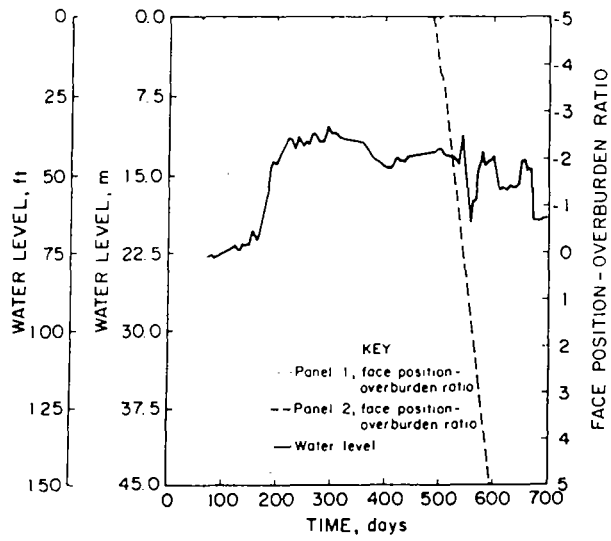

Figure 11. Comparison of water level change and face position to overburden ratio for Well 6

In addition to applying the regional model, described above, local behavior around the shallow well field (figure 12a) was also represented by a more refined model to accommodate more subtle water budget changes as a result of mining. The changes in hydraulic conductivities predicted from the initial calculations were applied to the zonation as defined in figure 12*b*. These zones represent average changes in hydraulic conductivities evaluated from the subsidence modeling. The refined mesh is capable of accurately representing local changes in the location of the phreatic surface. This mesh represents the region between the centerlines of both panels containing well nos. 5 and 6 within Zone I, well nos. 2-4 within Zone II, and well no. 1 within Zone III. No flow boundary conditions were specified along the base and on the left side (the centerline of panel No. 2) and constant head conditions were applied on the remaining vertical side (about 76.2 m (250 ft) away from the centerline of panel No. 1). The mesh, utilized uniform spacing, and was constructed of 326 nodes and 295 elements. Two different situations were simulated through use of the refined mesh. Fist, the same infiltration rate and premining ground water conditions were inputted, where the small-scale influences of topographically induced flow were accommodated. Secondly, the postmining hydraulic conductivity magnitudes of table 5 were incorporated to determine the anticipated postmining ground water levels. The magnitude of hydraulic conductivities were evaluated directly from the spatial distribution of strains, using equations 6 and 7. Average magnitudes of horizontal and vertical conductivities were then utilized in the refined model of figure 12. The changes in hydraulic conductivities are documented in table 6.

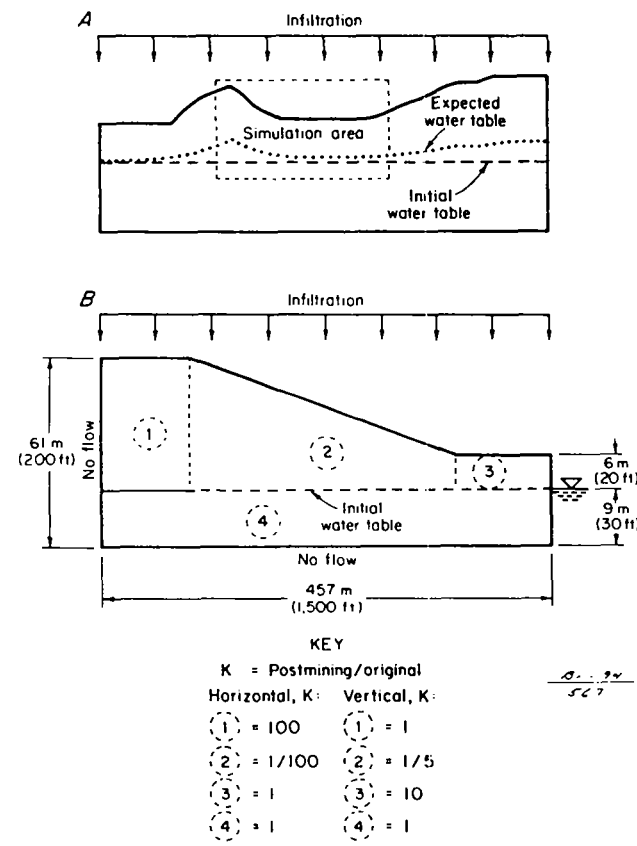

Figure 12. Model for topographic influence of infiltration on well elevations. A, comceptual; B, numerical

Table 6. Relative changes in hydraulic conductivity ($K_{postmining}/K_{original}$) as applied to the small-scale mesh

Zone	K_h	K_v
1	100	1
2	1/100	1/5
3	1	10
4	1	1

K_h = Horizontal conductivity.
K_v = Vertical conductivity.

The resulting postmining modification in the location of the phreatic surface is illustrated in figure 13. The subtle changes result from applying the same infiltration rate as used in the previous model run, however, the greater element density of this revised model highlights the influence of even minor topography and moderate changes in near surface hydraulic conductivities on the ground water system. The mining-induced development of hydraulic conductivities is such that water levels in the region of wells 5 and 6 remain the same, water levels in the region of wells 3 and 4 rise, and water levels in the region of wells 1 and 2 fall, relative to premining water levels. When the long-term recorded levels in these six wells are corrected relative to the control well (a well located approximately 425 m (1,400 ft) away from mining activity), this distribution of behavior, and of this magnitude is exactly as observed from the field measurements performed at the site (Matetic 1991) and as shown in figures 6-11.

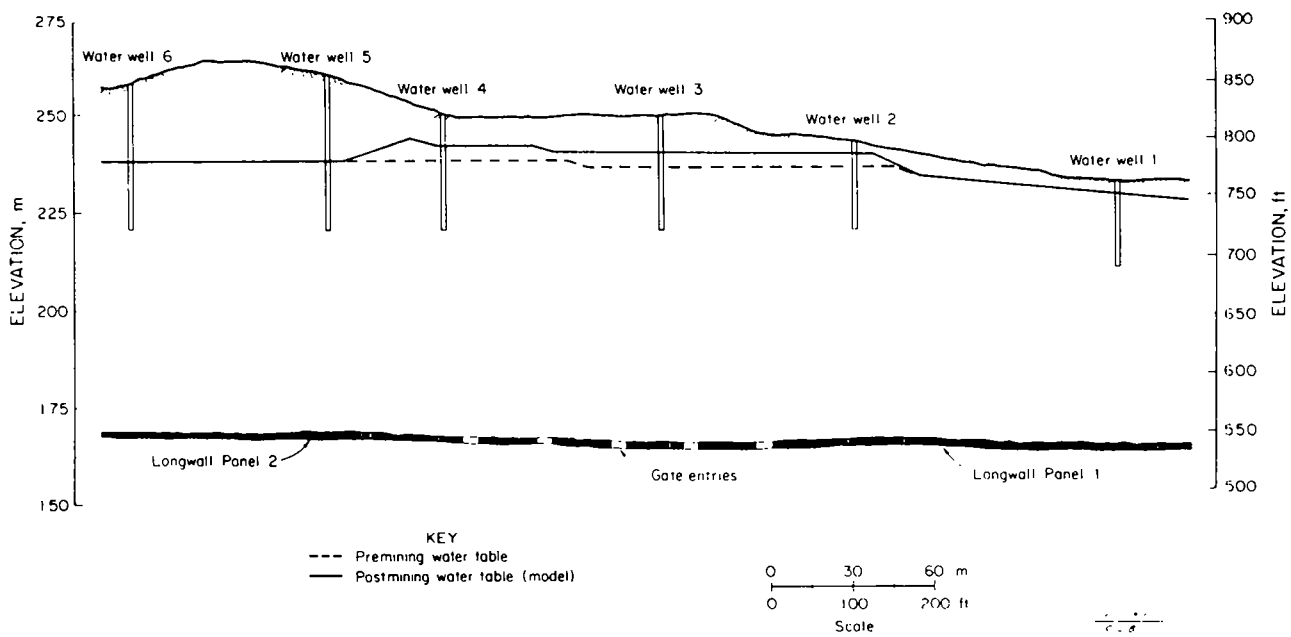

Figure 13. Postmining effects on ground water system using smaller-scale mesh around well area

Summary

1. The subsidence profile, as determined from the model, compares favorably with maximum subsidence measured at the field site.

2. The field data showed that minimal subsidence occurred above the gate roads between the two panels, while the predicted subsidence profile showed a maximum of 0.15 m (0.5 ft) subsidence at this location.

3. The predicted vertical displacement data obtained from the model for MPBX's Nos. 1, 2, 3, and 6 correlated well with the field data.

4. The field displacement data for MPBX No. 4 displayed minimal displacements, whereas, the predicted displacements for these boreholes showed slightly higher displacement. Also, MPBX No. 5 showed similar vertical displacements as compared to that predicted by the model.

5. The predicted effects of the postmining ground water system determined by the model correlated well with the field data collected at the site. Without infiltration added, the model determined that a static decrease of 3.0 m (10 ft) in water level would occur at the site due to the mining of both longwall panel nos. 1 and 2. If an infiltration rate was input to the model, no predicted effects for the long term would occur to the ground water regime which correlates well with field observations.

6. Where a fine mesh is used to define local changes in water budget within the well field area, the model is capable of replicating relatively subtle changes in long-term water levels. Minor local changes in the phreatic surface are consistent with the rolling topography of the site and the effects of mining-induced changes in hydraulic conductivity.

Conclusion

For this study, the numerical modeling results correlate favorably with the field data collected at the site. The surface subsidence information from the field, provided an excellent foundation for the modeling routine. If this information is not available, one should obtain and examine available subsidence prediction models to determine the profile information for the site. The assumptions made during the course of operating the model were based on knowledge of the subject area, experience with the model and insight gained from field tests. Again, if the field information is not available, the model routine requires some additional assumptions. Although a favorable correlation exists between the mining effects predicted and the field data collected, the authors feel that additional comparative studies at research sites with varying geology, longwall panel characteristics (thickness, width, etc.), hydrogeology, etc. should be performed to further substantiate the capabilities of the model.

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GROUNDWATER CONDITIONS AROUND A COAL-COMBUSTION RESIDUE SITE IN SOUTHERN ILLINOIS

by

Steven Esling, Merryn Fielding and Bradley Paul

Abstract. Coal-combustion residues, predominantly composed of scrubber sludge, served as backfill to an abandoned surface-mine pit. A low-relief valley now trends across the area that was once the old pit, replacing the former hummocky terrain left by the mining operations. Twelve wells, installed within and adjacent to the fill, provide data on hydraulic head at the base of the spoil, in the coal, and in the coal-combustion residues. The bases of the screens ranged in depth from 5.6 to 15.9 m. Three of the wells were sampled for groundwater chemistry. Hydraulic conductivity of the spoil from nine falling-head (slug) tests ranged from $1.9X10^{-7}$ to $4.1X10^{-3}$ cm/s, with a geometric mean of $1.8X10^{-4}$, somewhat greater than the $9.8X10^{-6}$ cm/s hydraulic conductivity of the coal-combustion residues. The mine spoil is heterogeneous, composed of disturbed surficial sediment, including loess and diamicton, blended with fragments of bedrock. Prior to reclamation, groundwater flowed away from the pit lake to topographic lows to the north, east, and south. Reclamation has not altered groundwater flow significantly. A two-dimensional numeric random-walk contaminant-transport model, based on 1) properties of the coal-combustion residue as determined from column studies, 2) data from a finite difference model of groundwater flow at the site, and 3) conservative estimates of dispersivity and porosity, suggests that leachate generated by the fill poses little threat to ambient groundwater quality.

Additional Key Words: Groundwater Modeling, Groundwater Quality, Mine Land Reclamation

Introduction

In an effort to meet new federal emission standards, power companies rely on scrubbers or employ new coal burning technologies, both of which lead to increased production of coal-combustion residues. Although these materials have numerous applications, including structural fill, light-weight aggregate, stabilization material, drilling fluid, and additive to Portland cement, production will exceed demand for the foreseeable future. Today, most excess residue produced from coal-combustion is either buried near the surface in a relatively dry state (landfill) or pumped into slurry ponds (Bahor and others, 1981), both of which take land suited for other purposes out of production, at least temporarily. Reclaiming abandoned strip-mines with these residues reduces the loss of productive land and returns otherwise non-productive and hazardous land to more productive use.

Coal-combustion residues, however, may produce a leachate containing hazardous substances. Numerous previous studies have investigated the environmental impact of coal-combustion residue disposal

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sites on surface water and groundwater quality (Villaume and others, 1983; Le Seur Spencer and Drake, 1987; Cherkauer, 1980; Hardy, 1981; Simsiman and others, 1987; Sakata, 1987; Theis and others, 1978; Beaver and others, 1987; Fruchter and others, 1988; Gerber, 1981; Le Seur, 1985; Hall, 1977; Rai and others, 1989; Rehage and Holcombe, 1990; Libicki, 1978; U.S. Waterways Experiment Station, 1979). Some of this research detected distinct contaminant plumes in the groundwater, downgradient from slurry ponds and landfills. In some cases, contaminants exceeded drinking water standards. Adriano and others (1980), Ferraiolo and others (1990), Theis and Marley (1979), and Theis and Gardner (1990) provided general reviews on residue disposal methods and environmental impacts.

This paper presents intermediate results of a long-term study of the environmental impact of filling an abandoned strip-mine pit in southern Illinois with coal-combustion residues. An unstable highwall associated with the pit threatened a county road. The immediate area lacked sufficient spoil needed to fill the strip-pit and establish a gentle graded contour. Coal-combustion residues, predominantly composed of flue-gas desulferization scrubber sludge with lesser amounts of fly ash, herein referred to as *residues*, served as a backfill.

Site Description

General Setting

The study area is an abandoned strip-mine located within the Herrin 7.5-Minute Topographic Quadrangle, just east of Energy, in Williamson County, Illinois. A private landfill and the Herrin Municipal landfill are located in the section to the north. The test site (figure 1) was an abandoned strip-pit lake that was partially filled with debris from the surrounding spoil piles and an adjacent highwall. The pit, which was about 315 m long, 50 m wide, and up to 12 m deep (as measured from the top of the highwall), was filled in stages. Initial work during the summer of 1993 reclaimed the western 90 m of the pit, with the residues filling the first 70 m of the pit behind a 20-m-wide spoil dam. Reclamation was completed during the summer of 1994, with residues filling the next 90 m and spoil filling the remaining 135 m of the pit. Two other abandoned pits are nearby, one to the west of the disposal site (West Pit) and the other to the south and east (South Pit).

Pre-strip-mining topographic maps clearly show that the test site was located on a drainage divide. Flow from the adjacent drainage basins emptied into the Big Muddy River to the north. At present, no distinct surface drainage flows away from the project site, as surface mining operations have disrupted the original drainage pattern and created a hummocky terrain, characterized by numerous small hills, swales, and depressions. Reclamation associated with the placement of the coal-combustion residues has created a more subdued topography with a gentle slope trending to the east in place of the abandoned pit.

Geologic Setting

The test site is on the southern edge of the Illinois Basin and lies within the outcrop zone of the Carbondale Formation (Pennsylvanian). The Paleozoic strata dip gently to the north toward the center of basin. No major faults exist within a 1.6 km radius of the project site. About 5 km to the west, however, several distinct fault systems cut the Pennsylvanian rocks. The Carbondale Formation is characterized by numerous shale, sandstone, limestone and coal members, many of which are not laterally continuous. Complex depositional environments resulted in abrupt lateral facies changes (Willman and others; 1975).

The undisturbed surficial deposits of the study area originated directly and indirectly from glacial processes. Less than 4 m of Illinoian Age glacial diamicton (Glasford Formation) and 2 m of Wisconsinan Age loess (Peoria and Roxana Silts)

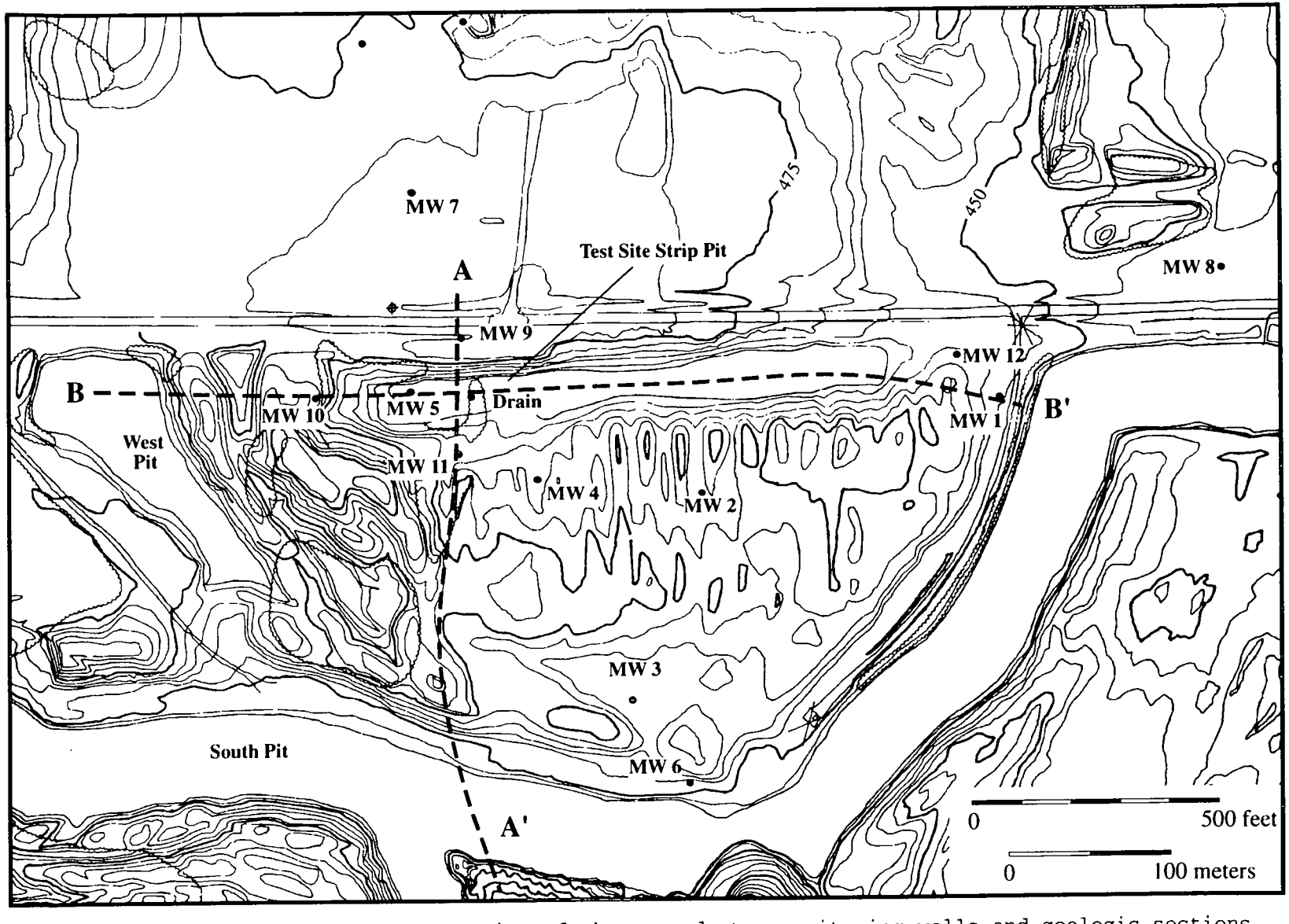

Figure 1. Topographic setting and location of the groundwater monitoring wells and geologic sections. Contour interval is 5 feet.

unconformably overlie the Pennsylvanian strata (Willman and others, 1975).

Strip mining in the study area removed up to six feet of the Herrin Coal seam and all overlying strata, with subsequent replacement of the overburden as spoil. The top of the Herrin Coal is generally at an elevation of about 137 m in the area of the test site. Mine spoil, a mixture of the natural surficial materials and bedrock fragments, constitutes the bulk of the materials on the land surface at the present time. In some places, the spoil consists almost entirely of rock fragments. In other areas, fine-textured matrix materials support the rock fragments.

Two geologic cross-sections (AA', figure 2, and BB', figure 3) constructed from data collected during the installation of monitoring wells, illustrate the general stratigraphy of the study area. Mine spoil lies unconformably on top of strata which underlie the Herrin Coal. Strip-mining operations apparently never removed the coal from beneath the road, but excavation of the highwall proceeded up to its edge.

The Herrin Coal remains in only a few places near the test site. Mining in these areas was either not economic or would have destroyed roads or town structures. Abandoned underground mines also exist near the project site. About 400 m to the north of the test site, the Herrin and Springfield Coals were removed from a room-and-pillar mine, which has since partially collapsed, leaving small subsidence troughs on the surface (Patrick Engineering, 1990).

Hydrologic Setting

Pryor (1956) studied the availability of groundwater in the southern Illinois area, including Williamson county. The report makes no reference to the quality of the water from subsurface supplies and suggests that bedrock units near the test site do not generally yield water, although Pennsylvanian strata elsewhere in the county are capable of sustaining small domestic supplies from depths of 15 to 250 m.

Environmental studies associated with the landfill north of the test site indicate that a shallow unconfined flow system has developed in the spoil and upper, more permeable bedrock units. Reports associated with these projects suggest approximately 3 to 8 m of mine spoil just above the shale unit which presumably underlies the Herrin Coal. Saturated conditions were encountered in all the wells in the unconsolidated materials, suggesting that the shales retard the flow of water moving through the mine spoil. Hydraulic conductivity measured in the field ranged from 10^{-4} to 10^{-6} cm/s, with the higher values in wells screened in coal and black shale (Patrick Engineering, 1990). The hydraulic head data from the monitoring wells suggest groundwater flows to the north at the southern boundary of the landfill and to the south at its northern boundary. The northern and southern flows converge and flow to the east near the center of the landfill (Patrick Engineering, 1990).

Field Investigations

Well Survey

We were unable to locate anyone utilizing a groundwater supply within a 1.6 km radius of the test site. Well logs on file with the Illinois State Geological Survey (ISGS) and Illinois State Water Survey (ISWS) indicate that some water wells were constructed at the turn of the century, but the more recent logs on file were for coal-, gas- or oil- exploration boreholes. A few of the logs of the exploration boreholes suggest brackish (non-potable) groundwater at depths greater than 150 m. Just outside the 1.6 km radius, several water wells were drilled and subsequently abandoned because of poor production.

Monitoring Well Installation

A total of twelve groundwater monitoring wells were installed in the study area for this project (Table 1). Three of the wells were designed to

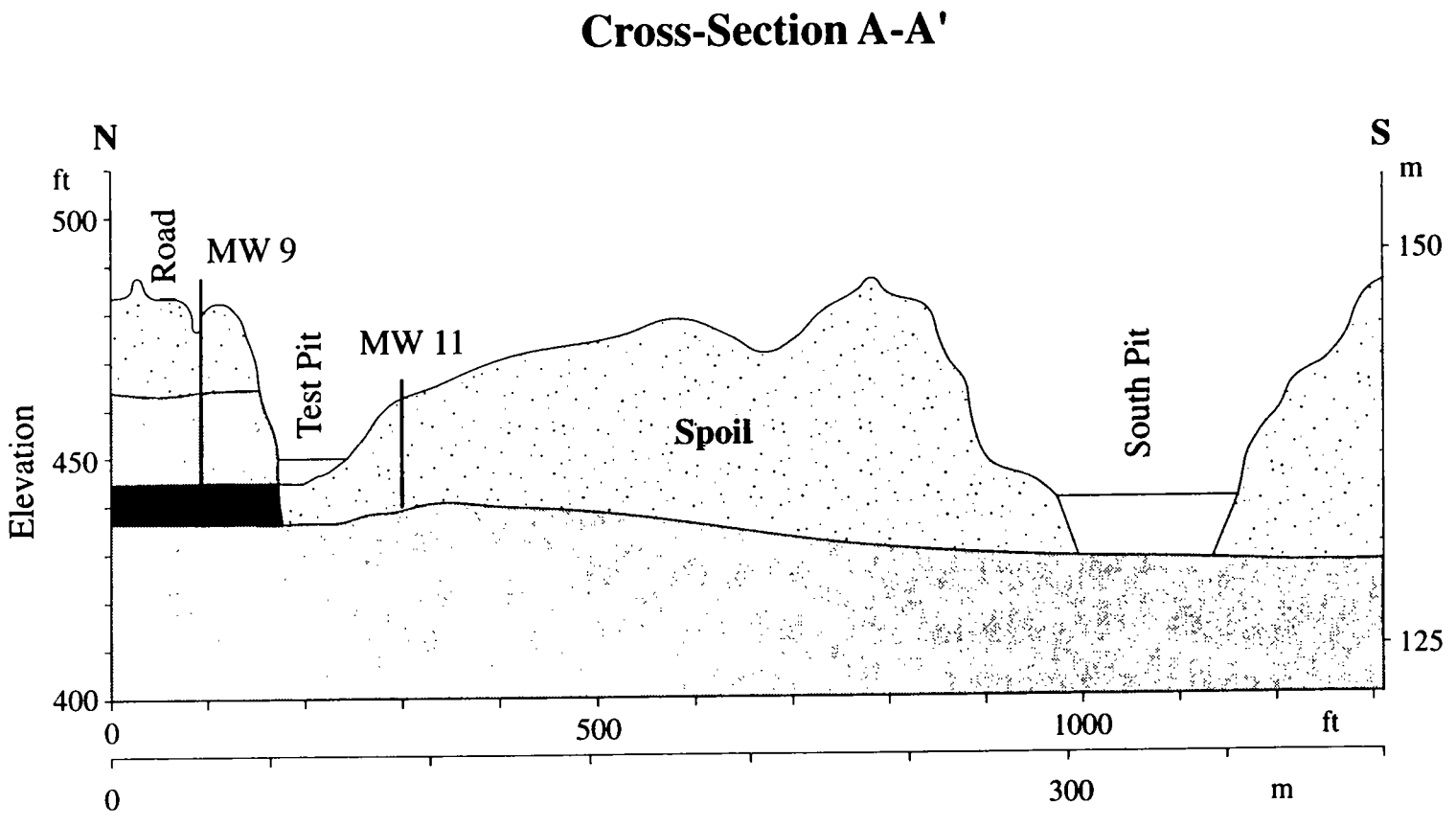

Figure 2. Cross-section showing general stratigraphic relationships along a north-south line through the site. The unit shaded in black is the Herrin Coal. See figure 1 for the location of the section. Verical exaggeration is about 4.95.

Cross-Section B - B'

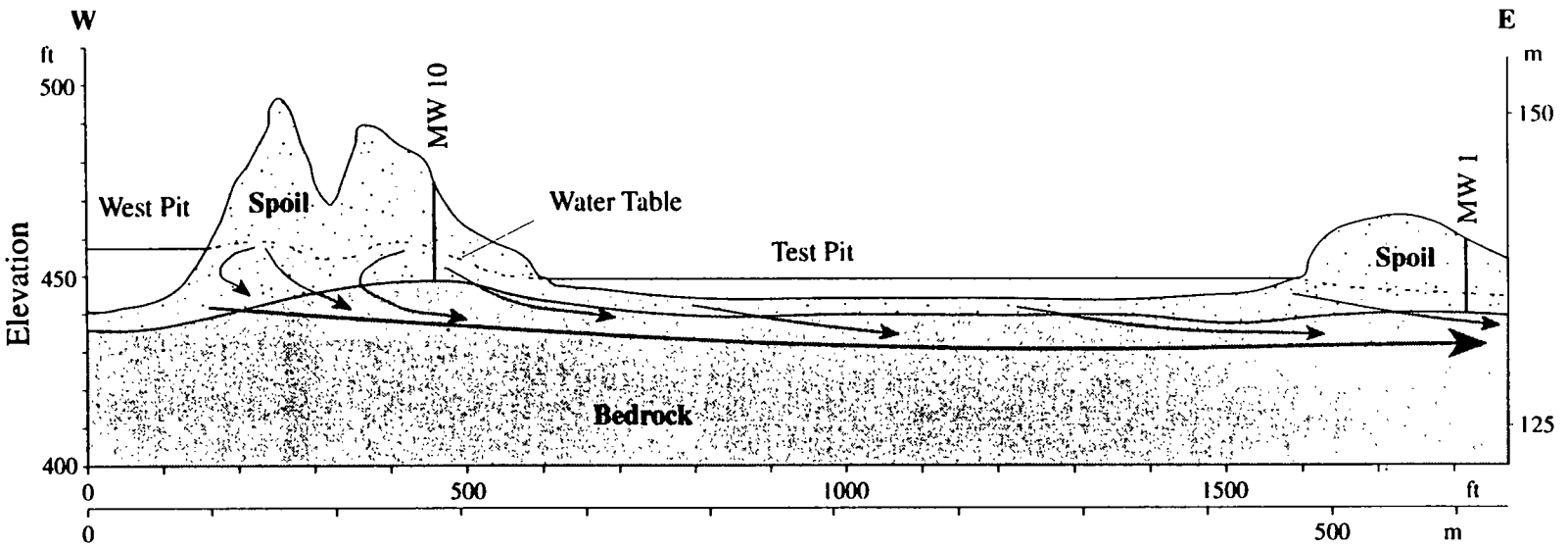

Figure 3. Cross-section showing general stratigraphic relationships and a hypothetical vertical flow system for the study area along a west-east line. See figure 1 for the location of the cross-section. Vertical exageration is about 5.46.

Table 1. Hydraulic conductivity measurements on monitoring wells near the test site.

Monitoring Well	Screened Interval	Maximum Depth (m)	Drilling Method	Hydraulic Conductivity (cm/s)
MW 1	Spoil	5.61	5.25-in Hollow-Stem Auger	$8.4X10^{-5}$
MW 2	Spoil	7.15	5.25-in Hollow-Stem Auger	$4.1X10^{-3}$
MW 3	Spoil	5.63	5.25-in Hollow-Stem Auger	$3.3X10^{-5}$
MW 4	Spoil	5.78	5.25-in Hollow-Stem Auger	$1.8X10^{-3}$
MW 5	Residues	10.32	5.25-in Hollow-Stem Auger	$9.8X10^{-6}$
MW 6	Spoil	5.91	5.25-in Hollow-Stem Auger	$1.9X10^{-7}$
MW 7	Fill/ Shale	10.36	5.25-in Hollow-Stem Auger	
MW 8	Spoil	3.94	5.25-in Hollow-Stem Auger	$8.4X10^{-5}$
MW 9	Coal	12.27	Mud Rotary	$6.8X10^{-5}$
MW 10	Spoil	15.89	Mud Rotary	$5.0X10^{-4}$
MW 11	Spoil	6.25	Backhoe	$1.9X10^{-3}$
MW 12	Spoil	9.24	6-in Hollow-Stem Auger	$5.7X10^{-4}$

sample water quality. These wells, herein referred to as MW 9, MW 10, and MW 11, surround the west end of the fill (figure 1). The installation of MW 9 and MW 10 followed accepted guidelines (USEPA, 1986). Monitoring well MW 11 was installed in a spoil pit excavated with a backhoe. A screen and casing (10.2 cm inside diameter(ID)) was centered in a 25.4 cm ID protective polyvinyl chloride (PVC) pipe prior to backfilling the trench. A successful sand pack around the screen was created by alternately adding a few feet of sand to the annular space between the 25.4 and 10.2 cm pipes, then allowing spoil to collapse around the sand by hydraulically lifting the protective pipe. Filling the remaining annular space continued through the same process, first with bentonite pellets and then with a bentonite cement slurry.

The remaining nine monitoring wells (figure 1) only provide data on

hydraulic head, and installation methods did not always follow USEPA guidelines. Table 1 summarizes the drilling method. Monitoring well casings were 5.1 cm ID PVC screen and casing. The annular space between the casing and borehole walls was filled with sand to a level at least 30 cm above the top of the screen. A cap of bentonite pellets with a minimum thickness of 30 cm was constructed above the sand pack, followed by either a bentonite cement slurry or a mixture of bentonite and well cuttings until a depth of about one meter. The remaining annular space was filled with either cement or concrete. A plastic outer casing protects the monitoring wells at the surface.

Wells were developed by alternately surging the well with a plug and bailing. All well elevations as well as staff gauges in the lakes within the test site and to the west and south were surveyed to a common datum. Two wells installed for monitoring hydrologic conditions in and around the private landfill located immediately north of the test site provide additional data on hydraulic head.

A 15-m-long drain was constructed in the west end of the test site prior to the initial reclamation work. The drain is a slotted pipe surrounded by quartz silica sand, with access through a vertical standpipe installed in the spoil dam. The drain captures leachate from the residues, prior to any natural attenuation, and provides samples from what is in effect a field-scale leachate column test.

Hydraulic Conductivity Measurements

Falling-head tests (slug test) were conducted on most wells to collect data on the hydraulic conductivity of the spoil and bedrock (Table 1). A pressure transducer connected to a datalogger with a cable was placed below the water level. A sealed cylindrical plug constructed of PVC and filled with sand was then dropped into the well, raising water level instantaneously. This method prevents contamination of the well by external sources of water. Data on the hydraulic head in the well was recorded by the datalogger along with the time required for the water level to return to within 80 percent of its pre-test level. Data on head and corresponding time were transferred to a personal computer, processed with a spreadsheet program, and analyzed following the procedures described by Hvorslev (1951) to obtain hydraulic conductivity.

Groundwater Hydrology

Flow

This section describes the conceptual model of the groundwater flow system for the project site, as well as the topographic and geologic factors that influence the flow system. The model of flow conditions is based on geologic data collected during the installation of the monitoring wells, previous geologic studies of the area, and hydraulic head data from the lakes and monitoring wells. Other observations supplemented these data, such as the locations of seeps and the depths to saturated conditions noted while drilling boreholes for monitoring well installation.

Figure 4 is a generalized map of hydraulic head at or just below the water table prior to any reclamation work. The map is based on head data collected on November 20, 1992, from the lakes and the seven monitoring wells that existed at that time. The head contours indicate that the lake to the west and within the test site recharge the shallow groundwater system. The lake to the south may also be a recharge point, but such an interpretation would require additional head data farther to the south. Water quality data also suggest that the lakes recharge the shallow groundwater. Lake-water quality is better than that from the sampled monitoring wells, indicating a significant influx of meteoric water. The quality of the water flowing from the lakes degrades as it reacts with the spoil.

If the lakes are recharge points, then MW 2 shows what at first seems to be an anomalous head, 8.5 cm higher

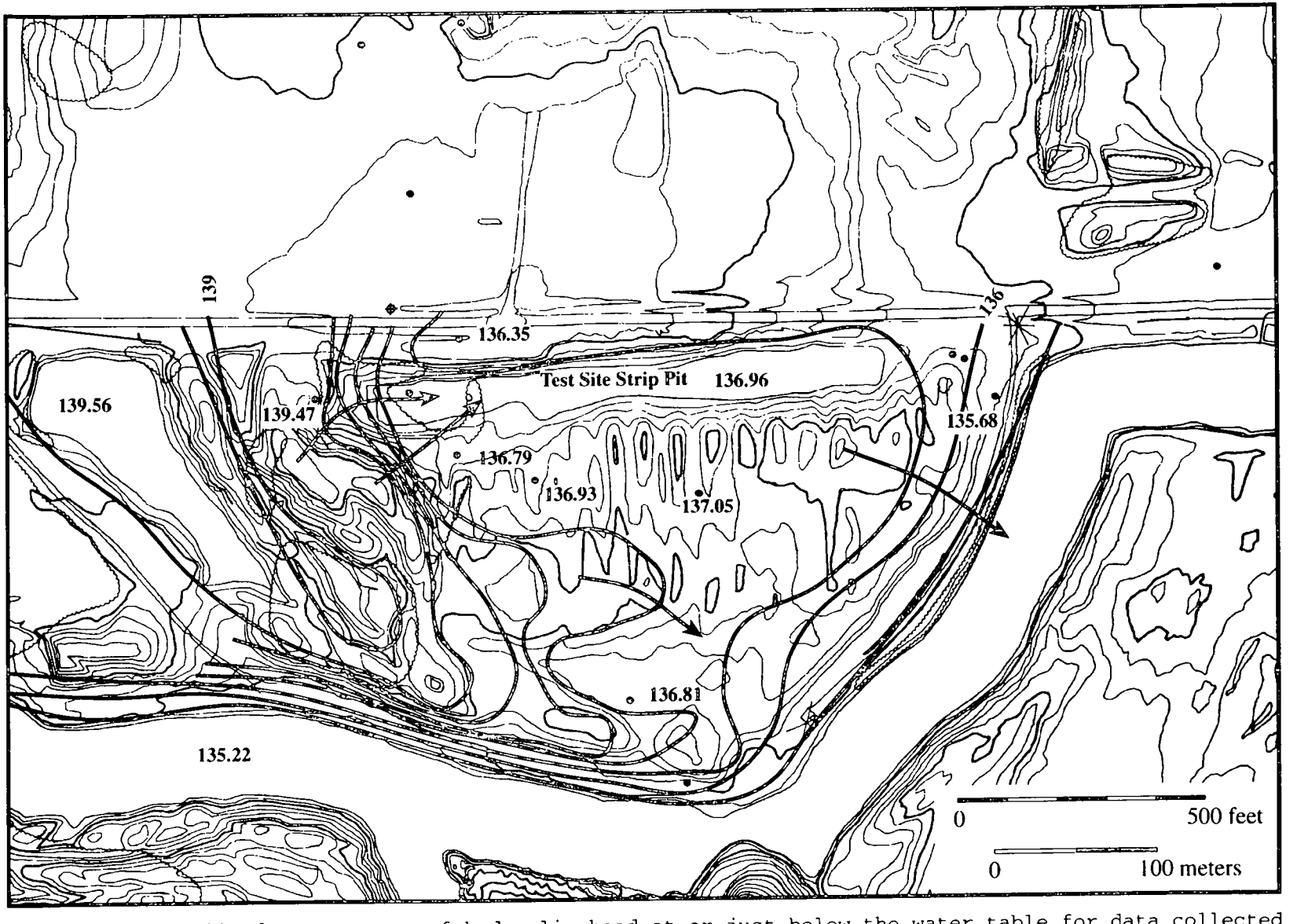

Figure 4. Generalized contour map of hydraulic head at or just below the water table for data collected 11/20/92. Arrows indicate flow direction. Contour interval is 0.6 m.

than the level of the lake water in the test site. Topography, however, influences the general flow pattern at a scale smaller than the map of the groundwater table. If more data were available and the contour interval of head were decreased, smaller local flow systems would become apparent. Typically in a humid temperate climate, like that of southern Illinois, the water table mimics the surface topography. Groundwater recharge occurs over divides, with groundwater discharging into the lows. Observed seeps in the study area suggest this type of system is operating. MW 2 is located near the crest of a spoil pile, and this well is probably influenced by local recharge.

Generally, groundwater flows to the east. Groundwater flows radially away from the lakes and from the center of prominent divides for a short distance before it converges with the general flow system. We need to emphasize that the map is a generalization of a complex flow pattern, ignoring the heterogeneous nature of the spoil. Fractures which can transmit significant quantities of groundwater may be present in the more cohesive spoil. The overall trend, however, does fit the interpretation of flow for the Herrin Landfill located to the northeast (Patrick Engineering, 1990). Flow lines on figure 4 show the shallow flow system at elevations very near the water table. Diagrams like this, however, can cause confusion because they imply horizontal flow when in fact vertical components of flow exist, especially in areas characterized by significant local relief. Figure 3, cross-section BB' (parallel to the interpreted general flow direction from west to east) illustrates the vertical components of flow expected beneath the upland areas.

Why does groundwater flow toward the east? First, mining east of the study area has removed the bedrock units down to the Springfield Coal (Patrick Engineering, 1990). The deep strip-pits and underground mines act as a sink, transmitting significant quantities of groundwater to surface drainage. Second, the study area is located on a topographic high and the elevated water table associated with this high is driving flow toward the east.

Reclamation has altered the original topography in the study area, but the general trends in groundwater flow have not shifted significantly. A map of hydraulic head from data collected in October 1994, after both stages of reclamation, still suggests flow to the east (figure 5).

The maps of hydraulic head were constructed from data collected from monitoring wells screened at the base of the spoil and in the coal. Investigations on the Herrin Landfill (Patrick Engineering, 1990) installed wells into the shallow bedrock. Hydraulic conductivity measurements conducted on these wells suggest that the shallow bedrock and spoil have hydraulic conductivities of similar magnitude in places. If the spoil overlies bedrock units with a lower hydraulic conductivity (such as gray shale), groundwater may concentrate above the bedrock, with flow along the spoil/bedrock interface. If, on the other hand, the spoil overlies rock with comparable hydraulic conductivity, groundwater would flow in the spoil and shallow bedrock, as indicated on figure 3.

The hydraulic conductivity measured in MW 9 is comparable to other values determined in coal (Table 1). Stone and Snoeberger (1977) measured a maximum hydraulic conductivity of 3.1×10^{-4} cm/s for a shallow Wyoming coal. Stoner (1981) reported hydraulic conductivities ranging from 2.3×10^{-4} to 8.1×10^{-5} cm/s for a shallow coal unit in Montana. The hydraulic conductivity determined in the wells with screens in spoil are comparable to those determined for spoil in other environmental studies in the area (Patrick Engineering, 1990). Table 2 summarizes the representative hydraulic conductivities for the various materials encountered in and around the test site.

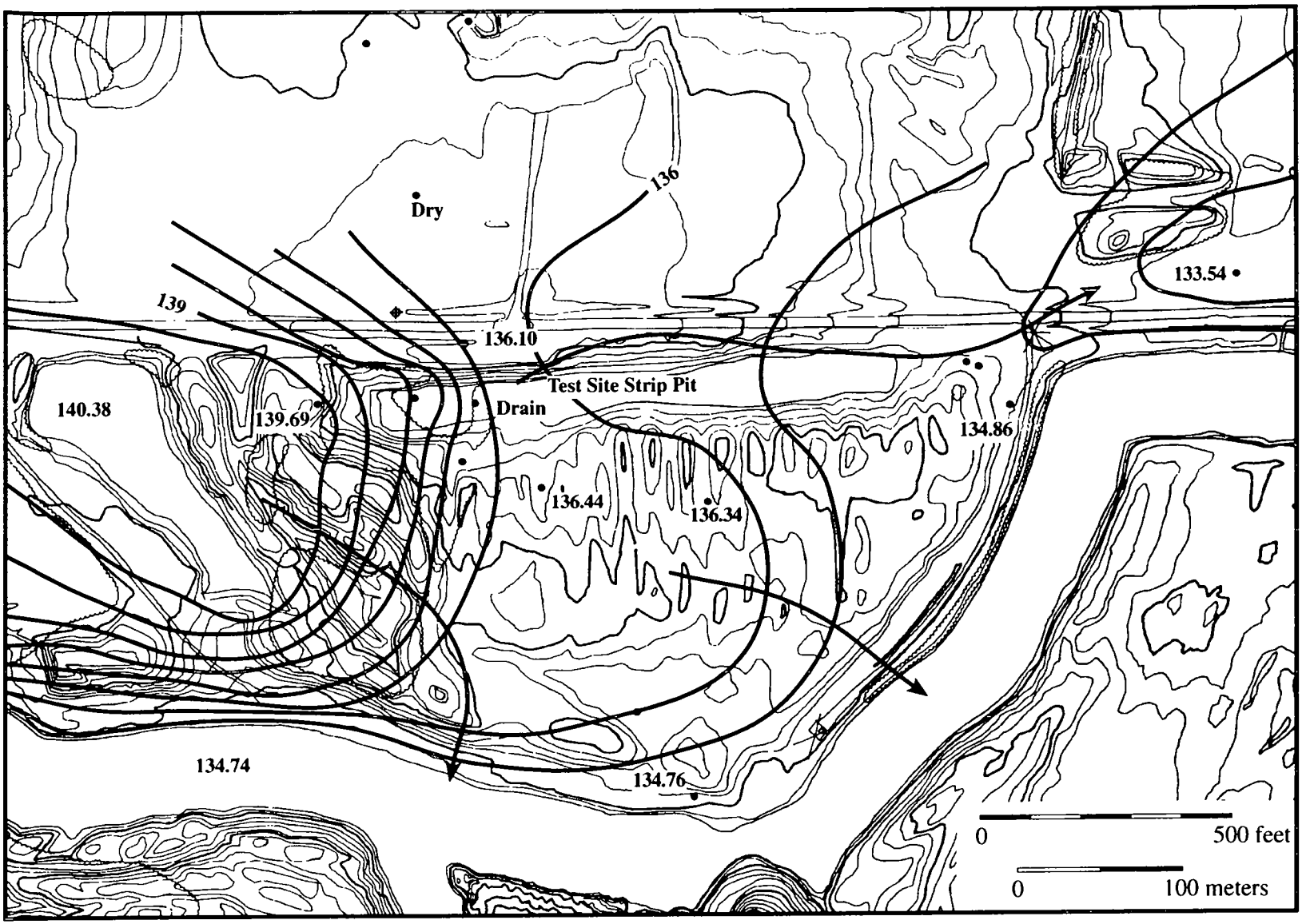

Figure 5. Generalized contour map of hydraulic head at or just below the water table for data collected 10/3/94. Arrows indicate flow direction. Contour interval is 0.6 m.

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SESSION 3: MINE SUBSIDENCE AND DAMAGE MITIGATION

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USE OF ANALOG GROUND MEDIUM OF TAYLOR-SCHNEEBELI TYPE FOR MODELLING OF MULTIPARAMETRIC DEFORMATIONS ARISING IN MINING-INFLUENCED AREAS[1]

by
Kazimierz Klosek[2]

Abstract: The results of the author's several years of model research using analog ground medium of Taylor-Schneebeli type to the multiparametric simulations of mining area deformations are presented in the paper. On the basis of the model similarity criteria the complete usefulness was proven of this research approach to the analysis of process of joint action of the mining rock mass and ground structures, underground infrastructures, engineering structures and road surface, also with use of geosynthetics.

Additional Key Words:
- model similarity criteria
- ground analog
- model research
- mining substrate
- kinematics of the disintegrated medium.

Introduction

A number of phenomena happening inside the rock mass and on its surface are impossible to record and to conduct direct surveys of analyse or research on the theoretical grounds. This is due to technical and economical reasons, as well as space-time limitations. Under these circumstances model research, conducted with the use of the equivalent and loose material, is gaining importance. The use of the equivalent materials (sand, gypsum, lime, borax, water mixtures, etc.) requires suitable selection of material constants in order to maintain the similarity laws. As a final result, this research allows to analyse not only the dislocations and deformations, but also forces and stresses occurring in the medium. Although the main domain of its application is ground and exploitation excavation areas [16], it remains beyond the scope of this paper. Models built of media, especially of sand allow conducting research limited to the kinematics of medium reduced to the recording of dislocations and displacements in the medium, caused by initially inflicted limiting displacements. Each of these groups of research has its advantages, but also substantial disadvantages. The paper presents a compromising approach to this problem which connects basic advantages of both main research trends by using the so-called Analog Ground Medium of Taylor-Schneebeli Type [8,15]. The possibility of full optical recording of undisturbed character of deformation of mining area surface and structures on the subsoil is given special consideration.

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THE CRITERIA OF MODELLING SIMILARITY

There are two basic rules of modelling similarity in the model research. The first one is the rule of geometric similarity based on fact that dimensions of modelled space are in proper relation (modelling scale) to the real object. The second is the rule of mechanical similarity, in form of three conditions:

- kinematic model similarity,
- dynamic model similarity,
- rheological (time) model similarity.

The kinematic similarity condition require the relative dislocations of the corresponding points in the model and in nature to be in geometrical interdependence. The comprehensive preliminary research proved [4,6] that corresponding parts of superficial zone of rock mass and of the Taylor-Schneebeli model are the subject to the same linear and formational deformations under identical excitations. For the process of modelling of dislocations of the surface of mining area for dynamic subsiding through, the conformance is shown in Fig.1.

The description of the profile of the subsiding through appearing on the surface as a result of the simulated exploitation show a full correspondence to the Gauss curves (normal distribution curves). This corresponds to the recorded distribution of real effects in the ground [9,10,16]. The other dislocations and deformations, which describe basic deformation parameters of surface, such as depression (W), sloping (T), curvature (K), horizontal dislocation (U) and unitary horizontal deformation (ε) are also fully corresponding. The theoretical shapes of these functions and their interdependence are shown, among others, in [2,3]. The fulfilment of this principal kinematic criterion of similarity guarantees the proper simulation of the behaviour of model surface objects founded on the ground analog T-S (Taylor-Schneebeli) type and related to the planar problems. The condition of dynamical similarity requires the ratio of model and real forces be the same as the ratio of mass and accelerations product, described by the relationship:

$$\frac{F_N}{F_M} = \frac{m_N \cdot a_N}{m_M \cdot a_M} \qquad (1)$$

where: a_N -acceleration of mass m_N

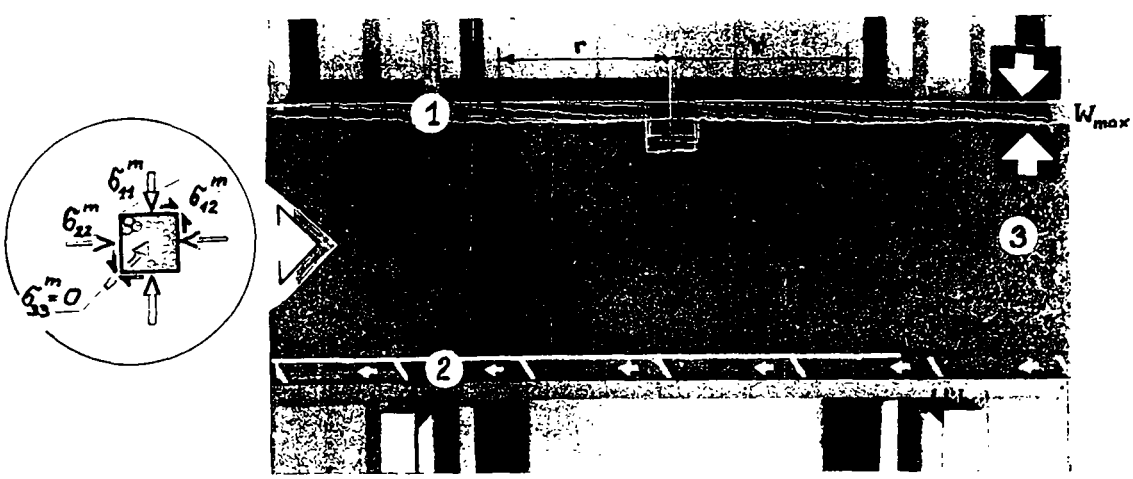


Figure 1. Dynamic subsiding through of a depression
(1) for variable position of exploitation front
(2) in an analog model of rock mass
(3) Taylor-Schneebeli (T-S) type.

under the force F_N in nature.
a_M -acceleration of mass m_M under effect of force F_M in the model.

This condition implies the constant value of the unit mass ratio, the masses being different by the factor b:

$$b = \frac{m_N}{m_M} = const \qquad (1a)$$

In the case of T-S type analog this factor is b=1. The condition of rheological similarity requires that the quotient of deformation velocity in nature and in laboratory was a constant value:

$$\frac{\dot{\varepsilon}_N}{\dot{\varepsilon}_M} = const \qquad (2)$$

Because the tested model was a granular rock medium and its cohesion was zero, this criterion was easily met. The physico-mechanical parameters of the materials used as equivalent should fulfil a number of additional conditions, calculated theoretically basing on the rules of modelling similarities. However, obtaining the wanted values of most of the parameters is very difficult and practically currently not possible. Therefore in practise the rule of functional characteristic is applied. It assumes as a basis of the mapping such mechanical index, which in analysed phenomena plays a vital role. It has been assumed that the principle will concern the preservation of the similarity of the phenomena of ground shearing as a basic strength criterion for this dispersed medium.

THE DESCRIPTION OF THE ANALOG GROUND MEDIUM.

The T-S type analog medium consist of a mixture of cylindrical rollers of two diameters: $\varnothing$4mm and $\varnothing$6mm, L=47mm long and weight by volume of γ=1.4G/cm^3. The side surfaces of the rollers are fine finished, ensuring the full stability of the pile, which then not need head walls. The assumed ground model ensured exactly flat state of strains and deformations. The Schneebeli's experiments [15] have shown that the pile of circular rollers of different diameters fulfil the condition of Coulomb's limiting equilibrium [12,13] . The necessary condition for obtaining the macroscopic and isotropic structure of the medium is the use of a set of rollers of different diameters [1,11]. Deformations of the single-fraction medium have discrete character, caused by structural anisotropy of the homogeneous pile. The anisotropy of the structure leads to the law of anisotropic elasticity of the analog making the research difficult and its results hard to interpret. The example of that is presented in interpret [7]. The conditions of model similarity of the mixed task require the fulfilment of adequate dependencies between strains, weights by volume, strength and geometric parameters of the model and the real object. This can be described as:

$$\alpha_\sigma = \alpha_\gamma \cdot \alpha_L \cong 1$$
$$\alpha_\gamma = \frac{\alpha_\sigma}{\alpha_L} \qquad (3)$$
$$\alpha_\phi \cong 1$$

where: $\alpha_{\sigma,L,\gamma,\varnothing}$:the scale of strains, lengths, weights by volume, internal friction angle. After substitutions it is possible to describe the state of strains in nature on the basis of the ground analog parameters:

$$\frac{\alpha_\sigma}{\alpha_L}\left(\frac{\partial\sigma_{22}}{\partial x_2} + \frac{\partial\sigma_{12}}{\partial x_1}\right)m = 0$$
$$\frac{\alpha_\sigma}{\alpha_L}\left(\frac{\partial\sigma_{11}}{\partial x_1} + \frac{\partial\sigma_{12}}{\partial x_2}\right)m = \alpha_\gamma \cdot \gamma_m \qquad (4)$$
$$\frac{\alpha_\sigma}{\alpha_L^2}\nabla^2(\sigma_{11} + \sigma_{22})m = 0$$
$$(\sigma_{11} - \sigma_{22})m = (\sigma_{11} + \sigma_{22})m \cdot \alpha_\varphi \cdot \sin\varphi_m$$

The parameters with index „m" correspond to the model. The first two equations describe the limiting equilibrium state, the next concern the condition of inseparability and the Coulomb-Mohr plasticity criterion. The homomorphic model of T-S type ground does not meet exactly all of the similarity model conditions. The research results are burdened with certain errors resulting from non-fulfilled criterion of that weight strain scale and geometric similarity of grains criterion. However it does not influence significantly the results, especially of the preliminary, qualitative estimation of the investigated effects of interrelation of structure and mining deformed ground.

USE OF THE ANALOG GROUND FOR THE MODELLING OF THE INFLUENCE OF UNDERGROUND MINING ON SELECTED OBJECTS ON SURFACE

The research carried out by the author using the T-S type analog medium preceded the complementary and verifying experiments realised with the use of sand in a specially constructed box apparatus (Fig. 2, [14]).

This apparatus made possible the physical realisation of the basic index of mining deformation of a terrain. The index were the unitary horizontal deformations of compacting and slackening character ε.

The direct recording of the kinematic effects and verifying the computational models was in this case impossible. Only the T-S analog placed on a specially constructed research stand shown in Fig. 3 makes the recording of these effects fully possible. In the model shown in Fig.1 the displacement of the working face of underground exploitation is simulated by the stages of sliding of a batten in the bottom part of the T-S pile. The contrasting (white) rollers of the T-S pile make possible clear dynamic recording of the freely chosen points of the medium. The research stand creates the possibility of simulation of all the deformation parameters in the near-surface zone as well as in the rock mass.

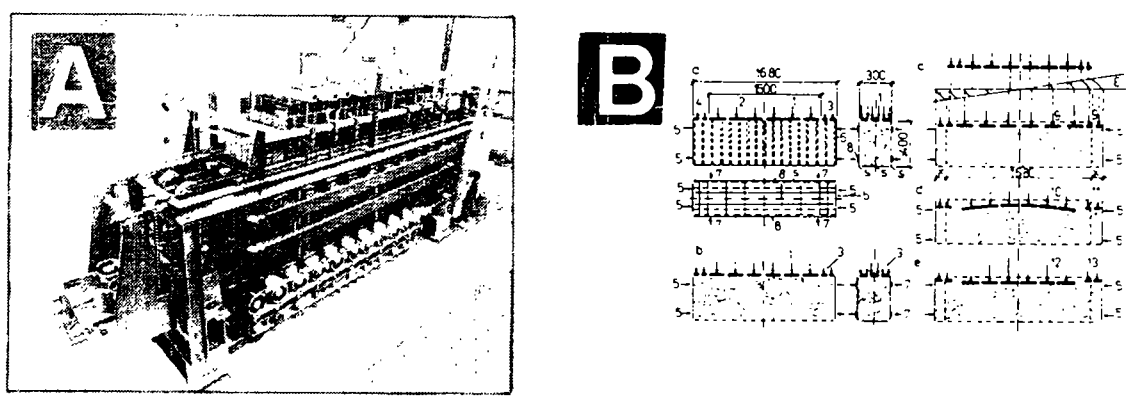


Figure 2. Apparatus for modelling of horizontal mining deformations in loose granular medium
A) General view
B) Principle of apparatus operation taking under consideration an external load.
Denotations: 1-2: vertical load, 3-4: horizontal load, 5-8: head-walls and side-walls, 9: flexible load, 10: stiff load.

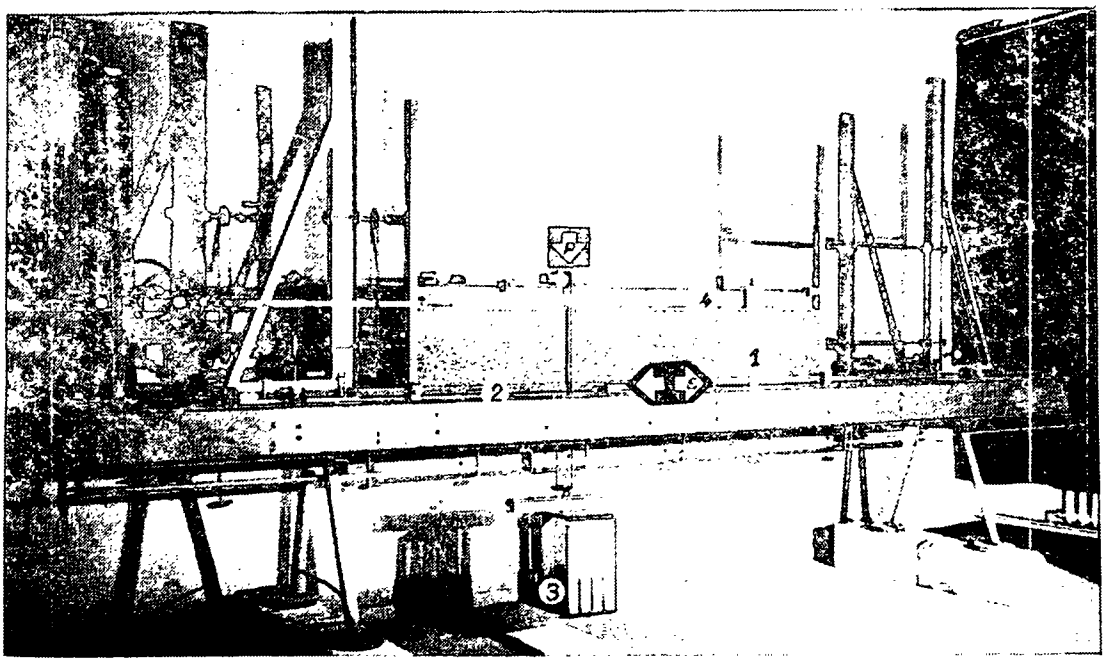


Figure 3. Test rig for modelling of the mining deformations in T-S type analog medium.
Denotations: 1: T-S analog, 2: rubber band, 3: external load, 4: model of the foundation.

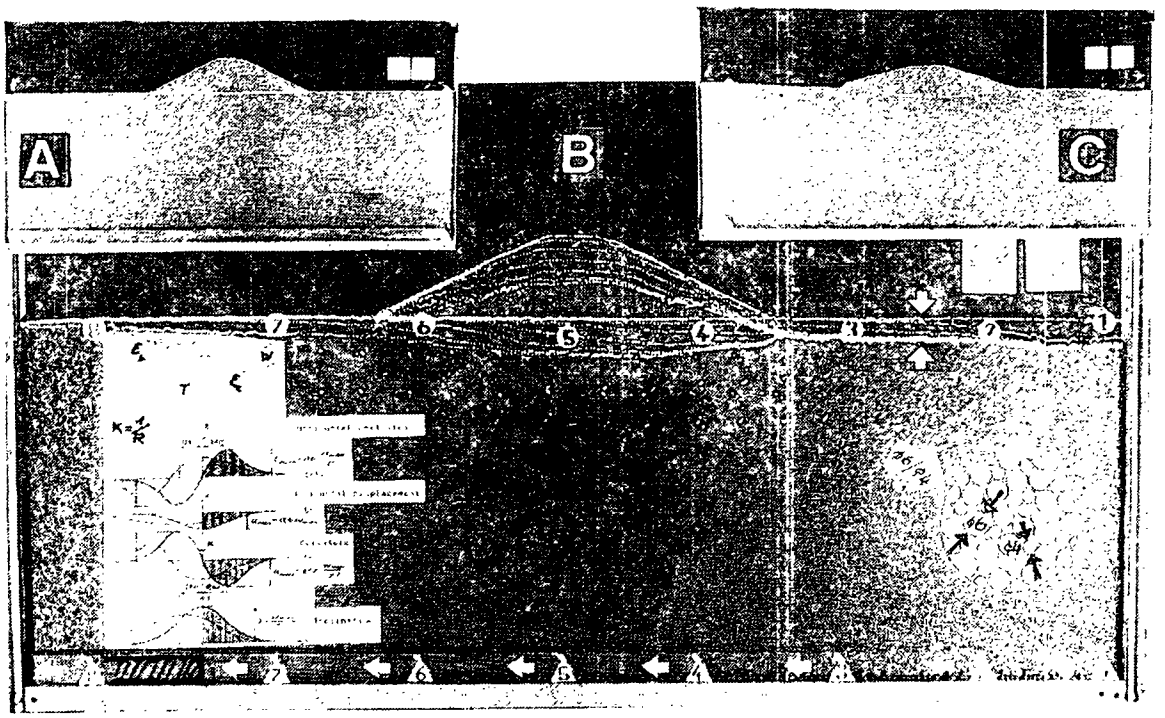


Figure 4. Modelling of the mining deformations within an embankment earth construction
A) Initial state
B) Passing of dynamic subsiding trough under the object, perpendicularly to its longitudinal axis,
C) Final state, after the excitations have ceased.

The research stand shown in Fig.3 makes possible the simulation of the main index of the mining deformation, i.e. the unitary horizontal deformation ε of the compacting and slackening character. The container filled with T-S pile has in its bottom part a rubber batten stiffly attached to the properly bedded side walls. The central drive makes possible exerting a uniform tension on the rubber belt and eventually creates a horizontal simulation of homogeneous deformations ε, typical for a mining area. This apparatus additionally makes possible the simulation of the vertical curvature of the bed K, both concave and convex.

The most important and fully successful tests of application of T-S analog for geotechnic applications in mining-influenced areas include:

- modelling of the deformations arising in mining-influenced areas caused by underground mine exploitation together with the global assessment of the influence of the dynamic subsiding through, on the surface objects such as earth structures (embankments, cross-cuts - Fig. 4), tunnel constructions (Fig. 1), supports, etc.
- estimation of the influence of slackening horizontal deformations of the building foundations in wall footing or grillage systems (Fig. 5a), together with the assessment of the influence subgrade reinforcement on the reduction of additional axial forces in the wall footing - Fig. 5b.
- assessment of the influence of compacting horizontal deformations on the surface retaining constructions - Fig. 6a, b, and the constructions hollowed in the ground - Fig. 6c.
- the determination of the character of interrelation of the road surface and the permanent way with the mining-influenced area; a detailed investigation was carried out to find the sources of additional forces in the rails of the permanent way resulting from thermal effects and braking vehicles - Fig.7. The studies were also aimed at finding the sources of slackening deformations (Fig. 8) of the mining-influenced subgrade and determining the reinforcement of load capacity of track grillage subgrade by geotextile undersleeper containers and horizontal membrane - Fig. 9.

The detailed analysis of the results of this research is impossible to be presented here, because of the space limitations of this paper. More details concerning this subject may be found in [3,4,5,6,11] and other works.

The results of the kinematic research show the essential differentiation of the character of interrelation of structures with the mining-influenced areas opposed to the case of a non-mining area. As a consequence a necessity arises of a separate theoretical approach to this class of complex, interdisciplinary computational problems encompassing the issues of rock-mass mechanics, mining geotechnic and the construction theory. Without applying the described research techniques, it would be extremely difficult, if not impossible, to obtain the described results.

CONCLUSIONS

The application of the Taylor-Schneebeli type analog ground in a model research of mining substrate proved a number of essential advantages in comparison with the traditional models using equivalent materials and natural granular substances. The advantages are:

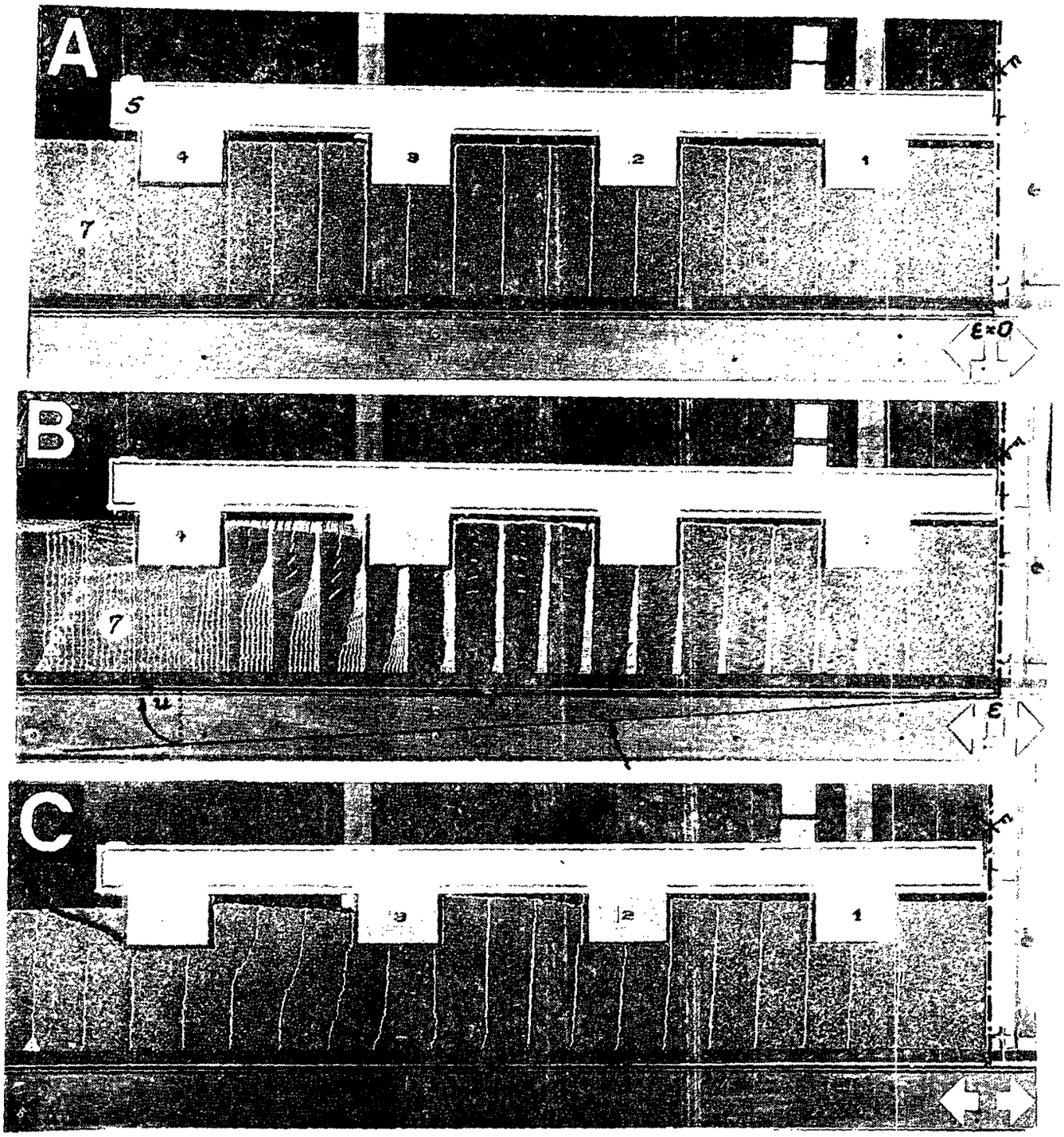


Figure 5. Modelling of the interrelation of the transversal foundation benches under the conditions of horizontal slackening of subgrade without reinforcement
A) Initial state
B) Growth of slackening deformations in mining subgrade
C) Final state, after the excitations have ceased
Denotations: 1-4: transversal benches, 5: longitudinal bench

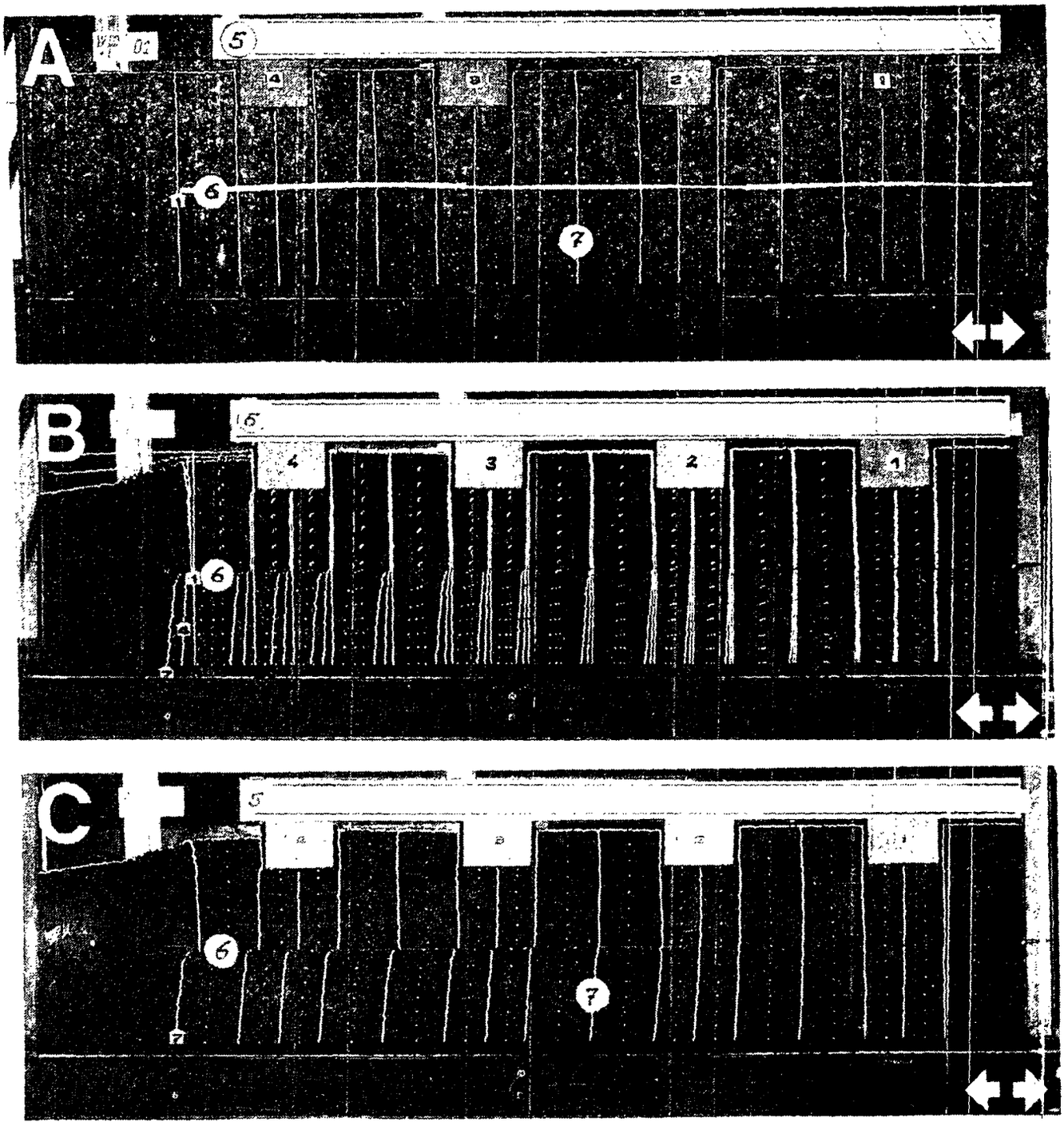
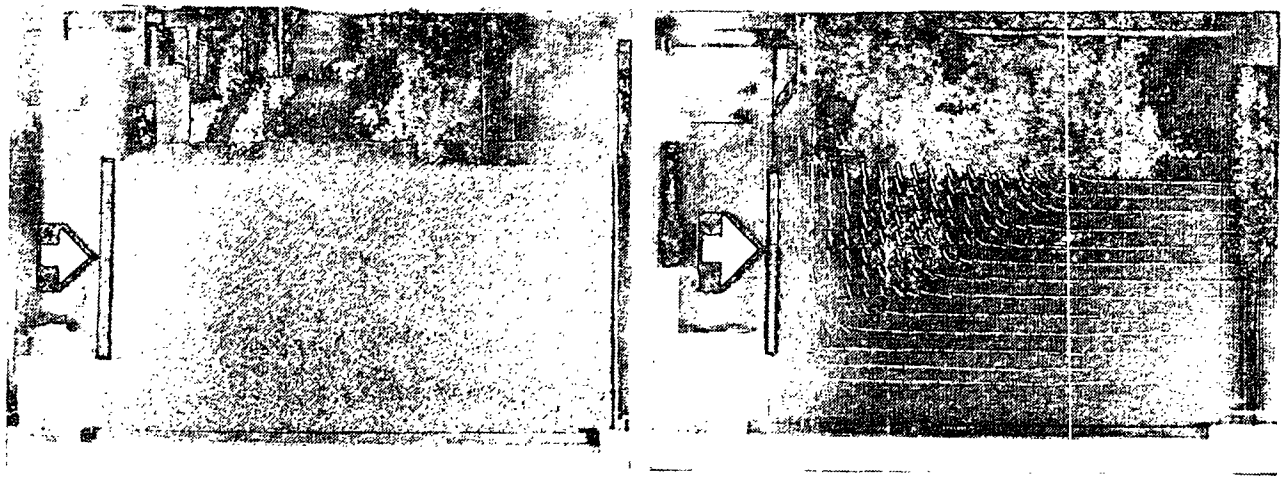


Figure 6. Modelling mining influences on transversal foundation benches under the conditions of slackening deformations of subgrade with geosynthetics strengthening
A) Initial state
B) Growth of slackening deformations in mining subgrade
C) Final state of deformation after the influences ceased
Denotations: 1-4 transversal benches, 5: longitudinal bench, 6: subgrade with geosynthetic strengthening, 7: contrasting elements of T-S pile.



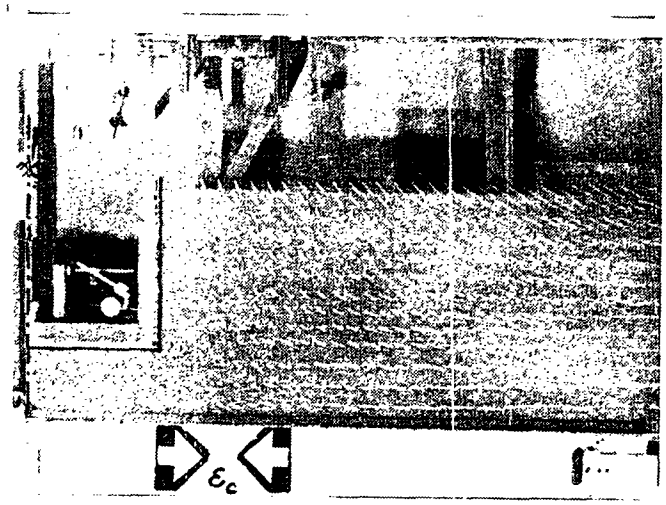


Figure 7. Influence of horizontal thickening deformations on the retaining constructions
A) Passive ground pressure without mining influences (shift of the construction towards the ground-left side, relative shift of the ground in relation to the construction-right side)
B) Pressure of the ground under the conditions of thickening deformations of mining subgrade with kinematic analysis (a-horizontal component, b-vertical component, c-resultant)

- comparatively simple construction of test stands of universal use, allowing the recording of the deformation field and the components of the stress state in complex flat state of deformation, recording being complete and undisturbed by the boundary conditions
- preservation of the main criteria of modelling similarity, creating the basis for the further qualitative and quantitative analyses. The adoption of the obtained results to the engineering practice is possible thanks the values of the scale coefficient α_i being close to those corresponding to the natural conditions.
- relatively short duration of the research and test programme, low cost of labour and apparatus and the consistently good repeatability of the results
- the characteristics of the phenomena of the dislocations and deformations in the rock mass and the sub-surface layer fully correspond to the qualitative representation of the same effects in the experimental research in T-S type analog medium. The latter statement concerns also the process of the interrelation with the surface objects located on the mining-influenced areas, also under different conditions of mining, for instance in case of strip-mining;
- the results of the research on the analog medium have mainly introductory and qualitative character. Therefore, the attempts to transfer directly the qualitative criteria to any real object should be considered with due care and attention.

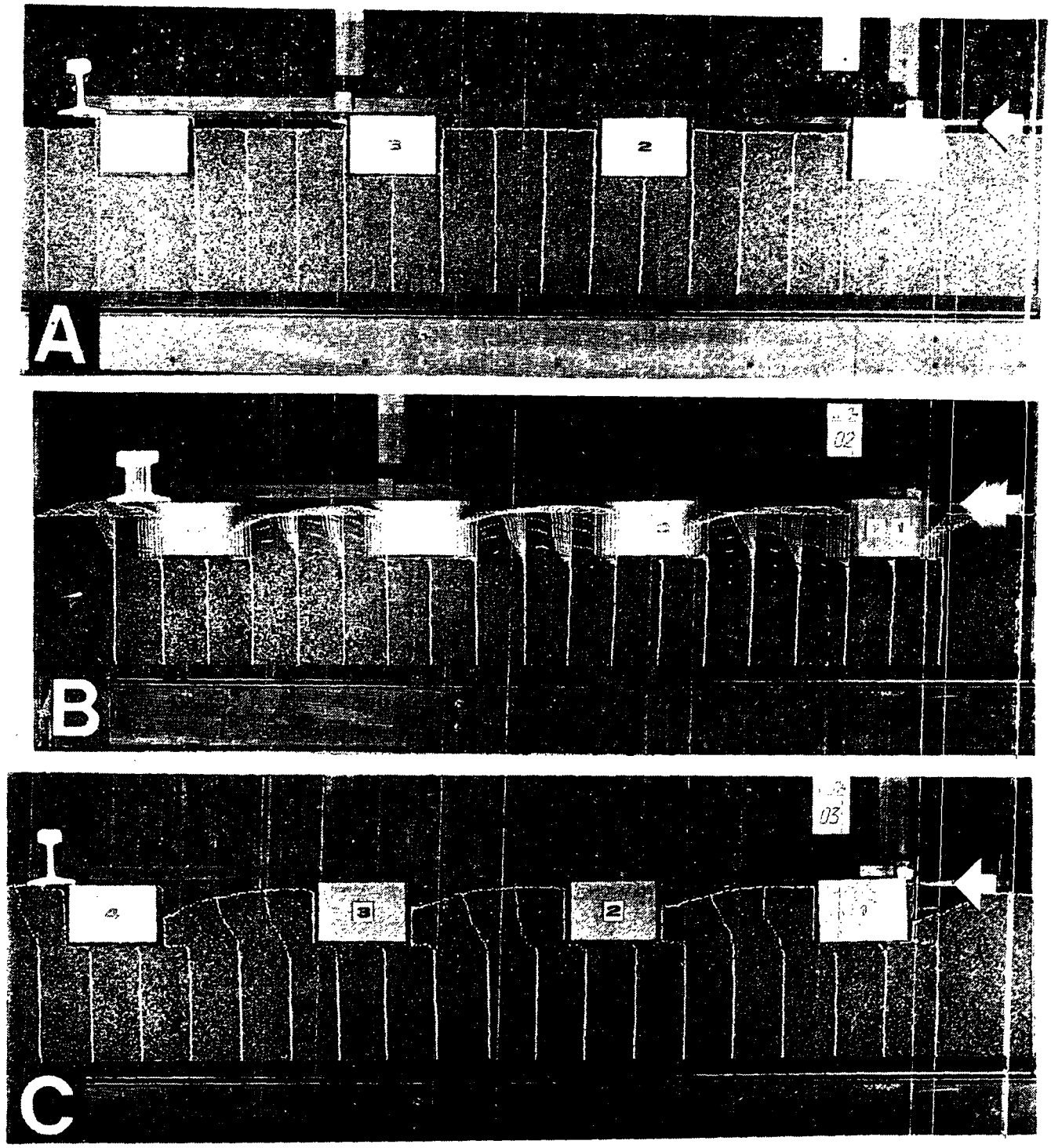


Figure 8. Modelling of kinematic phenomena related to the rise of axial forces in railroad track on the non-mining areas.
A-Initial state
B-Growth of the axial forces in permanent way (temperature, retarding of vehicles etc.)
C-Final displacement state

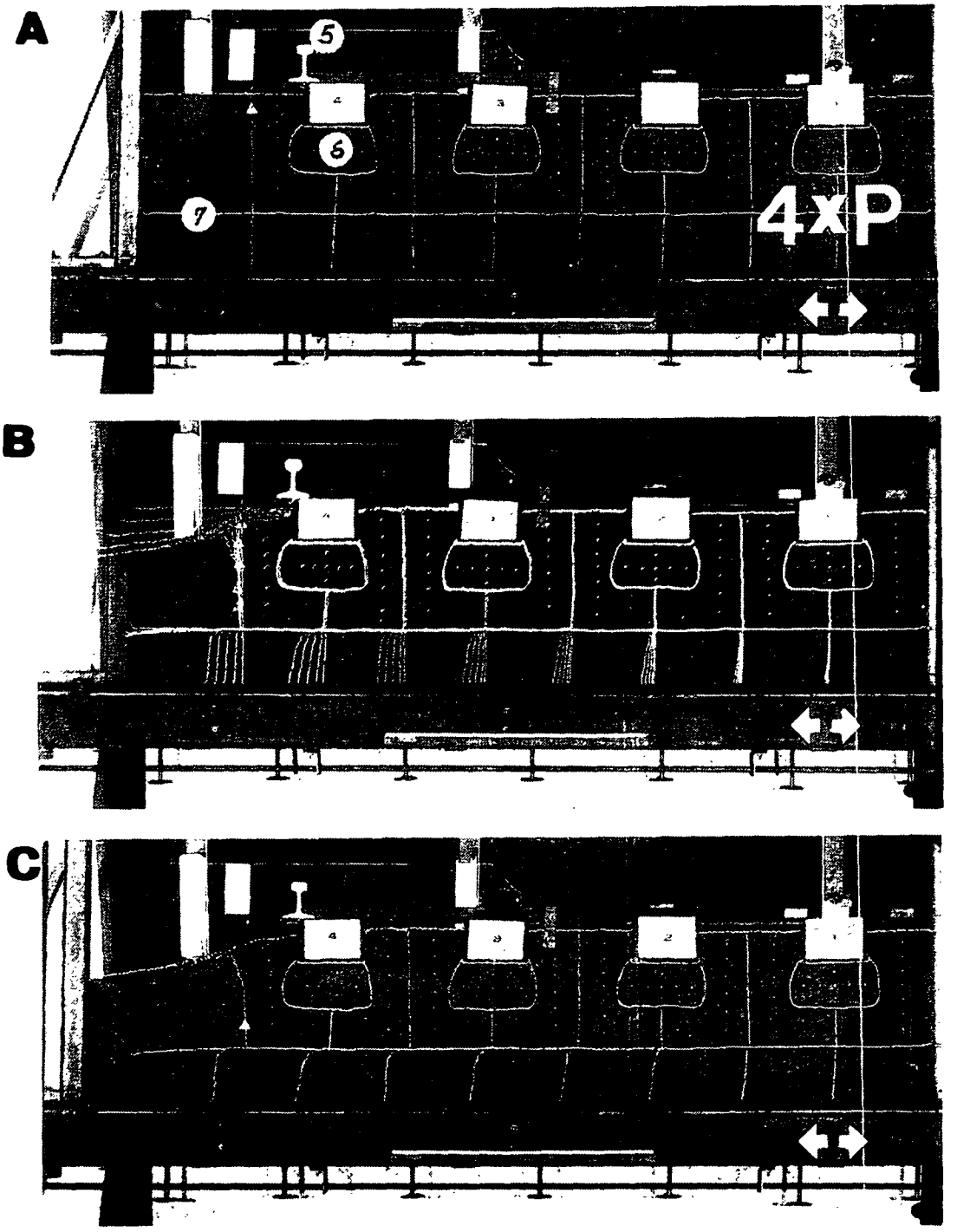


Figure 9. Modelling of the influence of slackening mining deformations on the permanent way strengthened with undersleeper geotextile containers and horizontal membrane.
A) Initial state
B) Growth of slackening deformations in mining subgrade
C) Final deformation state, after the influences ceased
Denotations: 1-4: sleepers, 5: rail, 6: undersleeper geotextile container, 7: horizontal geotextile membrane.

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RELEVELING AND BEHAVIOR OF STRAP RETROFITTED DAMAGED TEST FOUNDATIONS EXPOSED TO MINE SUBSIDENCE[1]

by Gennaro G. Marino, Ph.D., P.E.[2]

Abstract. Test foundation walls were constructed in an area of planned subsidence. These crawl space-sized block bearing walls were located in the tension zone of a longwall panel. The test walls were 12.2 m (40 ft) long and were vertically loaded on top with soil bins to simulate the house weight. As the longwall proceeded past these test foundations subsidence movements damaged the test structures. Using a steel strap retrofit and applying a cementitious surface coating, the foundations were structurally and aesthetically repaired. The repaired test foundations underwent significant subsequent subsidence as an adjacent longwall was mined beneath. The response of the repaired foundation is summarized. The second part of the paper describes releveling another set of the damaged test foundations after the first subsidence event by using an innovative procedure. First the straps were applied to the block bearing walls and then with wall jacks the top of the walls were successfully made level. This releveling procedure is outlined with the results.

Additional Key Words: mine subsidence, foundation response, foundation leveling, foundation repair.

Introduction

In the first part of this paper, a field test of a newly developed technique for retrofitting residential foundation walls damaged by mine subsidence is described and discussed. There were three main objectives of the field test: 1. to evaluate the constructability of the new repair technique, 2. to investigate the performance of the retrofit under subsidence induced bending and climatic conditions, and 3. to relate the aesthetic and structural performance of the retrofit to factors such as subgrade bearing capacity, soil stiffness, angular distortion, and tilt. All results from all three objectives are discussed herein except for the influence of the subgrade characteristics on foundation behavior. Discussion of this latter topic would make this paper too lengthy.

The research described in the second part of this paper involved the field testing of an advanced tilt correction procedure that was proposed for houses tilted from mine subsidence (Marino, 1987). The objective of the field test was to determine: 1. the feasibility, 2. the constructability, 3. releveling capability, and 4. the level of difficulty of the proposed procedure.

Complete data summaries and analyses of the work reported herein can be found in Marino et al, 1992A; Marino et al, 1992 C; and Marino and Gamble, 1993.

Background and Site Conditions

The test site was located at a rural location just outside of West Frankfort, Illinois. At the test site a series of foundation walls were built by the University of Tennessee (Bennett, et al, 1992). Originally, the test walls were built to evaluate the effects of different foundation characteristics and subgrade materials on the structural response of similar foundations to mine subsidence movements.

The test walls were built with ordinary concrete blocks. These walls were not grouted nor reinforced. Block dimensions were 8" x 8" x 16". The

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walls were four courses high and 12.2 meters (40 ft) long. All of the walls ran side by side. Each wall was paired with another and supported soil-filled wooden bins which were used to simulate the load of a house (see Figure 1).

One pair of test walls was used for the retrofit test. They were designated as walls T5 and T6. Although the walls were of the same construction, different footing designs were used. The eastern-most foundation wall, T5, was founded on a footing constructed of normal concrete with a 0.004" plastic sheeting around it in an attempt to reduce friction between it and the sub-grade. Foundation wall T6 was on a footing constructed of concrete with steel fibers in it. Figure 1 shows cross-section diagrams of the two footings.

The soil subgrade was a mottled brownish gray and orangish brown silty clay, CL, with trace to little fine sand. Hand penetrometer readings for the soil were consistently above 2.0 kg/cm^2 (4 tsf), so the silty clay is classified as being hard. Moisture content of the material was between 16 and 20 percent. Plastic indices for the clay ranged from about 24 to 36% with the natural moisture content from 5 to 10% greater than the plastic limit.

The walls were erected near two future longwall mine panels. Figure 2 shows the location of the north-south test walls in relation to the relevant longwall panels. The walls were built during the spring of 1990 before any subsidence took place. During mid-May of 1990 the face of the first longwall panel was beneath the test site. Significant damage in the walls was induced by the subsidence movement over that panel. Figure 3 shows the main cracking in both T5 and T6 as of May, 1991. The walls were repaired using the new strap technique between 5/13/91 and 5/17/91. The face of the second panel was adjacent to the test foundations in early to mid-June 1991.

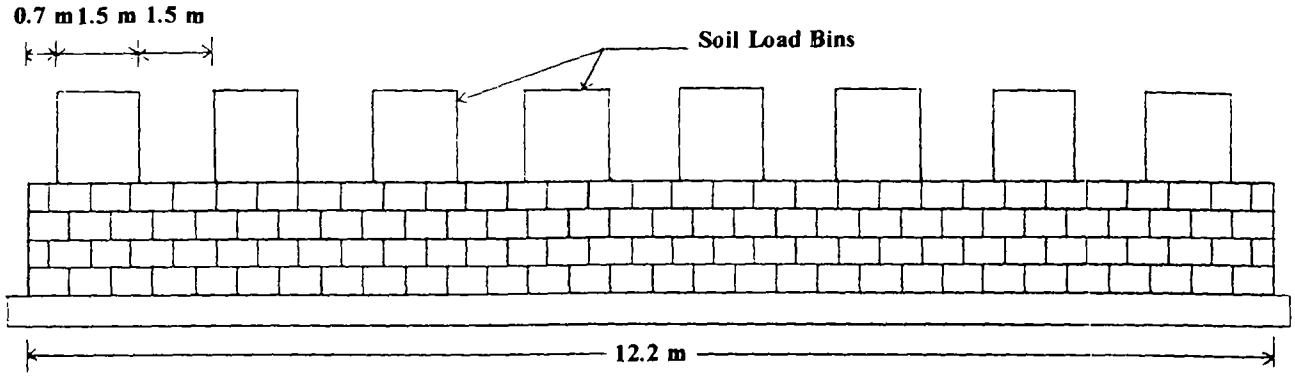
The tilt correction methodology was tested on the middle set of foundation walls (Numbers T3 and T4). These block walls were of the same construction as T5 and T6. The releveling of these walls was performed after the north panel was mined and the foundations were tilted 20-23 cm (8-9 in.) to the south with angular distortions of about 1/200.

Field Test of Foundation Retrofit Procedure

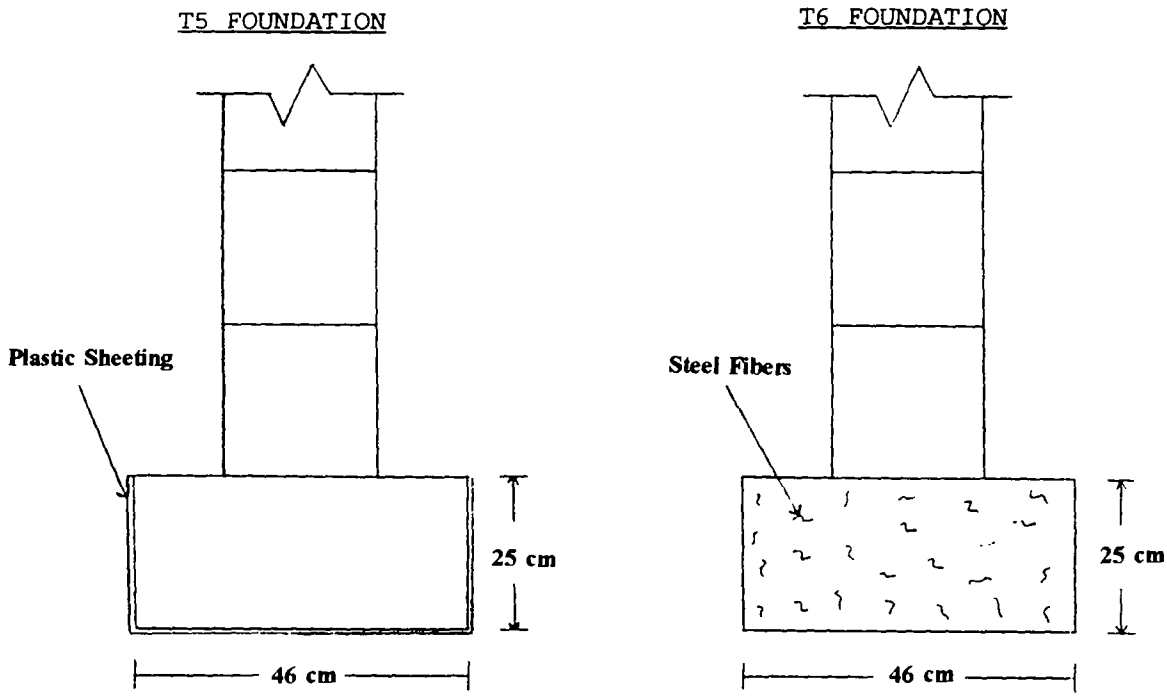
Overview of Wall Retrofit

The strap retrofit system used in the field test is illustrated in Figure 4. For test foundation T5, the strap retrofit was applied to both the east and west sides, and for T6 a strap was applied only to the east side. The total cross-sectional area of the steel was the same, however, for both walls. The purpose of attaching straps to both faces of wall T5, and only one face of wall T6 was to evaluate single strap performance and compare it to the performance of the wall with reinforcing straps on both faces. It was important to evaluate single strap performance, because if the larger single reinforcing straps perform adequately, the necessary reinforcement can be applied entirely to the interior wall faces. This procedure modification would be more cost efficient than applying straps to both sides especially where excavation would be required. Also note as shown on Figure 4, the amount of steel applied to the wall surfaces increased towards the middle of the walls. This increase in steel was calculated based on the increased bending moments estimated from the expected hogging-like subsidence movement resulting from when the southern panel is mined.

There were five main steps taken in retrofitting the damaged test foundations. First, the steel required for the repair was determined. Second, the first stage of repair was done and the walls were cleaned. This included filling open wall cracks in the expected zone of compression with mortar. By doing this large wall deflections due to crack closure were prevented. Shear strength was restored to the walls by grouting reinforcement into the cells of the concrete blocks in areas exhibiting diagonal shear cracks. Cleaning the walls was then completed, thus enabling the epoxy and fiber-cement to adhere to it. Third, the steel reinforcing straps were prepared for attachment. Fourth, the external reinforcing straps were attached to the walls to increase wall bending capacity. Fifth, a surface finish of fiber-cement was applied to the walls to improve the retrofit

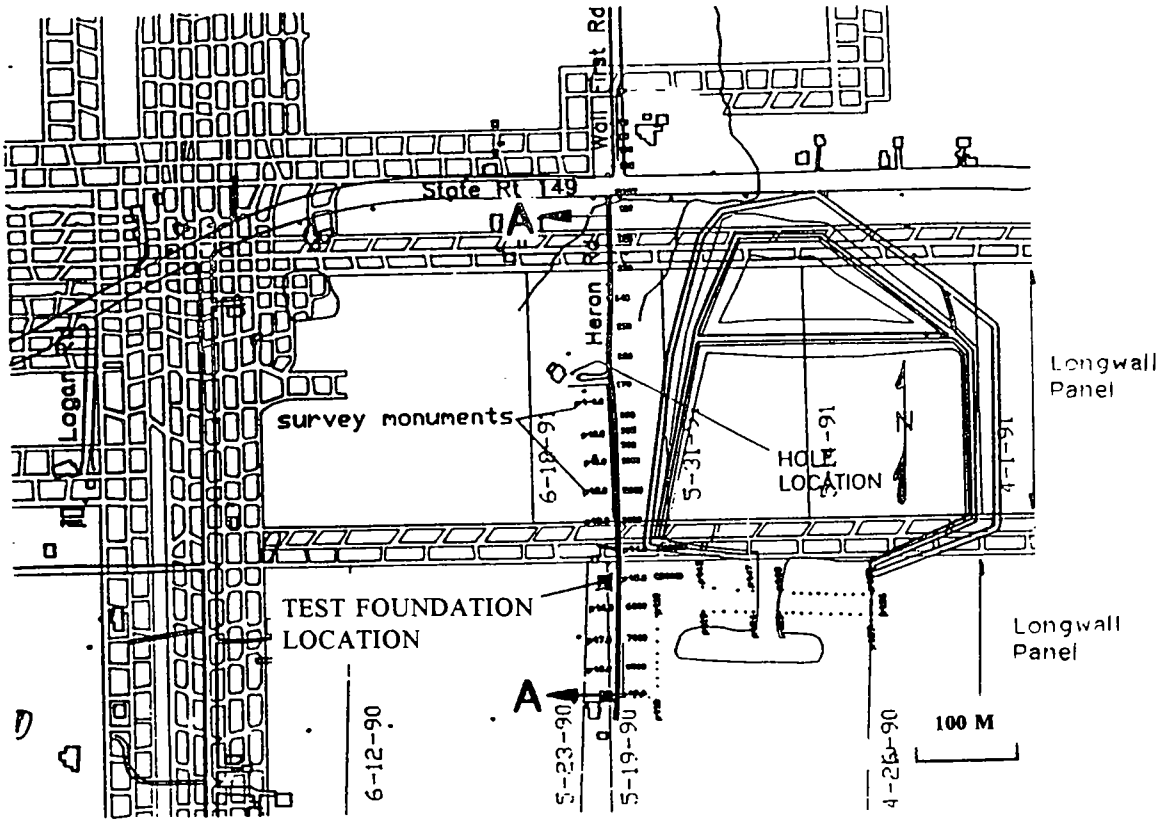


ELEVATION

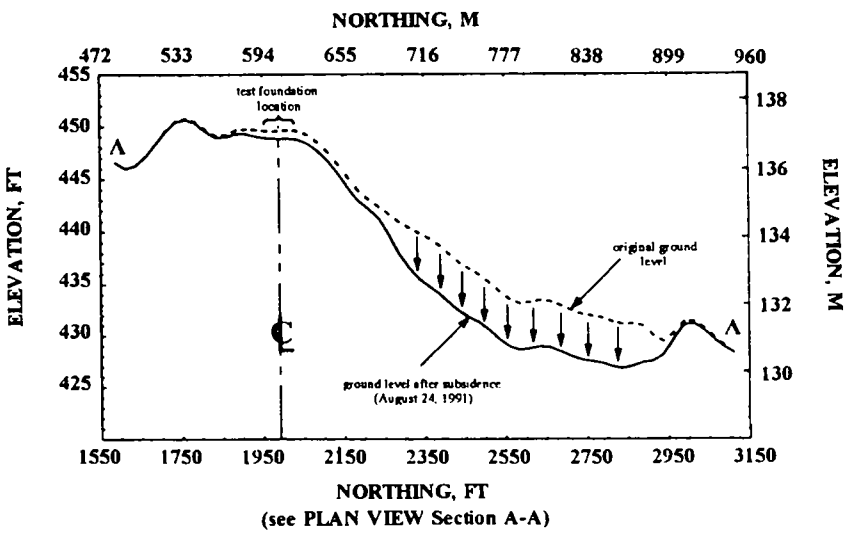


CROSS-SECTION

FIGURE 1 CONFIGURATION OF TEST WALLS



PLAN



PROFILE OF SECTION A–A

FIGURE 2 POSITION OF THE TEST FOUNDATIONS RELATIVE TO MINING ACTIVITY AND SUBSIDENCE PROFILE

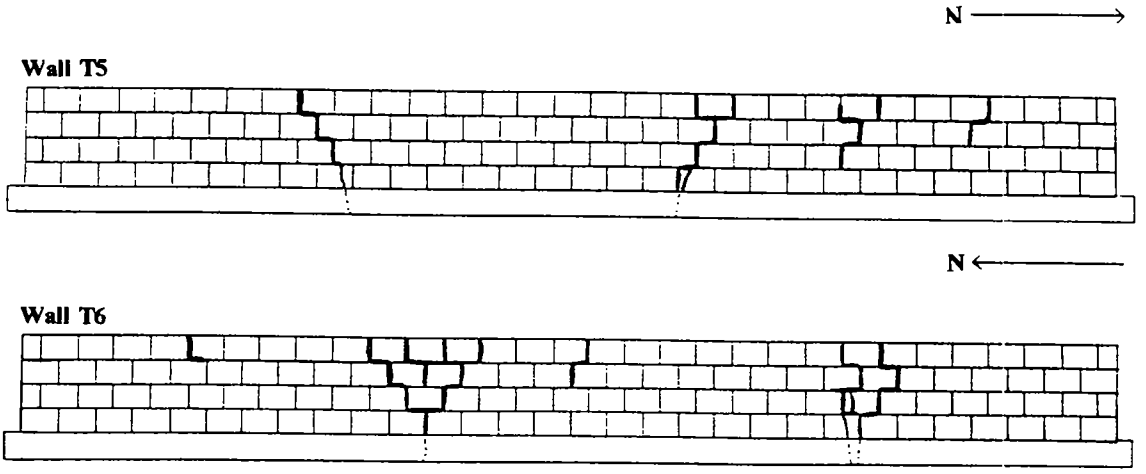


FIGURE 3 MAIN CRACKING IN FOUNDATION T5 AND T6 (MAY, 1991)

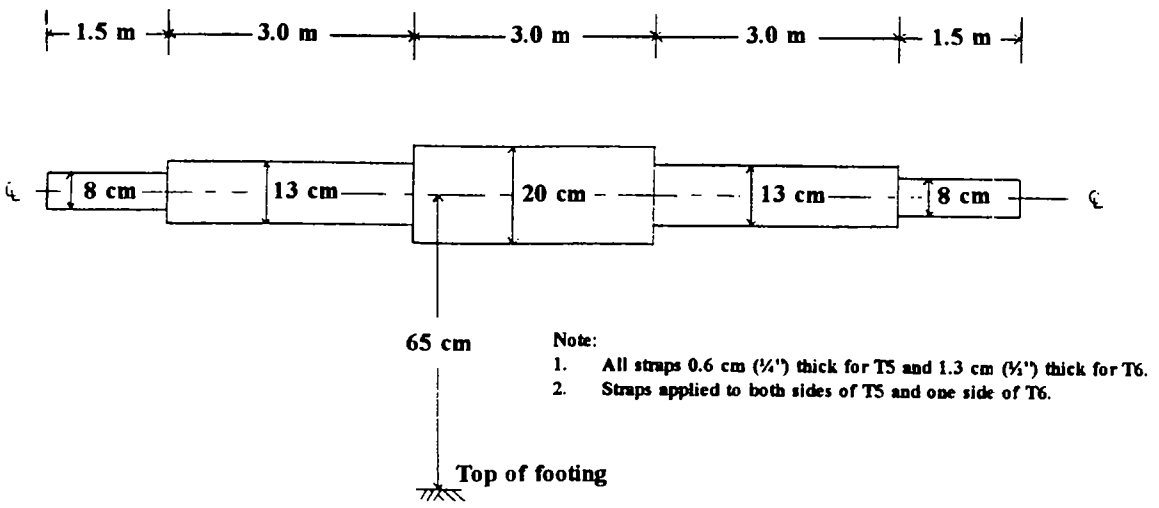


FIGURE 4 STEEL REINFORCING STRAP DIMENSIONS FOR WALLS T5 AND T6

appearance. A detailed description of the application of this strap retrofit process is given in Marino, 1992.

Instrumentation

A set of measurements was obtained periodically for one and one-half years after strap installation. This included: level survey of points on foundation walls, on wall footings, and on soil monuments along the foundations; continuous strain gauge measurements on the foundation walls along the straps and near the base of the retrofitted sides of the foundation walls; and relative vertical and horizontal displacements between the foundation wall and footing.

Consistent measurement procedures were maintained during the monitoring period. Strain measurements along the wall and footing and relative displacements between the same were performed using a Whittemore strain gauge. Details on the instrumentation and measurement specifications are given in Marino et al, 1992A and Marino and Gamble, 1993.

Response of the Retrofitted Foundation

Foundation Settlement

Based on the survey data taken during subsidence, the wall and footing settlement profiles for each retrofitted foundation are similar (even though steel was placed on only one side of T6 compared to both sides of T5). Furthermore, these profiles show the foundation walls and footings undergoing a more definitive hogging curvature with time. To demonstrate these points, foundation settlement measurements for T5 relative to the positions on 14 May 1991 are shown in Figures 5 and 6. Note only selective profiles are shown for clarity.

Using the ground and foundation profiles, the tilt and average angular distortion were calculated for three dates since the repairs were made. These calculated angular tilt and distortion values are compared in Tables 1 and 2 respectively. As can be seen in Table 1, the walls and footings for both foundations T5 and T6 have essentially tilted (or rotated) with the ground, reaching values of 0.48%. On the other hand, the angular distortion experienced by the repaired walls was typically less than that experienced by the footing and about one-half that of the soil at the severest levels (1/1200). Also, it is interesting to note that Wall T6, which has been strap retrofitted only on one side, appears to have bent less than Wall T5 with straps attached to both sides (i.e. 1/3,125 for T6 compared to 1/2,174 for T5).

Foundation Straining

Particle displacement diagrams along the straps and near the bottom of the retrofitted sides of foundation T5 and T6 were prepared and all showed similar trends. As examples of this data the particle displacement diagrams for the east face of foundation T5 at strap level and near the base of the foundation wall are depicted in Figure 7. The particle displacements were determined by adding the displacements measured between each set of Whittemore strain points starting from the north end of the foundation. Therefore, each point plotted, such as on Figure 7, shows the total of all point to point displacements measured south of Whittemore point in question.

As can be seen in Figure 7 little overall change in strain (as determined by the slope of the line) occurred between November 26, 1991 to November 29, 1992. Changes in the average longitudinal strain over this period of time ranged from about -2.0×10^{-5} to 3.0×10^{-5}. Even peak average strains during the entire test period were small and were 2.4×10^{-4} in tension and -4.8×10^{-5} in compression. Even though these strain values may appear low compared to the increase in the angular distortion (or deflection) of the walls (see Table 2), they are actually greater than values that would be predicted by the classical curvature-bending strain relationship (Popov, 1968) which is:

$$c = \epsilon / y$$

where: c = beam curvature
ϵ = longitudinal bending strain
y = vertical distance from neutral axis (assumed at bottom of footing)

The final strains measured in the wall are all tension, which is compatible with the wall being in the

tension zone of the second subsidence event. However, the tensile strains at the bottom of the wall were larger than at the steel straps, 26 in. above the bottom of the wall on all three sides measured. This strain distribution corresponds to a case of sagging curvature in tension, while the measured vertical movements of the walls clearly indicate small hogging curvatures. The reasons for this inconsistency are not clear. A possibility for this inconsistency may be measurement error as the level of straining was low.

No comparison of foundation strain could be made with ground strain based on the soil monuments as these measurements were found to be not accurate.

Overall Performance

The November 1992 readings indicate the amount of ground distortion and tilt experienced by Foundations T5 and T6 since the repair was sufficient to cause damage to conventional residential construction. The most recent ground movements (shown in Tables 1 and 2) have been plotted on Figure 8. In Figure 8 are damage criteria for sag subsidence for houses resting on conventional block crawl-space foundations. This damage relationship assumes that ground distortion is empirically related to ground horizontal strain, i.e., as a house is exposed to some angular distortion level it is also exposed to a certain amount of ground strain. Therefore, ground strain can be considered to be inherently included in the damage plot shown in Figure 8.

As can be seen in Figure 8 a house exposed to the level of ground movement by the test foundations will likely sustain 10 to 40 percent Relative Repair Costs (RRC). RRC is defined as the percentage the necessary repair cost relative to the house replacement cost. This range of RRC values will usually include minor damage to the foundation and superstructure (see Table 3). At the most, possible replacement of accessory foundation element(s) or partial replacement of the brick veneer would be necessary. As noted above, the bending of the walls was about half that of the ground. This level of distortion reduces the estimated damage to Level I (see Figure 8). In other words, no damage to only RRC of 10 percent would be expected. Clearly then, based on this performance, these retrofitted test foundations are performing better than conventional construction since essentially only one minor discontinuous crack has appeared.

This crack is located about 9.40 m (370 in.) from the north end of Wall T6. The width of this crack has increased slightly over the last year of monitoring. This crack was more serious than the shrinkage cracks aesthetically, but is discontinuous and could probably be adequately covered with elastomeric paint if it were a problem. It should be noted, however, that this crack may close if the overall hogging curvatures in the wall become great enough with time as the crack originates at the base of the wall.

Aesthetic performance can be defined as the ability of the fiber-cement (Quikwall) coating to maintain an aesthetic appeal. Deterioration of the aesthetics of the surface cement is considered to occur when the propagation of cracks are visible to the naked eye at a distance of a few feet. The cracks may be load induced or the result of temperature, application, and curing.

Shrinkage cracks form when the fiber-cement coat contracts from drying or lower temperatures. Also, crazing or map-type cracks can occur depending upon the way the cement is applied. This type of cracking was found on all three retrofit faces and has not changed with time. These cracks could only be found through close observation. Therefore, none of the shrinkage cracking that occurred was considered to be an aesthetic failure. Figure 9 contains a photograph of the face of the fiber-cement on November 29, 1992.

Field Test of Tilt Correction Procedure

General Procedure for Releveling

For a tilted and damaged foundation in the tension zone, straps are attached with epoxy to the upper part of the foundation wall (see Figure 10). Before straps are installed, however, any necessary treatment to existing cracks or shear reinforcement to this upper wall section must be completed. Jacks are then installed in windows made in the wall below the

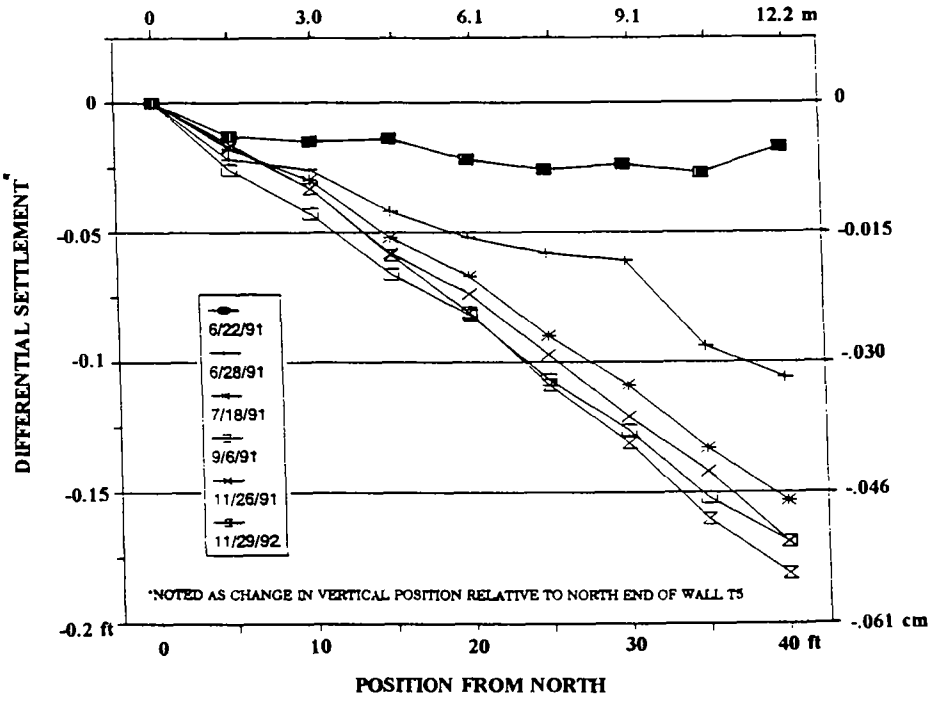


FIGURE 5 DIFFERENTIAL SETTLEMENT OF RETROFITTED WALL OF FOUNDATION T5

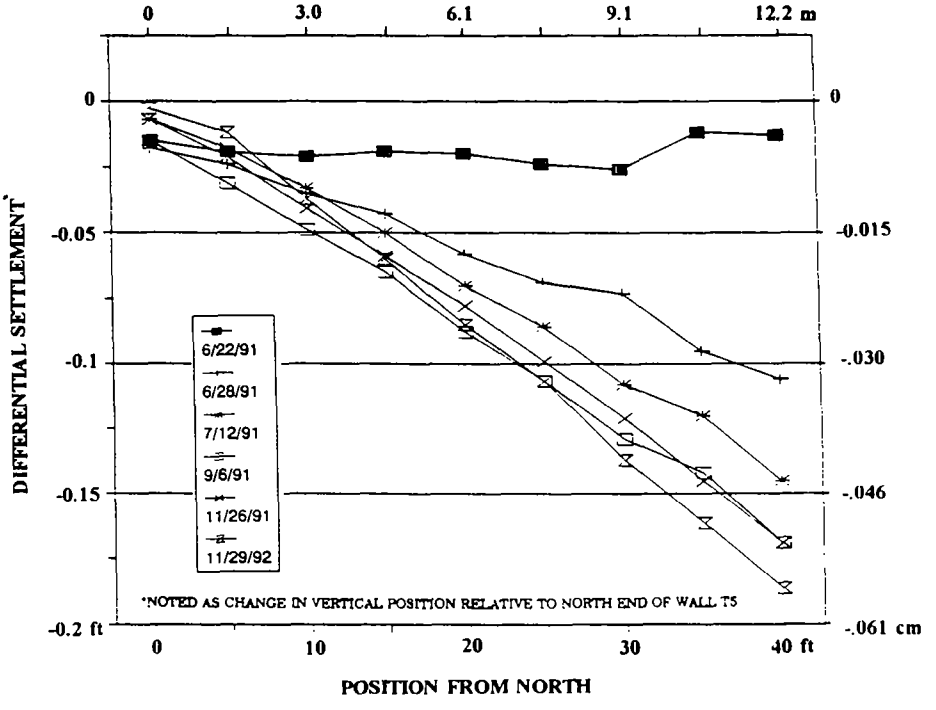
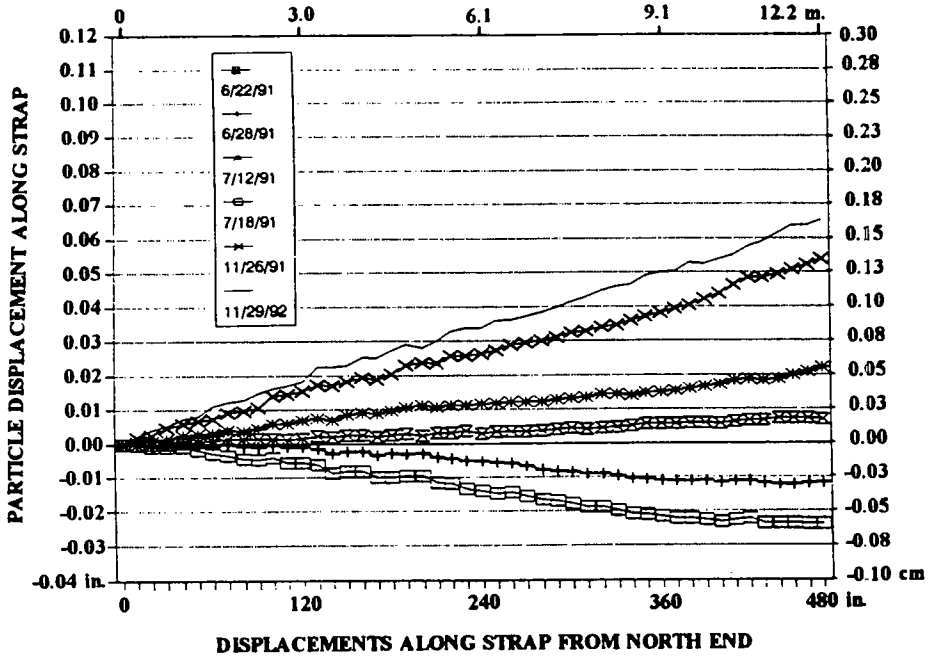


FIGURE 6 DIFFERENTIAL SETTLEMENT OF FOOTING OF FOUNDATION T5



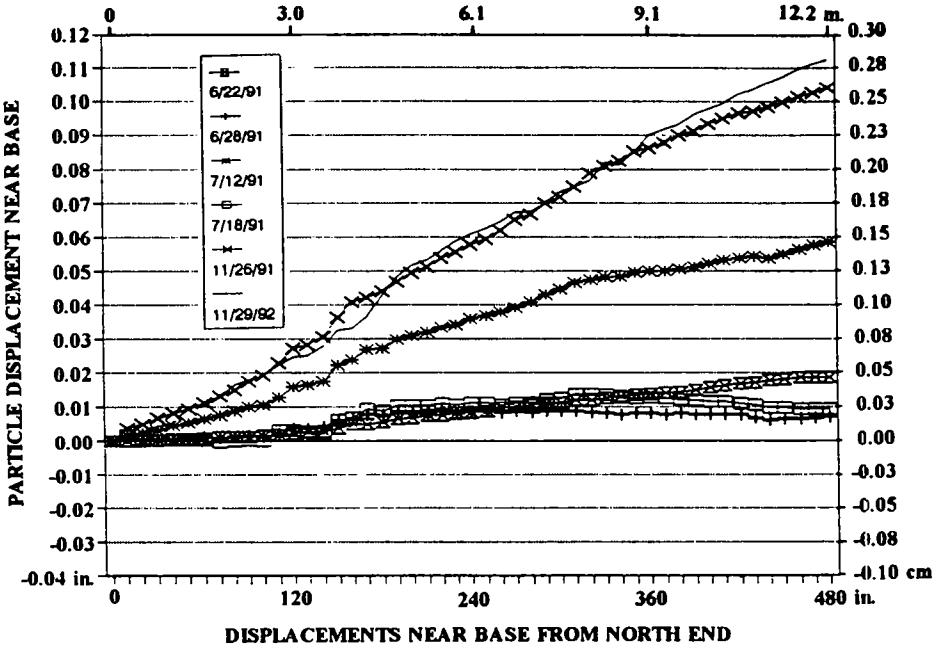


FIGURE 7 RELATIVE PARTICLE DISPLACEMENTS ALONG EAST FACE OF RETROFITTED WALL OF FOUNDATION T5

TABLE 1 TILT COMPARISON*

T5	7-12-91	11-26-91	11-29-92
Retrofitted Wall	0.37%	0.43%	0.45%
Footing	0.35%	0.40%	0.46%
T6			
Retrofitted Wall	0.36%	0.47%	0.48%
Footing	0.36%	0.43%	0.46%
Soil	0.39%	0.44%	0.48%

* Average tilt computed over entire length of profile since 5-30-91

TABLE 2 ANGULAR DISTORTION COMPARISON*

T5	7-12-91	11-26-91	11-29-92
Retrofitted Wall	roughly straight	1/2,500	1/2,174
Footing	straight	1/2,206	1/6,250**
T6			
Retrofitted Wall	straight	1/6,250	1/3,125
Footing	1/1,667	1/2,500**	1/1,500
Soil	1/1,667	1/1,282	1/1,210

* Measurement taken over entire soil monument profile (i.e., middle 30 ft), and includes angular distortion since 5-30-91.

** Possibly low value due to an erratic point.

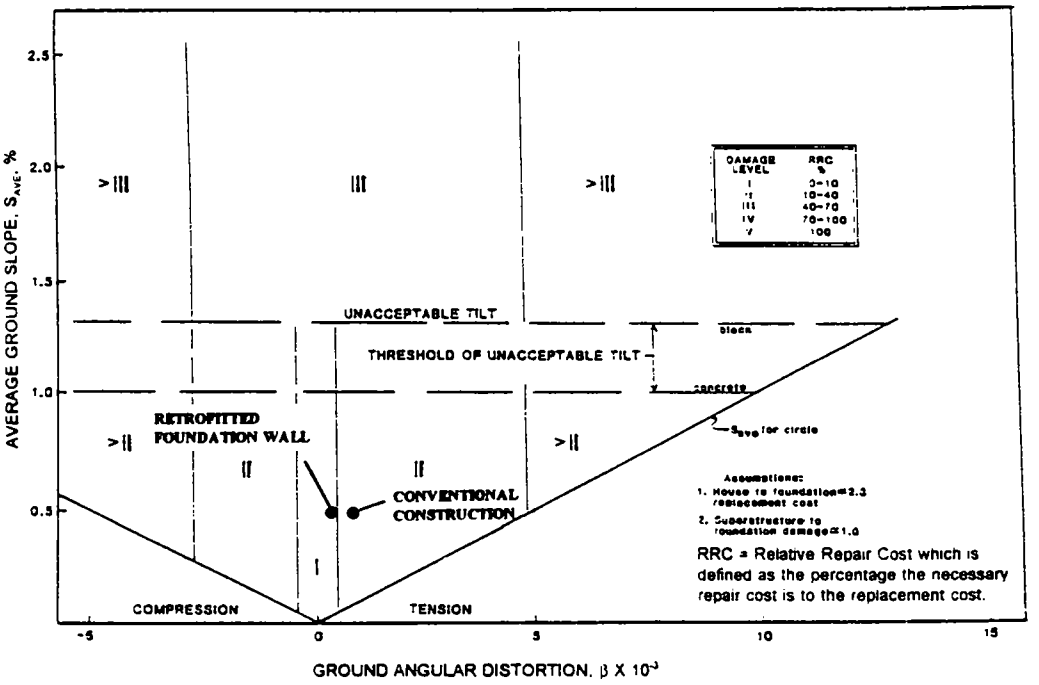


FIGURE 8 DAMAGE LEVELS RELATED TO THE ASSOCIATED ANGULAR DISTORTION AND AVERAGE SLOPE OF THE SAG PROFILES FOR HOUSES ON CRAWL-SPACE FOUNDATION (Marino and Mahar, 1985)

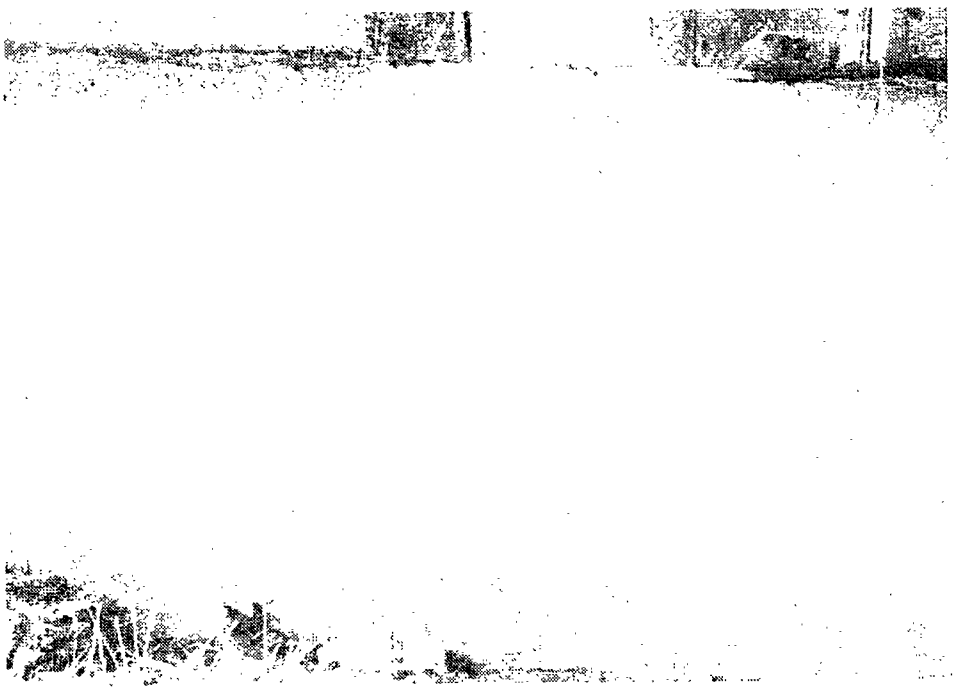


FIGURE 9 PHOTOGRAPH OF FACE OF FIBER-CEMENT COATING ON NOVEMBER 29, 1992

TABLE 3 COMMON SUBSIDENCE REPAIRS FOR HOMES WITH CRAWL SPACES AT DIFFERENT LEVELS OF DAMAGE (Marino and Mahar, 1985)

DAMAGE LEVEL	RRC	COMMON REPAIRS FOR CLASS B HOMES†		
		FOUNDATION	EXTERIOR	INTERIOR
I	≲ 0.1	Patching of minor wall cracks.	Tuck pointing of cracks in brickwork. Possible trimwork of siding/paneling. Repair of molding, and caulking separations. Possible repair of spout-roof separations.	Patching of minor wall cracks and painting of the dry or plaster walls in typically less than half the inside. Racked openings.
II	0.1 - 0.2	Same, possible replacement of accessory foundation element(s) (e.g.; displaced/cracked isolated block crawl-space section, slab foundation, exterior concrete, etc.)	Same.	Same, but with generally larger cracks and possibly paint entire inside. Repair caulk separations around fixtures and openings.
	0.2 - 0.4	Same.	Same, with possibly some replacement of brick.	Same.
III*	0.4 - 0.55	Same, replacement of crawl-space or crawl-space with a small basement section. House jacking necessary. Redecoration of living areas on a slab.	Same, but also partial replacement of siding, replacement of masonry veneer panel (less than one side and one story).	Same.
	0.55 - 0.7	Same.	Same, but replacement of 1 story brick veneer.	Same.
IV*	0.7 - 1.0	Same.	Same, replacement of exterior on wall or roofing. Repair of cracking in chimney-fireplace.	Same.
V*	1.0	REMOVE AND REPLACE HOUSE.		

† A class B home has a masonry crawl-space foundation, with possibly a small basement section. The house is wood-framed with brick, paneled, and sided exterior(s) and can have an attached garage.

The term "same" refers to damage conditions which are present at a lower level and have not been superseded by a worse condition.

* Possible abatement measures necessary.

strapped portion. The jacks are used to lift and level the upper part of the foundation and house. The jacking either occurs directly below a common mortar joint if the foundation is block, or a cut is made parallel and below the strap if the wall is concrete. During jacking a gap in the wall is created and is extensively and continually shimmed to prevent the occurrence of dangerous out-of-plane eccentricities, and to protect against the effects of loss of support due to hydraulic jack leaks.

Once the house and the upper strapped section are level the gap created in the wall can be filled with mortar or concrete and any treatment required for existing cracks can be taken care of. After the foundation walls are completely retrofitted and the house is level, the walls can be covered with fiber-cement for mainly aesthetic purposes. Of course, interior beams and/or walls are also appropriately lifted during the jacking process.

Wall Preparation for Jacking

Structural preparation similar to that conducted on Foundation T5/T6 was performed on T3/T4. (Note here, however, vertical steel straps were used in lieu of grouted-in rebar for shear strength repair. This was the first attempt at using vertical straps on the walls for shear strength repair.) Windows for the jacks were knocked out in the lower course of block at the prescribed places along the walls (see Figure 10). Bottle jacks were then inserted in the windows (see Figure 11).

Because the test foundations to be lifted are essentially two-dimensional structures interior bracing was felt necessary to provide some lateral support. The wood bracing system was designed to allow the jacked part of the foundation to move only in a vertical plane.

Instrumentation

In order to evaluate the amount of lift imposed on the strapped wall sections and soil bins, and to assess the induced strains in the steel straps, survey control points and Whittemore gauge points were established before any jacking was done. The survey control points were set on the top of the wall and were spaced about every 1.5 m (5 ft). Whittemore points were also located on the straps where maximum tensile strains were expected during the jacking process.

Jacking Process and Performance

The jacking process was conducted in three stages: 1. uniformly lifting walls 2.5 cm (1 in.); 2. rotating walls to level positions; and 3. adjusting jacks to straighten the top of the walls (see Figures 11 to 13). The main purpose of Stage 1 was to provide sufficient separation as to allow the north ends of the walls to rotate as the lower south ends were jacked upwards.

It was found during the Stage 2 of the jacking, the test foundation could be lifted 2.3 cm (0.9 in.) without causing excessive strain in the steel (a maximum strain of 0.00022 was measured at 2.3 cm jack displacement). Therefore, 2.3 cm was used as the maximum lift criterion. During Stages 2 and 3 the lifting was monitored mostly by measuring induced separation of the mortar joints with only occasional level surveys. Also when jacking, the surveying effort was lessened because negligible foundation settlement seemed to occur during jacking. Once Stage 2 was thought complete a survey was taken. Jacks were adjusted accordingly to bring the foundation walls level end-to-end. In order to bring the walls level they had to be lifted as much as 20-23 cm (8-9 in.) at the south end.

Although the walls were level end-to-end it was discovered that the middle of the walls were about 3.8 cm (1.5 in.) higher. This was determined by stretching strings from one end to the other at the tops of the walls. The tops of the walls were then straightened out by adjusting the jacks until both strings were coincident with the tops of the walls (see Figure 13). With the walls now straight the jacking was complete. The induced mortar joint separations were supported with additional shims and the jacks were removed.

After all the jacking was complete and the walls were straight and level, the most stressed point measured in the steel (based on Whittemore gauge measurements) reached only about 20 percent of its yield strength.

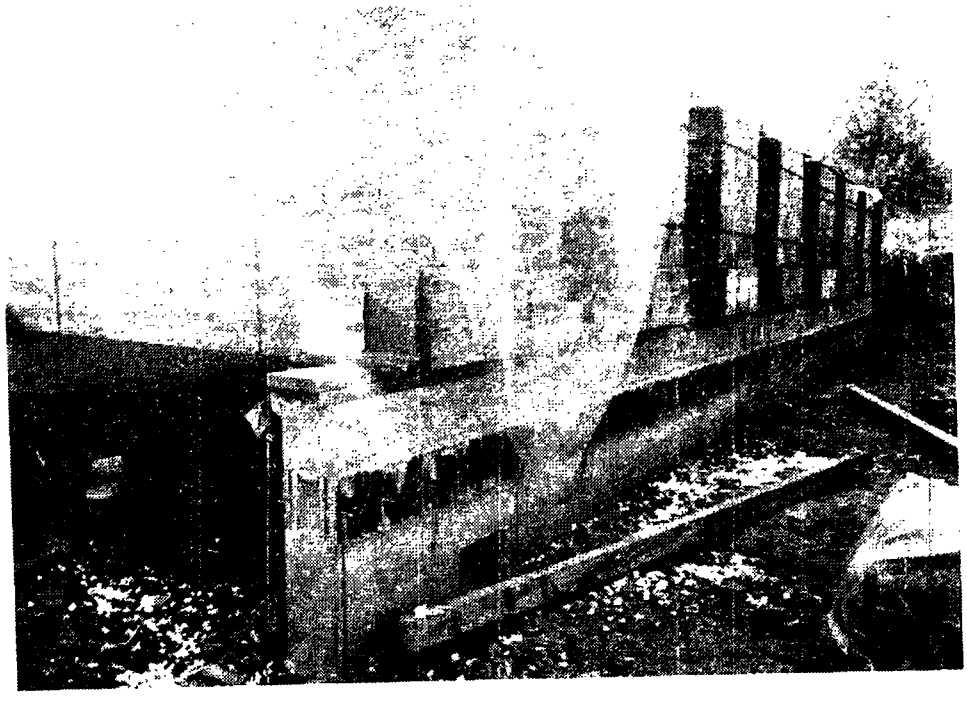


FIGURE 10 ALL JACK WINDOWS KNOCKED OUT

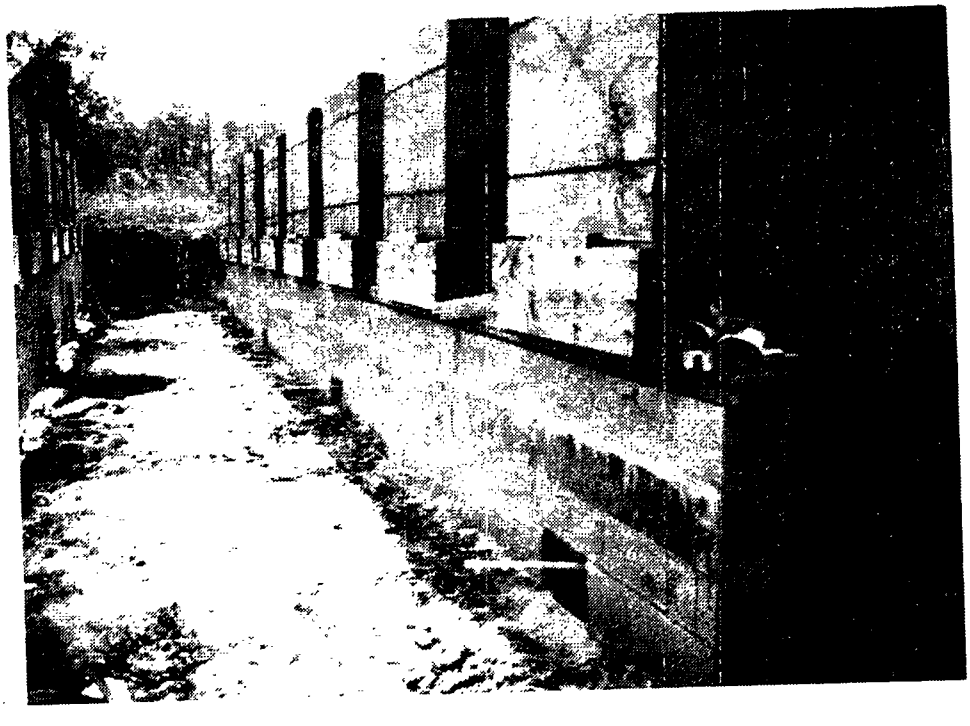


FIGURE 11 JACKS INSERTED IN WINDOWS

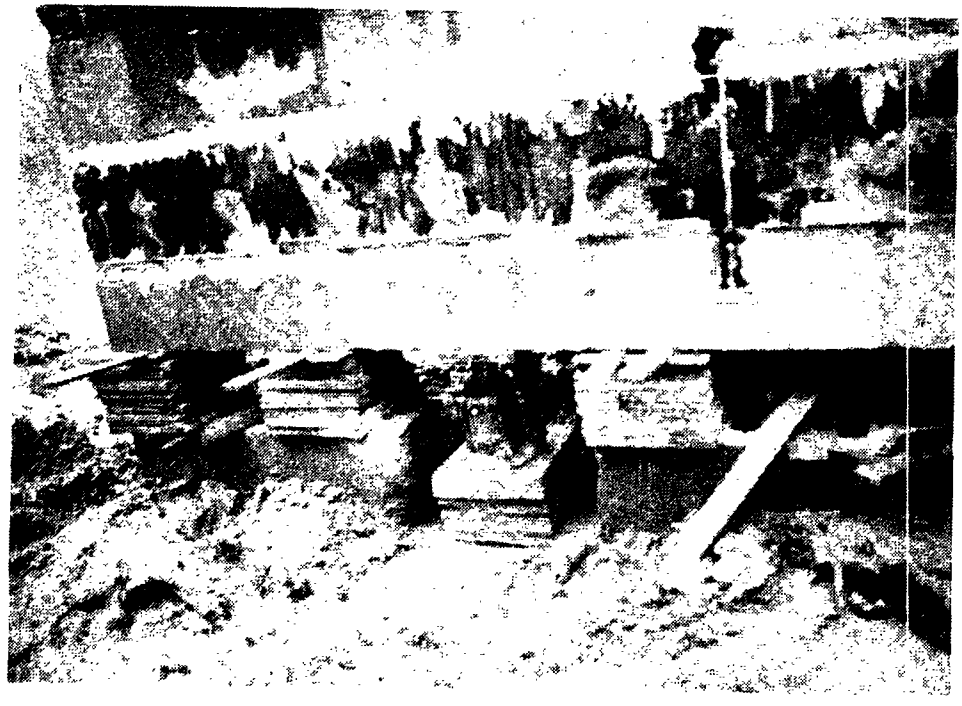


FIGURE 12 JACK ELEVATED IN WINDOW BY THE USE OF PLYWOOD PLATES

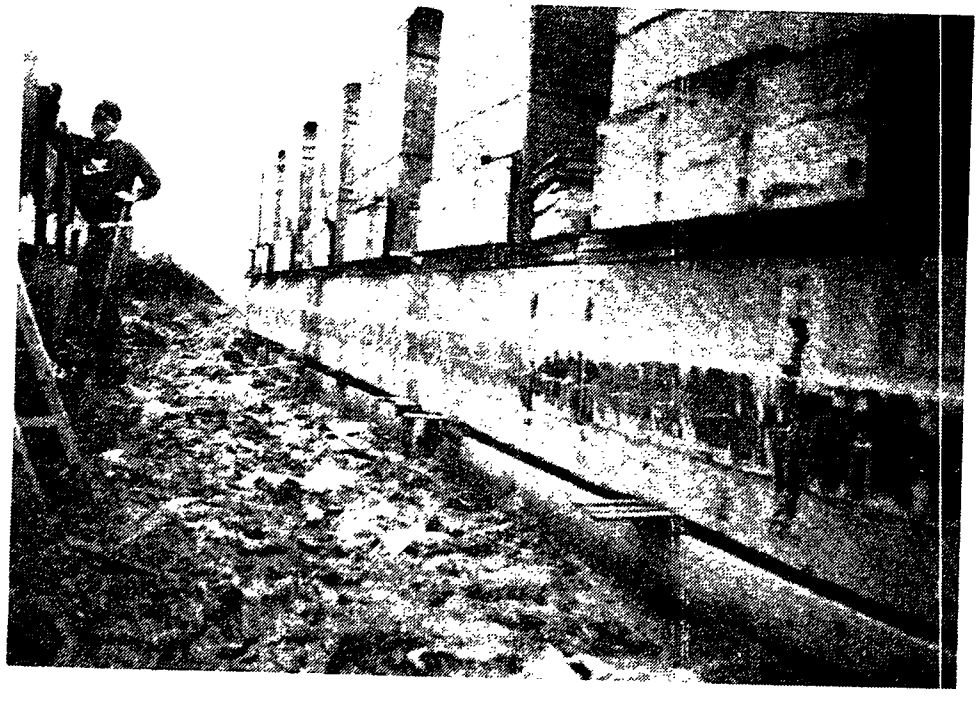


FIGURE 13 FINAL JACKED POSITION OF WALL

Summary and Conclusions

In 1987, new, more cost-effective methods of retrofitting damage foundations and tilt correction to sloped house foundations were proposed. The application of this retrofit technique is described herein and involves a method where steel straps are attached to foundation bearing walls. The advanced tilt correction methodology, also described in the paper, basically consists of: 1. reinforcing the upper section of the foundation walls using the advanced strap reinforcement technique; 2. inserting jacks in windows cut in the walls immediately below reinforced sections; 3. incrementally jacking level the reinforced wall sections and house; and 4. backfilling the separation created by lifting the reinforced section off the remainder of the foundation with mortar/concrete.

In this paper a field test of the proposed repair technique for retrofitting residential foundation walls damaged by mine subsidence is described and evaluated. The retrofit technique was designed to be cost efficient, simple to build, effective in strengthening foundation walls, and crack resistant. The method used consists of repairing wall cracks, externally reinforcing the walls, and covering the walls with a crack resistant aesthetically pleasing fiber-cement coating. The field test was located at a rural location just outside of West Frankfort, Illinois. Four block high, forty-foot long block walls, resting on strip footings were used in the test. These walls were located over a longwall coal mine.

The retrofit was simple to construct using only current methodology and was cost-effective in resisting the subsequent ground movements.

Over the monitoring period of 1.5 years, the hogging curvature of the ground had increased slightly to an angular distortion of 1/1,210. The angular distortion of the retrofitted foundation walls were about one-half that of the adjacent ground. Only slight changes in longitudinal strain have occurred over the monitoring period. Both the two-strap and one-strap foundation walls, T5 and T6 respectively, performed well. Also, the fiber-cement finishes have retained their aesthetic appeal after one and one-half years of weathering.

This proposed method of tilt correction was successfully and quite easily applied to relevel two field test foundations near West Frankfort, IL. These foundation walls were jacked up differentially about 20-23 cm (8-9 in.) In addition to jacking the walls level, it was also possible to eliminate the hogging in the wall. The wall slopes and deformations were caused by longwall mining activity. After all the jacking was complete and the walls were straight and level, the most stressed point in the steel reached about 20% of its yield strength, thus leaving adequate capacity to resist residual subsidence movements. In fact, it was found that the upper wall could be lifted about an inch at one jack without stressing the steel excessively.

Acknowledgements

By invitation from the U.S. Bureau of Mines in Minneapolis, MN and the cooperation of the University of Tennessee the test foundations at West Frankfort, IL became available for performing repair research. The author would like to especially thank Professor R. Bennett at the University of Tennessee, Knoxville, and Dr. L. Powel at the U. S. Bureau of Mines, Minneapolis, MN for their assistance in performing this work. We also are deeply indebted to Mr. E. Murphy of the Illinois Mine Subsidence Insurance Fund for his continuing support. The research was funded by The Illinois Mine Subsidence Insurance Fund. Also, the author would like to thank the staff of the Civil Engineering Department at the University of Illinois, especially Professor W. L. Gamble, for their contributions to the research projects. The manuscript was typed and some of the graphics were prepared by Mr. M. Reinking of Engineering Consultants.

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CONSIDERATION OF SUBSIDENCE IN LAND USE PLANNING POLICIES AND PROCEDURES IN ENGLAND[1]

David Brook[2]

Abstract. A widespread potential for subsidence requires consideration within a planning system which aims to control development in the public interest. A research strategy involving review at the national level of causes of subsidence followed by specific studies to develop techniques for use by planners provides the basis for planning guidance on the development of unstable land. This is now being taken forward with specific respect to subsidence to enable proper consideration in development plans and in considering applications for planning permission.

Introduction

England has a long history of mining dating from prehistoric times, although the principal period of activity was from the late 18th to the middle of the 20th century. Natural underground cavities due to dissolution of soluble rocks (chalk, limestone, gypsum, halite) are also widely distributed. In consequence there is a widespread potential for subsidence which may pose a threat to public safety and to property and land use.

When events occur, sub-editors have a field day with headlines such as:-

- "Crater fright for family" (Hitchen & Stewart 1995)
- "Falling apart at the coal seams" (Prior 1989)
- "Mine of information needed as homes face collapse" (Porter 1992)

Such events often represent a historical legacy of the time when little was known about the potential for subsidence and there was little, if any, control over the location of development. There are, however, a number of examples where recent development has been affected by subsidence. If the developer and those controlling development had taken account of the potential for subsidence most of these events could have been avoided. Whilst recognising that the responsibility for safe development and secure occupancy of a site rests with the developer and the landowner, we need to look at existing controls over development, how they have been exercised in practice and the improvements that can be made.

Controls on Development

The main controls over development and land use in England are through the operation of the planning system and the building regulations.

The Planning System

The legislation is now contained in the Town and Country Planning Act 1990, as amended by subsequent acts. The fundamental requirement is that development may not be undertaken without planning permission.

Development is defined in the 1990

[1] Paper presented at the 1996 National Meeting of the American Society for Surface Mining and Reclamation, Knoxville, Tennessee, May 18-23, 1996.

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Act as :- "The carrying out of building, engineering, mining or other operations in, on, over or under land, or the making of any material change in the use of any buidlings or other land." However, certain works and uses do not constitute development (eg the use of land for the purpose of agriculture or forestry) and the Town and Country Planning General Permitted Development Order 1995 gives general permission in advance for certain defined classes of development, mainly of a minor character (eg small extensions and alterations to dwelling houses).

Subject to these exceptions, specific planning permission is required for development upon application to the local planning authority (lpa). The Secretary of State for the Environment (SoS) has call-in powers to require in exceptional circumstances that applications be referred to the SoS for decision and aggrieved applicants may appeal to the SoS against the lpa's decision, ie against refusal of permission, against conditions imposed on the grant of permission or against failure to determine an application. There is no provision for appeal by other parties, though decisions may be challenged in the courts on legal grounds.

The lpa's decision on an application must "have regard to the provisions of the development plan, so far as material to the application, and to any other material consideration." Other material considerations are not defined and they can cover a wide field. They must, however, be genuine planning considerations, ie they must relate directly to the physical development and use of land. Permission may be granted subject to such conditions as the lpa or the SoS think fit, provided that they are necessary, relevant to planning, relevant to the development to be permitted, enforceable, precise and reasonable in all other respects. There is a presumption in favour of allowing applications for development, having regard to all material considerations, unless the development would cause demonstrable harm to interests of acknowledged importance. A developer is not required to prove the case for the development proposed and, if the lpa consider it necessary to refuse permission or to impose conditions, the onus is on them to give reasons for permission being refused or conditions imposed.

Development plans, as prescribed by the Acts, provide the framework for the control of development. They are intended to provide a firm basis for rational and consistent decisions on planning applications and a means of coordinating the needs of development and the interests of conservation. Where the development plan is material to a development proposal, the Planning and Compensation Act 1991 requires an application or appeal to be determined in accordance with the plan unless material considerations indicate otherwise. The system is thus essentially plan-led.

Structure Plans and Part I of Unitary Development Plans (UDPs) provide the broad strategic framework based on a written statement which may be accompanied by diagrams. Local Plans and Part II of UDPs are more specific, develop the strategic policies in the local context and contain land-use proposals which are map-based. Development plans are mandatory and are required to be consistent with national planning policies, with any regional planning guidance issued by the SoS and with each other.

<u>The Building Regulations</u>

The Building Regulations 1991, made under the Building Act 1984, provide an additional means of controlling built development. They are concerned primarily with the safe construction of a building and the provision of services to and from it so as to secure "the health, safety, welfare and convenience

of persons in and about the building." The regulations apply to building work in general, control of services and fittings and material change of use. However, there are exemptions such as greenhouses and agricultural buildings, temporary buildings, some small detached buildings and extensions.

The building regulations authority has a duty to see that building work complies with the regulations. If the work fails to comply, the developer may be required to alter or remove it. However, the submission of detailed plans for checking of compliance with the approved document is not compulsory and works may commence two days after the submission of a building notice.

Relevance of planning and building regulations to subsidence

Subsidence is not specifically mentioned in the Town and Country Planning Acts. It is clear, however, that planning matters are related to the physical development and use of land. This relates to the content of development plans, to decisions in individual cases and to the imposition of planning conditions.

The preparation of development plans requires lpas to undertake a survey of their area which must include "a review of the principal physical characteristics of the area" and the plan must contain statements on "measures for the improvement of the physical environment." In principle, there is as much justification for including policies on subsidence as a material planning consideration as there is to include other commonly accepted planning constraints such as Sites of Special Scientific Interest, Ancient Monuments, areas of high landscape value or high quality agricultural land.

Similarly, the physical characteristics of a site are relevant to the development and use of that site and can be considered as material considerations when deciding a planning application. The relevance of subsidence to planning decisions was recognised by the Ministry of Town & Country Planning (1949) when arrangements were announced for lpas to obtain the advice of the mineral valuers of the Valuation Office of the Board of Inland Revenue where any question of subsidence is involved.

In 1961, the Minister of Housing and Local Government gave his opinion that "decisions should not be taken to permit surface development without giving due weight to what is known or can be conjectured about the stability of the site" (Ministry of Housing & Local Government 1961). Whilst this advice was directed specifically to lpas in coal mining areas, it is clearly a general principle which is more widely applicable to other mining and other forms of land instability. There had also been a long-standing procedure for consultation with the nationalised coal industry. With privatisation of the coal industry, the Town and Country Planning (General Development Procedure) Order 1995 requires a lpa to consult the Coal Authority before granting planning permission for "development which involves the provision of a building or pipe-line in an area of coal working notified by the Coal Authority to the local planning authority."

Despite these provisions, it was clear by 1990 that the potential for subsidence was often not considered by lpas when deciding planning applications. Many planners regarded subsidence as a matter for the developer alone, or for the building regulations. At that time, the building regulations covered specific causes of ground movement such as swelling, shrinkage or freezing but there was dispute as to whether they covered subsidence due to mining, except in so far as the subsidence was caused by the weight of the building itself. Concern had also been expressed about the liabilities of lpas granting planning

permission for developments which were subsequently affected by subsidence.

In consequence, the Department of the Environment (1990) issued Planning Policy Guidance Note (PPG) 14 - *Development on unstable land* to clarify the position with respect to all forms of ground instability. This explained the responsibilities of the various parties to development and stressed the lpa duty to take all material considerations into account when reaching decisions on development proposals. The stability of the ground, in so far as it affects land use, is a material consideration and should be taken into account when deciding a planning application. Ways in which instability might be treated in development plans and in considering applications for planning permission were outlined and the role of expert advice was emphasised.

PPG 14 explained that it is the function of the planning system to determine, taking account of all material considerations, of which instability is only one, whether a proposed development should proceed. Having made that decision, for certain types of development, it is the function of the building regulations to determine whether the detailed design of buildings and their foundations will allow the buildings to be constructed and used safely.

Subsequently, the Building Regulations 1991 drew attention to potential problems caused by land instability in requiring that "The building shall be constructed so that ground movement caused by (a) swelling, shrinkage or freezing of the subsoil; or (b) landslip or subsidence, in so far as the risk can be reasonably foreseen, will not impair the stability of any part of the building."

Minerals Planning Guidance Note (MPG) 12 (Department of the Environment 1994) has taken forward the general principles in PPG 14 to advise lpas, landowners and developers of the specific consideration which needs to be given to the problem of disused mine openings and to demonstrate the need to compile information on mined ground in readily accessible, well organised data systems. Particular attention was given to the need for and operation of planning control over the treatment of mine openings in different circumstances (eg on closure of a mine, for public safety reasons, to safeguard conservation interests and to facilitate development) and to the consideration which needs to be given to mine entries and mine openings in determining planning applications and in development plans.

Department of the Environment Research on Subsidence

The planning guidance that has been issued has been based on a solid foundation of research on land stability issues which proceeded in line with a well defined strategy. This involved, firstly, reviews on a national scale of individual causes of land instability to determine what work had been done and what was needed. The second element of the strategy was to conduct specific area and topic-studies to develop techniques for the investigation and management of particular problems which would be more generally applicable.

The National Reviews of Land Instability

The general objectives of all the reviews were to examine the geographical and geological distribution of problems, causes and mechanisms, effects on the land surface and on development, methods of investigation, risk assessment and remedial measures, the legislative and administrative provisions for dealing with problems and priority topics for further research.

Review of Mining Instability in Great Britain. Carried out by Arup Geotechnics (1992), this involved the compilation

and interpretation of data from readily available publications and documentary records. For the purposes of the review, minerals were classified into five types :- metalliferous (non-ferrous) ores;rocks (eg sandstone, limestone,chalk); coal and associated minerals (eg fireclay, Coal Measures ironstone); iron ore outside coalfields; and evaporites (eg rock salt, gypsum).

Mining has been found to have taken place throughout England and Wales, although in some counties it is of only minor extent, and in all Scottish Regions except Western Isles (figure 1). Extensive coalfield mining covers more than 20 per cent of the areas of 16 counties in England and Wales and 3 Scottish Regions. Mining methods include both partial extraction (primitive shafts and headings such as bell pits or chalk wells, pillar and stall mines and brine pumping and solution mining) and total extraction methods (pillar and stall, stoping, longwall extraction and sub-level or block caving). The principal types of mining instability which may affect the ground surface are:-

- collapse of mine entries (shafts and adits);
- crown holes (localised collapses into mine voids); and
- general subsidence (also called areal subsidence, a wide saucer-shaped depression of the land surface without the massive discontinuities associated with crown holes).

The results of the review are presented in 3 volumes comprising:-

- Ten regional reports describe the extent of mining for the 5 mineral types on a county and district basis and contain advice to planners to be used in conjunction with PPG 14. Comprehensive schedules of mining areas indexed by reference codes, mine names and National Grid references provide a link to the associated !:250,000 scale maps showing areas where mining is known or suspected to have taken place and, therefore, should be regarded as a planning consideration.
- Five technical reports consider the effects of mines, investigation methods, preventive and remedial measures, monitoring methods and procedures for locating disused mine entries. Guidance is given on the procedures that should be adopted, their fitness for purpose, procurement and the priorities for further research as a basis for better technical guidance.
- Eleven case study reports, which provided the basis for the technical reports illustrate the variety of subsidence problems experienced due to mining instability. Each outlines the characteristics of mining and associated subsidence events in a specific area and summarises the site investigations and preventive and remedial measures which were used to deal with the problems.
- A separately published summary report and an executive summary provide a brief outline of the findings of all elements of the work undertaken for this review together with two summary maps, for northern and southern Britain respectively, at 1:625,000 scale of areas of mining consideration.

<u>Review of Natural Underground Cavities in Great Britain.</u> This review was carried out by Applied Geology Ltd (1993). Natural cavities were classified into 4 categories based on the mode of formation :- by dissolution (limestone, chalk, gypsum, salt and calcareous sandstone); by cambering (of competent cap-rock overlying less competent mud-rocks associated with past glacial and periglacial episodes); by marine erosion (sea caves in coastlines composed of competent rocks); and by other processes (eg soil piping, scour hollows resulting from periglacial processes and erosion of natural discontinuities).

Natural cavities were found to be widespread but not uniformly distributed throughout Great Britain (figure 2); 20,000 occurrences are recorded in the project database. The most significant concentrations are dissolution cavities in the Chalk of south east England and

East Anglia and the Carboniferous Limestone of the Yorkshire Dales, Derbyshire, north and south Wales and Avon and Somerset, with lesser concentrations due to cambering in the Permian, Jurassic and Cretaceous sequences of North Yorkshire, the East Midlands, Gloucestershire and Dorset. The cavities may be filled with air, water or unconsolidated deposits. Most instability is related to the downward movement of cover deposits rather than the cavity host rock. It may be sudden where air-filled voids are destabilised or occur more slowly with consolidation of loose infill deposits. The most common subsidence trigger involves water, often related to human activities.

The results of this review are presented in a similar format to those of the mining review with:-

- 10 regional reports (with accompanying 1:250,000 scale maps and a computerised database) describing the types and spatial distribution of natural cavities related to administrative areas and assessing them as constraints and amenities. Particular attention is given to regional experience of cavities affecting construction and engineering works, mining and tunnelling, mineral extraction, waste disposal, water resource development and public open space. Brief case histories are included within the regional reports.
- 3 technical reports discussing the nature and occurrence of natural cavities and their influence on planning and development, the range of site investigation techniques which may be appropriate for cavity detection and the ground treatment and structural design techniques available to overcome potential problems.
- A summary report with accompanying 1:625,000 scale maps of northern and southern Britain.

Techniques for Use by Planners.

Four examples are given below of the specific area- and topic-studies which have been carried out to assist planners in giving consideration to the problems of subsidence.

Development Advice Maps for Mining Subsidence. This project aimed to develop a method for preparing development advice maps (DAMs) giving advice to planners on abandoned shallow mining in the South Wales coalfield. A pilot study in the Ebbw Vale area (Ove Arup & Partners 1985) was followed by a further study in the Borough of Islwyn (Scott & Statham 1995) to confirm the methodology and to devise procedures enabling the maps to be used in the planning process.

Both studies were supplemented by a detailed review of subsidence incidents in the South Wales coalfield (Statham & Treharne 1991). This showed that 75 per cent of the 400 subsidence events recorded in about 35 years were collapses into workings at outcrop or mine entrances. The remainder were crownholes with an upper limit of migration of 8 to 12 times void height. Almost 70 per cent occurred in open land and less than 20 per cent caused damage to buildings or structures; only one example of injury to people was traced. Overall the risk of subsidence in the South Wales coalfield is very low, though it is higher at mine entrances and near to seam outcrops.

Preparation of the DAMs involved a detailed desk study from geological, mining, historical and other archival records to establish workable seams and their seam reputation for subsidence. Working maps showing modified geology, mining records, rock cover to seams and drift cover were then used to prepare maps showing zones where:-

- abandoned shallow mining is known or anticipated;
- abandoned shallow mining is possible;
- abandoned shallow mining is unlikely; and
- positions of mine entrances.

Draft "Planning guidelines" and "Guidance notes for potential applicants" were incorporated in the planning process by Islwyn Borough Council and the operation of this system was monitored for a year to assess the accuracy and worth of the DAMs and their usefulness in the planning process. It was concluded that the maps are a valuable tool, that their development should be extended to other areas of the South Wales coalfield and that the techniques could be applied to other coalfield areas in the United Kingdom.

Subsidence in Norwich. Following a number of fairly spectacular incidents in the late 1980s, a thorough survey of the causes of subsidence in Norwich was undertaken (Howard Humphreys & Partners Ltd 1993). This involved an examination of the records of subsidence and their causes, determining the extent and finding ways of managing the subsidence problem.

The study showed that not all of Norwich suffers from subsidence problems. Where they have occurred they have often been of a relatively minor nature, though there have been a number of severe subsidence events. The majority of subsidences were found, somewhat surprisingly, not to be associated with underground chalk mines, though the most severe ones were. A high proportion of the subsidences are associated with damaged or leaking water pipes and many of the subsidence problems have been exacerbated by poor construction and maintenance of property. The study concluded that the risk was relatively low, despite the many factors which contribute to the subsidence problem (chalk mines, cellars, undercrofts and wells, natural underground cavities in the chalk, collapse of sewers and leakage from services and the presence and nature of backfills and infills).

Most of the subsidences can be dealt with by simple preventive or remedial works provided appropriate investigations are made by developers and good practice is observed in construction and alteration of buildings. Dealing with subsidence is an integral part of Norwich City Council's local plan and account is taken of potential instability when considering development proposals and planning applications.

The recommendations for similar studies elsewhere emphasise the importance of a comprehensive data collection exercise, the importance of and requirements for underground inspection of old mines and the value of using a computerised database and geographic information system.

Assessment of Subsidence Hazard due to Gypsum Dissolution. The review of natural underground cavities had identified as a priority for further work the gypsum-bearing strata on the eastern side of the Pennines. In furtherance of this a study was commissioned to assess the working of the planning system in reducing hazards due to subsidence related to gypsum dissolution and to prepare a draft framework of advice suitable for use by planners, developers, land and property owners, insurers and others.

The study was focused on the Ripon area of North Yorkshire, where such subsidence was well known through the work of the British Geological Survey (Cooper 1995), but also covered the wider extent of the outcrop of gypsum-bearing strata. The geological, geomorphological and hydrogeological controls on gypsum dissolution and associated subsidence were examined and a conceptual model of gypsum dissolution in the Ripon area was established as a basis for assessment of the subsidence hazard. Recommendations were made on the engineering and planning responses to gypsum-related subsidence (Thompson & others 1995). The wider area study enabled confirmation of the conceptual model and conclusions to be drawn on the wider applicability of the framework of

advice for planners and others.

Treatment of Disused Mineshafts. Because of concern over the incidence of subsidence damage associated with disused mine opening, the Local Authority Associations and the Royal Institution of Chartered Surveyors made separate representations to the DOE about the need for advice. A Working Group was established and a program of research was undertaken on the treatment of disused mine openings and on the information base on mined ground (Freeman Fox Ltd 1988a,b).

Existing practice and methods of treatment were reviewed, classified according to their function and evaluated in respect of the most significant factor, depth to bedrock, and cost. Based on the concept of treatment objectives as a prime starting point for determining treatment policy, procedure, practice and selection of treatment method, a draft framework of advice was presented to indicate the most appropriate and cost-effective measures covering the majority of situations.

Similarly, the need for information on mined ground was examined along with information sources and types. Mining activity on which information is required was classified into shafts, roadways and workings and storage and retrieval options, including current practice in those authorities who already had information systems, were examined and recommendations made.

These two research programs provided the basis for MPG 12.

The Way Forward.

The need for consideration of potential subsidence in land-use planning has long been recognised but there have been some deficiencies in practice. A number of initiatives have been taken in recent years founded on a solid basis of research into the types of problems, their distribution and significance to planning and development and the appropriate planning responses.

Advice which makes it clear to lpas that subsidence and other forms of instability are material planning considerations which should be considered when preparing development plans and determining applications for planning permission has been issued in PPG 14. This advice has been augmented in respect of the treatment of disused mine openings and the availability of information on mined ground in MPG 12. It may yet be too early to see the direct benefits of this advice but a number of other initiatives are being taken forward.

Recent and current research is providing the basis for the planning guidance to be taken further in respect of subsidence from mining, natural underground cavities and other causes of subsidence and settlement. The aim is to reach a wider audience than the technical expert so that awareness is raised amongst land use planners, developers and financial institutions, enabling proper consideration to be given to the issues that arise. The DOE has started to prepare guidance which will advise planners and others on how they can identify and assess the potential for subsidence in order to take it into account in their everyday planning decisions. It will also advise on the information requirements for sound planning and as a basis for investment by developers. This will be based on the research described above.

Proper consideration of subsidence in the planning process will assist in reducing the impact of subsidence on the economy and on people. Whilst the risk has, in most cases, been found to be generally low to very low, when subsidence occurs the results can be spectacular and costly. They cannot be ignored by decision-makers. There will undoubtedly be some impacts due to the historical legacy of development in areas which are

prone to subsidence but it should be possible to avoid these impacts in most new development and redevelopment. It is also recognised that most forms of subsidence can be mitigated by appropriate engineering but that this is generally more cost-effective when undertaken in advance of development rather than as a result of incidents occurring during or after the development has taken place.

It is important to emphasise that the development of policies on subsidence and planning is not aimed at preventing development, though in some cases that may indeed be the appropriate response. Rather it is to ensure that development is suitable, that any risk is mitigated by appropriate engineering and that both natural and human-induced physical constraints on the land are considered at all stages of planning. Proper consideration of the potential for subsidence will thus contrbute to the broad objectives of the planning system to achieve economy and efficiency in the use of land and the protection of the environment.

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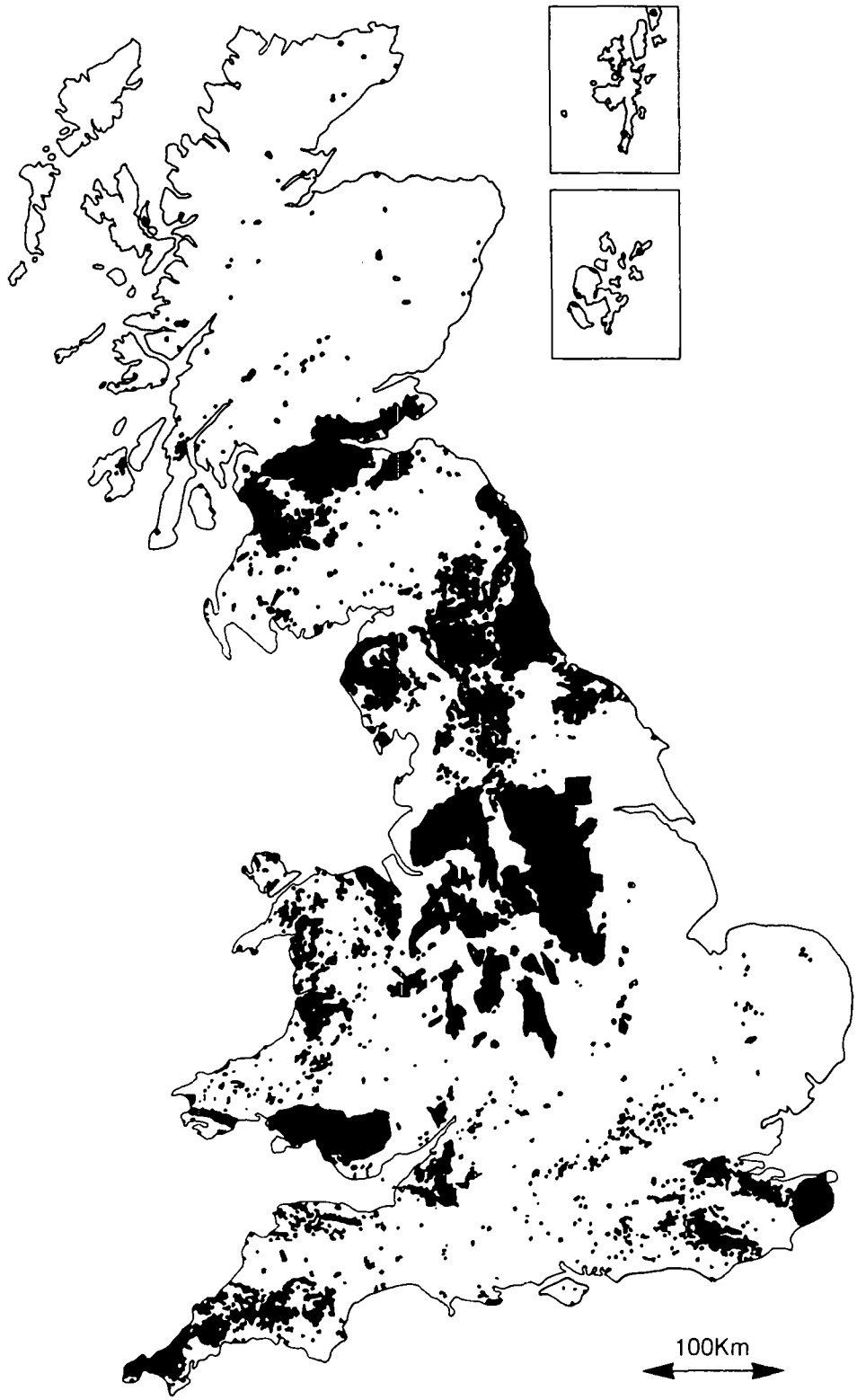

Figure 1. Mining areas in Great Britain (after Arup Geotechnics, 1992)

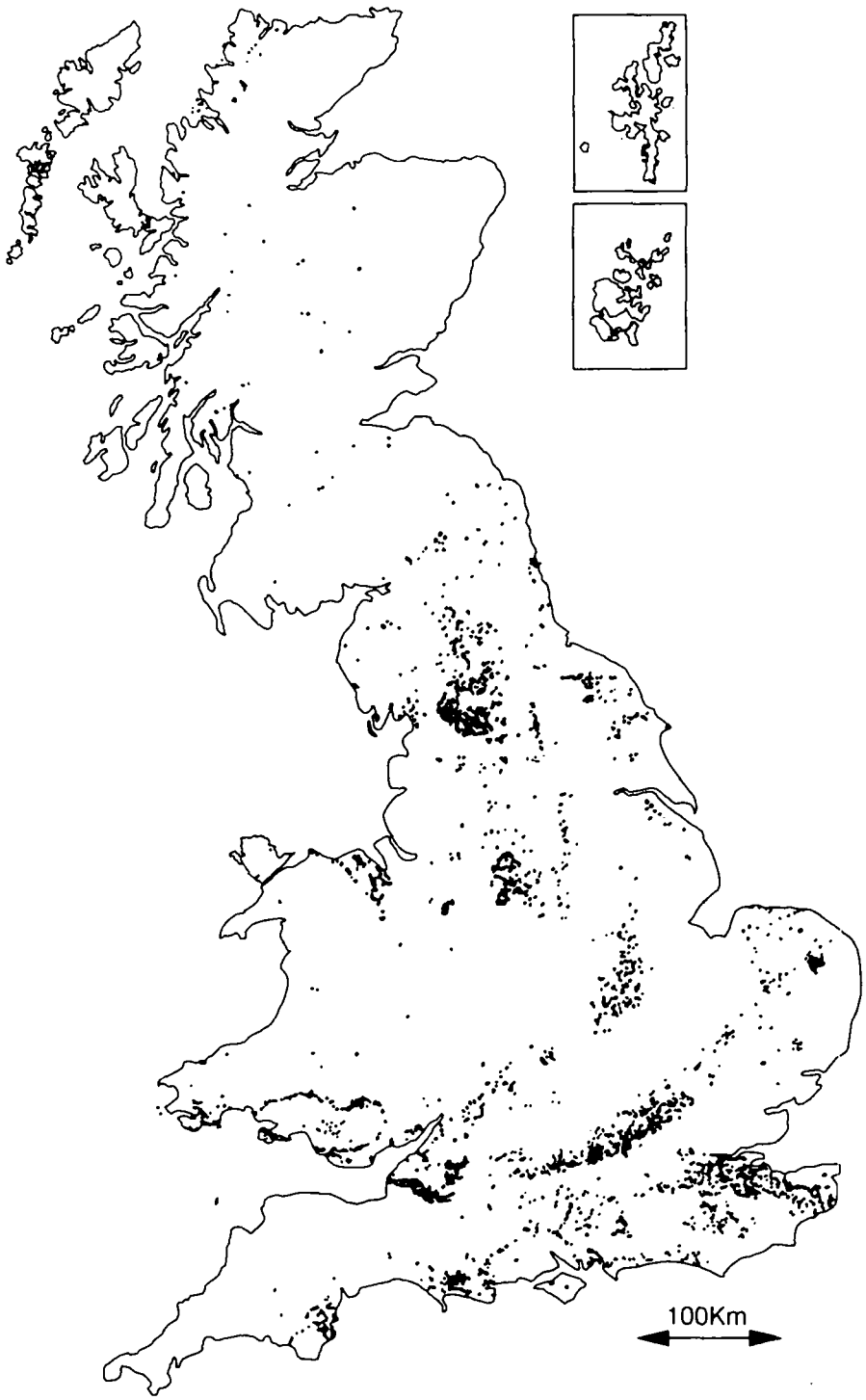

Figure 2. Distribution of natural underground cavities in Great Britain (after Applied Geology Ltd, 1993)

REPAIR OF THE CZAR SHAFT SINKHOLE

by

J. David Deatherage, Jack Starkey, Nils Gustavson

Abstract. After a portion of the Queen Mine parking lot collapsed into a sinkhole, an engineering study was conducted to confirm that the cause of the damage was settlement into the Czar Shaft in Bisbee, Arizona. Historical records were researched to describe and locate the original shaft, and a seismic refraction survey was performed to define the foundation conditions within and around the shaft. Recommended mitigation measures were developed to repair the parking lot. Mitigation included design and construction of a flexible geosynthetic cap and telltale settlement monitoring system.

Additional Key Words: geotextile, settlement, subsidence.

Introduction

The City of Bisbee operates the underground Queen Mine as a tourist attraction. Tourists park in a lot 18 meters by 46 meters (60 feet by 150 feet) in dimension set into the northeast side of a limestone hill overlooking Bisbee. In March of 1995 the center of the parking lot collapsed under the weight of a tour bus. After safely removing the tour bus, a sinkhole roughly one meter deep (three feet) and 4.5 meters (fifteen feet) in diameter was found under the parking lot, as presented in Figure 1. Water was flooded into the sinkhole at a rate of 20,000 to 40,000 liters (five to ten thousand gallons) per day for a one week period, in an attempt to consolidate the underlying fill. Subsequent excavation in search of lateral settlement limits widened the sinkhole to a diameter of 12 meters (40 feet). Settlements within the sinkhole of 25 to 33 centimeters (10 to 13 inches) per day were observed during the flooding. The settlement stopped when the flooding was discontinued. The sinkhole was temporarily backfilled with 32 truck loads of soil and rock. The City of Bisbee contracted with DEI Professional Services (DEI) to confirm the cause of the sinkhole, and to develop a mitigation design to bridge and cap the failure.

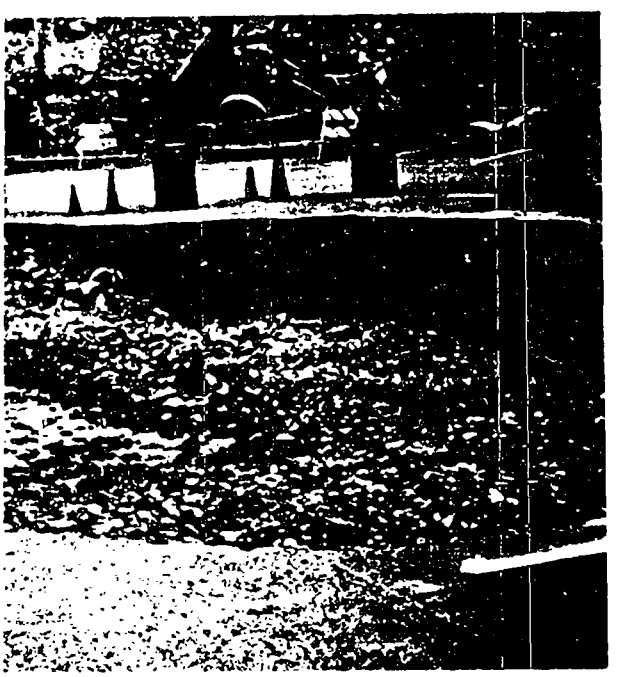

Figure 1: Queen Mine Parking Lot Sinkhole

(1) Presented paper at the Annual Conference of the American Society for Surface Mining and Reclamation, Knoxville, TN, May 19-25, 1996.

(2) J. David Deatherage, P.E., is the senior geotechnical engineer and Jack Starkey, C.D.T., is the senior civil designer with DEI Professional Services in Phoenix AZ, 85254. Nils Gustavson is Public Works Director for the City of Bisbee, AZ, 85603.

Investigation

DEI reviewed available literature relating to the site and conducted a limited geotechnical field investigation as part of the study. The field work included seismic profiling of the sinkhole, and additional sinkhole water flooding and settlement measurement.

Literature Review

Research of historical information relating to the site was conducted from literature and publications in the Bisbee Mining and Historical Museum. The Czar Shaft was sunk by the Copper Queen Company in 1885 to a depth of 134 meters (440 feet). The Shaft provided access to a carbonate copper ore deposit, (Graeme, 1981). The carbonate ore is found within early Cretaceous limestone rock included in the Bisbee Group. (Nations, 1983). The proximity of copper-rich igneous rock and limestone rock produced a classic zone of enrichment. Metallic solutions percolated into seams and fractures in the porous limestone, depositing metal-rich minerals, (Francaviglia, 1982).

Of particular interest in the Francaviglia publication were early 1900 period photos of the Czar Shaft site that show considerable construction related filling of the site. The filling was necessary to produce a flat bench on the side of the limestone hill for the mining buildings relating to the Copper Queen Mine and the Czar Shaft operations.

The Czar Shaft operated for sixty years until closure in 1944. The conditions within the shaft were reportedly very wet and cold. The shaft was capped with concrete in 1961 to stop trespassing. Shaft timbers reportedly decomposed from the lack of air, and in 1973 the head-frame collapsed. The shaft was then closed by backfilling, (Graeme, 1981).

Field Investigation

The parking lot is situated on the northeast side of a limestone hill. The Shaft is approximately 45 meters (150 feet) to the left of the Queen Mine Tour buildings, as shown in Figure 2. The sinkhole area in the parking lot is in the center of Figure 2, and barrels ring the temporarily backfilled area. Considerable filling is evident in the areas around the parking lot.

A limited geotechnical field investigation was conducted to differentiate the loose Shaft fill from the surrounding fill and native rock materials. DEI assisted AGRA Earth and Environmental in performing a refraction seismic survey of the sinkhole area. A 12-channel Geometrics signal enhancement seismograph was used for velocity data recording. Maximum depths of 4.5 to 6 meters (15 to 20 feet) were investigated using ten foot wide intervals for the geophones. Two seismic lines were run through the sinkhole, and two seismic lines were run on the inside and outside portions of the parking lot. Loose fill inside the sinkhole exhibited compression wave velocities of 225 meters per second (mps). Fill on the sides of the sinkhole exhibited significantly higher velocities below a depth of three meters, with a range of 430 to 800 mps. The line closest to the limestone hill detected the limestone at a three meter depth, with a velocity of 2,100 mps.

Figure 2: Temporarily Backfilled Sinkhole

The shallow seismic investigation confirmed the horizontal limits of the sinkhole to the immediate area of the surface expression of the settlement. The investigation also revealed that the pad for the original mining operations and parking lot was composed of a mixture of loose and medium dense fill overlying a dipping contact with the original limestone hill.

Additional water flooding was conducted in an attempt to initiate settlements within the fill over the Shaft. From May 15 through May 29 (15 days) from 27,000 to 40,000 liters (7,000 to 10,000 gallons) per day of water was run into the depression. Four measurement points were monitored within the depression, and at the end of the flooding no significant settlement was observed in any of the

measurement points. Approximately 15 centimeters (six inches) of total settlement was observed in the fill on the northeast side of the sinkhole when the material was first wetted. No additional settlement was observed after repeated flooding.

Results and Conclusions

The investigation confirmed that the Czar Shaft was originally located under the parking lot sinkhole. The historic review revealed past foundation distress problems related to settlement movement within the Shaft fill materials. These movements include the collapse of the original headframe over the Czar Shaft. Soils surrounding the Czar Shaft fill are not sufficiently dense to support concentrated loading from a rigid structural cap without possible significant settlement and movement.

The settlement into the Czar Shaft is similar to problems experienced with limestone sinkholes in karst areas of the country. Construction techniques employed to stabilize these features include the placement of thick reinforced concrete slabs, injection of cement grout under high pressure, and filling of voids with solids and concrete. Complicating geotechnical conditions at the Czar Shaft site include significant thickness of non-engineered fill in the upper portion of the shaft area.

DEI concluded that a structurally reinforced cap bearing on the adjacent fill was not a positive or economical alternative to repair the sinkhole. A mitigation measure was selected to bridge over minor future settlements beneath the parking lot. The mitigation allows for monitoring and future additional corrective measures for more significant settlements, should they occur.

A flexible geosynthetic cap was designed to distribute the parking lot loading over the densified existing and new Czar Shaft fill. The design was based on engineering judgement that considered the removal of as much loose overburden as site access and utilities would allow, and replacement of the loose overburden with geotextile reinforced fill.

The planned construction consisted of excavation of the sinkhole area to a depth of 4.5 meters (15 feet). The excavation had a flat bottom ten feet wide, and was nine meters (thirty feet) in length. Side slopes were designed as one to one (horizontal to vertical), with a rectangular top of excavation 12 meters (40 feet) wide and 18 meters (60 feet) long. Fill was specified as imported or excavated material consisting of clean sand, gravel and cobbles. The material was well blended with no isolated nests of gravels and cobbles, no particles larger than 30 centimeters (12 inches), and less than 8 percent passing the No. 200 sieve. The fill soils were free of organic matter, construction debris, and other deleterious materials. Due to the coarse nature of the fill, the compaction was specified as 85 to 95 percent of the maximum relative density. The compaction was evaluated visually by the engineer's representative without testing.

High strength woven geotextile fabric was detailed in an alternating pattern between subsequent fill layers. A wide roll width of 5.5 meters (18 feet) was specified to reduce overlap connections between sheets.

The geotextile was specified as a woven polypropylene fabric with the following minimum average roll values:

Property	Method	Value
Grab Tensile	ASTM D-4632	500 lbs.
Elongation	ASTM D-4632	20%
Wide Width	ASTM D-4595	4800 lbs. / ft.
Elongation	ASTM D-4595	18%
Mullen Burst	ASTM D-3786	1350 psi
Puncture	ASTM D-3786	140 lbs.
Trap. Tear	ASTM D-4533	250 lbs.
Roll Width	-	18 ft.
Roll Length	-	150 ft.

The geotextile was selected based on its high strength, flexibility, and resistance to biological and chemical degradation. Availability and shipping delivery also were important to the timing of the construction. For estimating purposes the material cost of the geotextile was approximately $2.00 per square meter.

Details of the design showing the placement of the geotextile sheets into each of the five layers filling the excavation are presented in the following Figures 3 through 7.

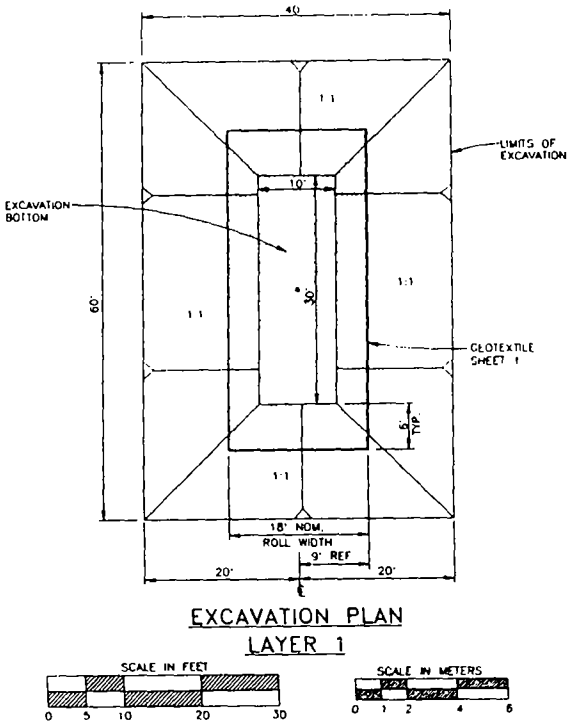

Figure 3: Plan View of Layer 1

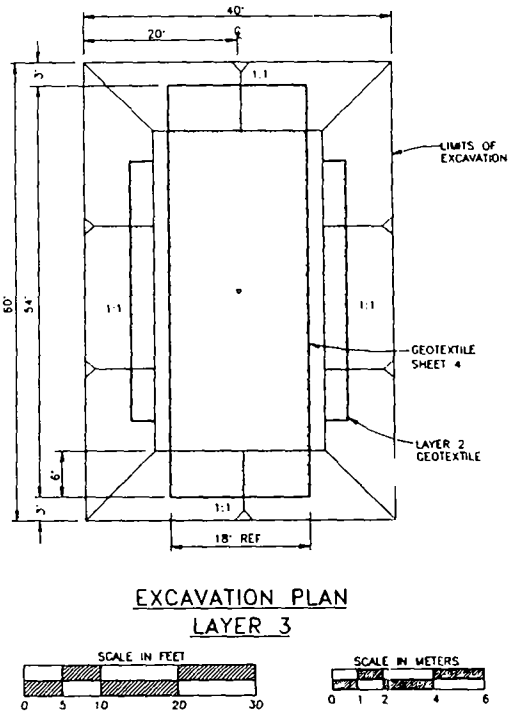

Figure 5: Plan View of Layer 3

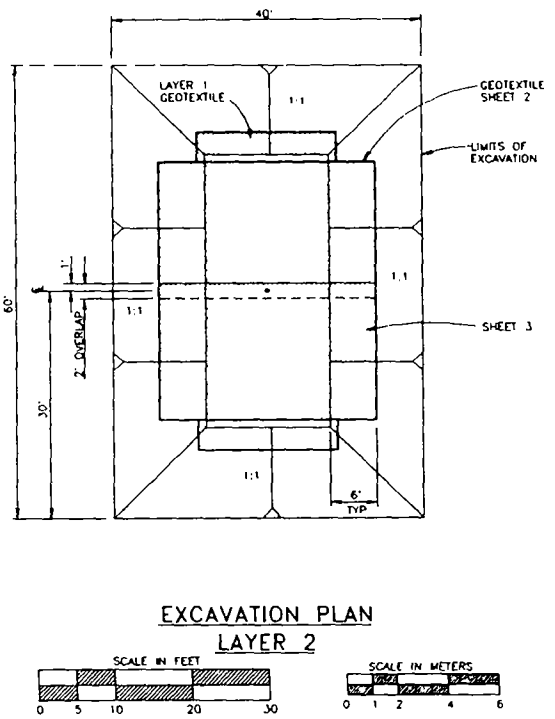

Figure 4: Plan View of Layer 2

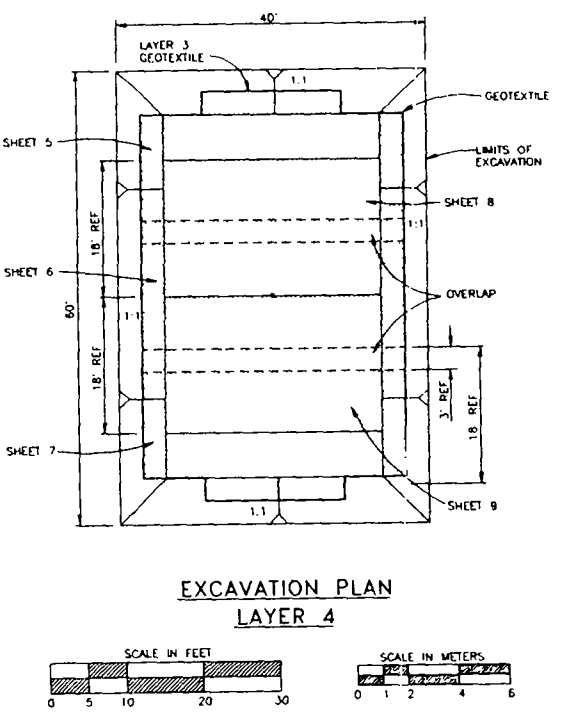

Figure 6: Plan View of Layer 4

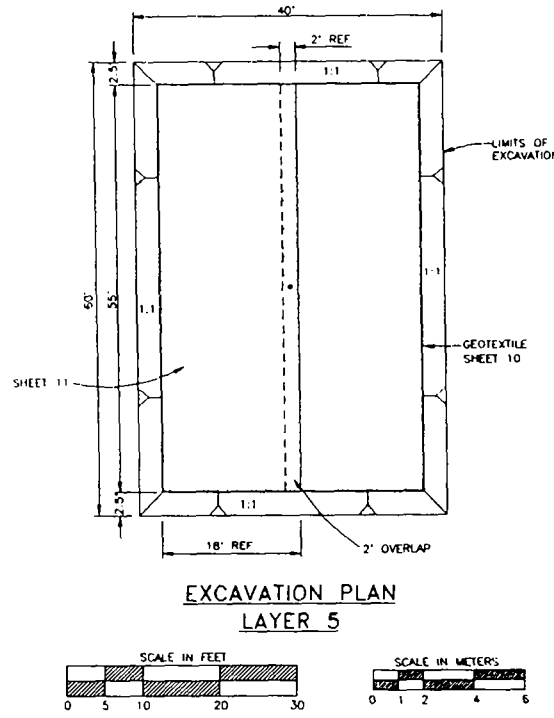

Figure 7: Plan View of Layer 5

Geotextile sheet arrangement on the side slopes of the excavation was carefully controlled to develop the maximum frictional resistance in the outer portions of each geotextile sheet. Care was taken not to overlap geotextile from one layer on top of another layer. This mitigated the significant loss in frictional resistance when one side of the geotextile is bedded against another sheet of geotextile. Frictional resistance of the geotextile sheet is similar to the backfill soil frictional strength if the geotextile is bedded on both sides by the soil. The full tensile capacity of the geotextile can be mobilized after a few feet of burial in the backfill soil. Figures 8 and 9 present sections through both the long and short axis of the excavation and geosynthetic cap system.

A telltale system was incorporated into the design to allow for periodic monitoring of the settlement of the Czar Shaft fill under the geosynthetic reinforced cap. A steel plate was specified to be placed in the center of the excavation, beneath the first layer of geotextile. A 15.25 centimeter (six inch) diameter Schedule 40 PVC pipe extends from the bottom of the excavation to the surface. The pipe provides an access port for the 2.0 centimeter (0.75 inch) diameter telltale pipe that extends from the bottom steel plate to the surface.

Construction

Construction started on August 11, 1995 with material removed from the planned excavation by a Cat 235L Trackhoe. All excavation and compaction operations were continuously monitored by a field engineer representative of DEI. Workers and equipment were not allowed into the bottom of the excavation because of the unknown Shaft fill conditions below the bottom of the excavation.

Excavated material was picked up with a Kawasaki 85Z2 loader and placed into a stockpile. DEI originally specified compaction with a vibrating plate compactor or compaction wheel attached to the end of the trackhoe bucket. The contractor was unable to provide the proper equipment, and elected to use the trackhoe as a battering ram to compact the soil with multiple impacts of the bucket. The base and side slopes of the excavation were compacted with repeated blows of the trackhoe bucket. A fire hose was used to supply water to the fill prior and during compaction.

The settlement plate and pipe were installed through the first layer of geotextile. Fill was then placed and compacted in two lifts totaling 90 centimeters (36 inch) thick over the geotextile. A minimum overlap of adjacent geotextile sheets was maintained at 0.6 meter (two feet). The geotextile was held in place during backfilling with steel spike nails driven into the walls of the excavation. The first 90 centimeter (36 inch) layer of fill was complete by the end of the second day of construction. By the end of the third day the third layer of geotextile was in place and covered with a lift of fill.

The settlement monitoring pipe was extended upward through each layer of the filling operations. On the fourth day the fill was placed over the fourth layer of geotextile, and a self propelled tamping foot compactor (Ingersoll Rand SD40D) was used to compact the fill layers. By the fifth day of construction the final geotextile layer was installed and the fill over the geotextile was compacted. Figures 10 and 11 present details of the settlement plate and telltale.

The contractor returned to the site several weeks after finishing the filling of the excavation to pave the filled portion of the parking lot, and to finish the surface completion of the telltale monitor. DEI

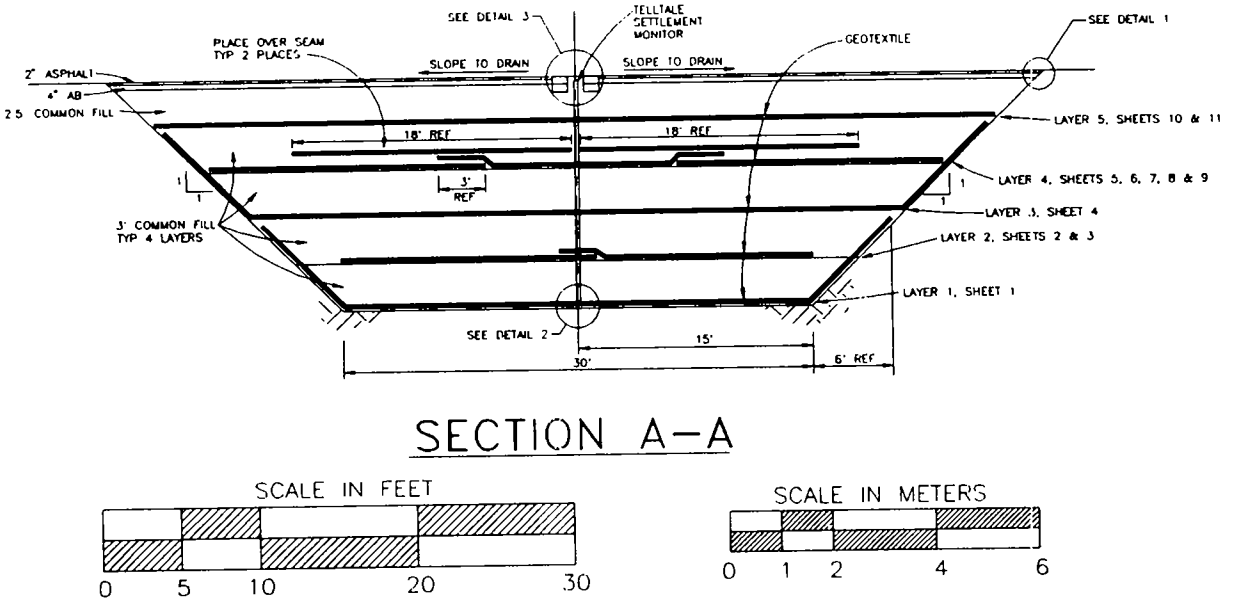

Figure 8: Section through the long axis of the cap.

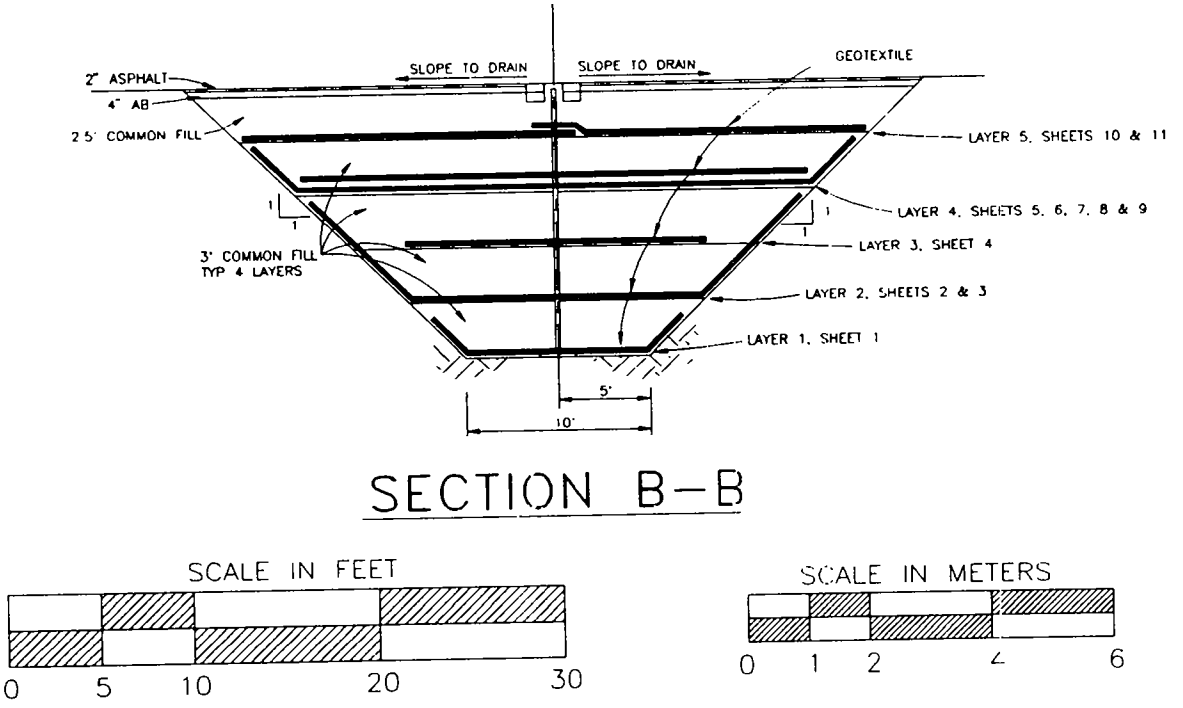

Figure 9: Section through the short axis of the cap.

returned to the site on November 6, 1995 to observe the completed construction. Figures 12 through 17 present photographs of the construction.

Telltale Settlement Monitoring

The telltale settlement monitor will be periodically inspected by representatives of the City of Bisbee. DEI suggested a monitoring frequency of weekly for the first month, then monthly for one year, then quarterly or twice yearly after the first year. Through the end of March 1996 there had been no measurable settlement beneath the geosynthetic cap. The intent of the geosynthetic cap and telltale settlement monitoring is to provide an engineering system to safely bridge minor settlements in the underlying shaft. The system will also provide a means to monitor and mitigate more significant settlements that may occur in the future. Monitoring will detect significant settlement and allow possible mitigation before the progression of damaging and potentially unsafe settlements. Mitigation will consist of injection of concrete grout through the monitoring pipe to fill any voids forming under the cap. If necessary this process can be repeated to maintain the integrity of the cap.

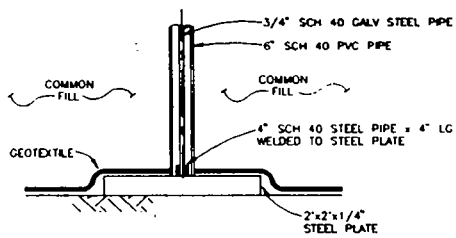

Figure 10: Detail of Settlement Plate

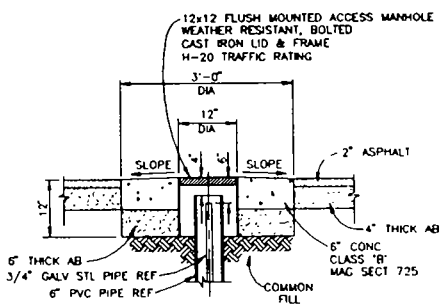

Figure 11: Detail of Settlement Telltale

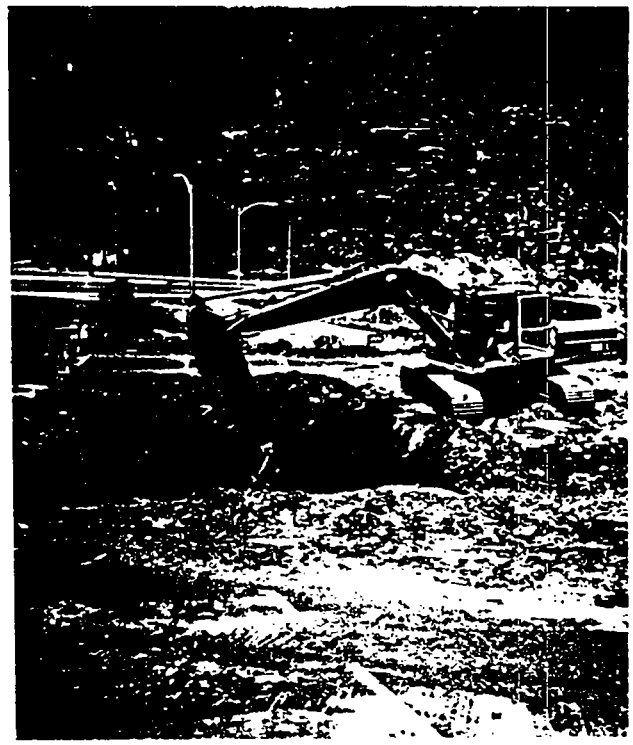

Figure 12: Excavation of Sinkhole

Figure 13: Compaction of Fill Over Layer 1

Figure 14: Placement of Layer 2

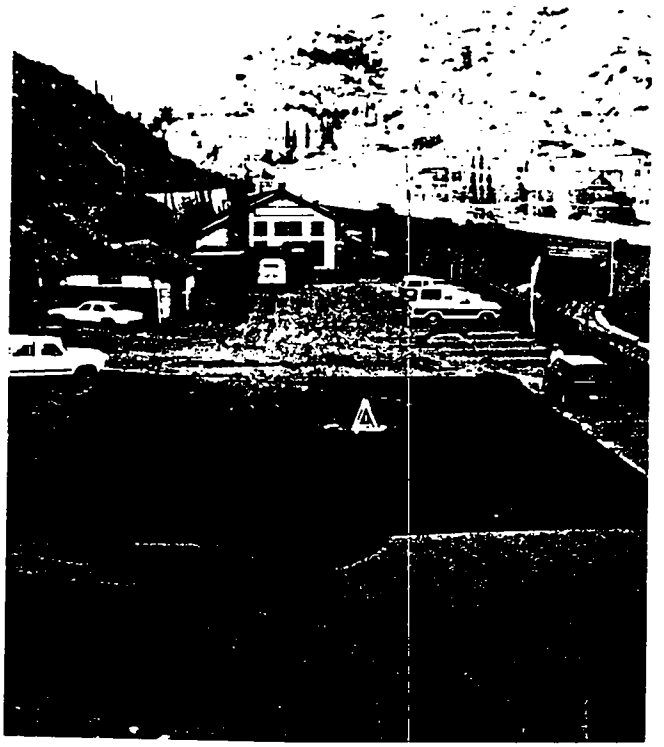

Figure 16: Completed Repaved Parking Lot

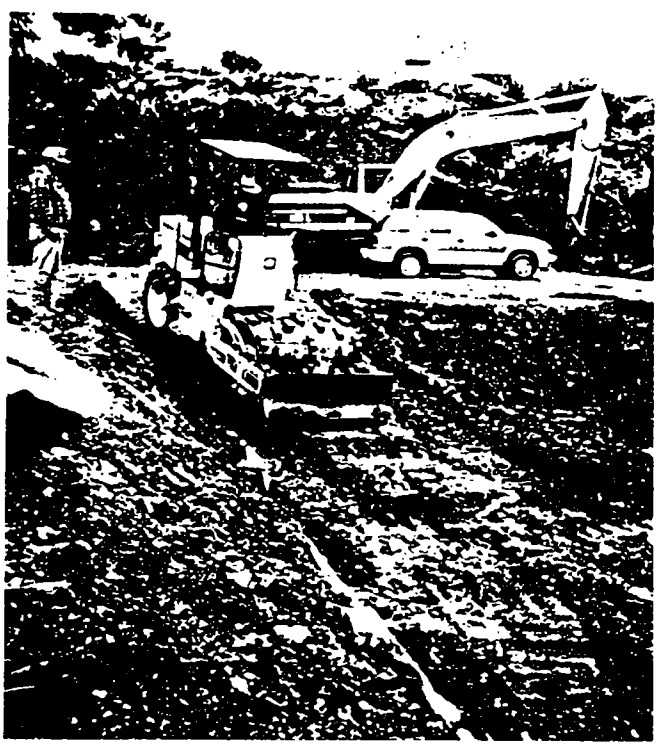

Figure 15: Compaction of Fill Over Geotextile

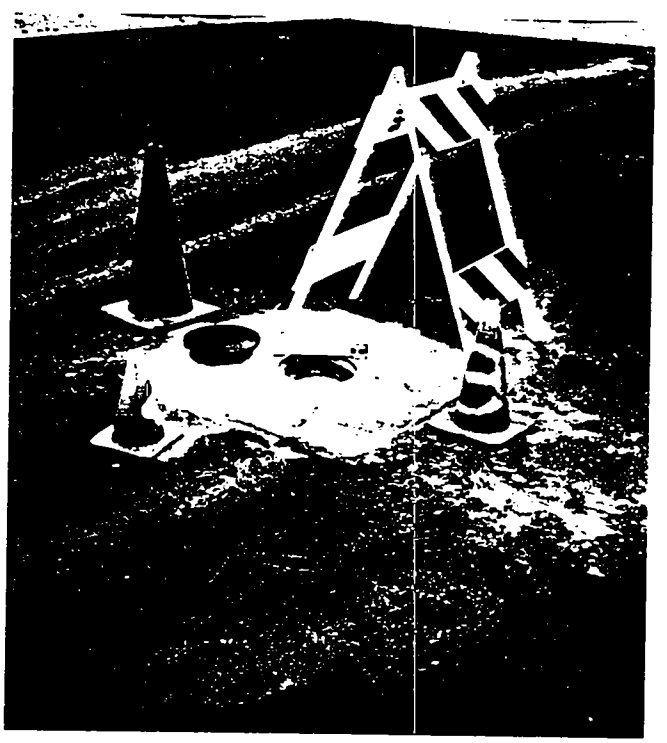

Figure 17: Settlement Monitoring Pipe

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MONITORING OF A HOUSE DURING MINING SUBSIDENCE

G. Lin[1], E.C. Drumm[2], R.M. Bennett[2], L. Powell[3]

Abstract: The structural response and subsequent damage to a residential house were monitored during an active longwall mining operation. The monitoring program consisted of survey monuments on the ground surface around the house and at various locations on the structure. Structural tilt, soil strain and soil pressures were also recorded. The structure was subjected to a cyclic loading consisting of initial tension, then compression, followed by a second tension in the ground surface. The unreinforced foundation experienced extensive cracking and was left with permanent displacements. Although the structure experienced significant shear and tilt during the subsidence, it was left with only minimal damage. The observations suggest that structures in subsidence-prone areas must be able to withstand significant angular distortions, especially during the initial tensile phase of the ground deformation. Current construction practices can be modified to induce composite action between the foundation and superstructure and thus improve the resistance to subsidence-induced ground movements.

Additional Key Words: subsidence, structural damage, foundation damage

Introduction

With the growth in demand for coal, and the resulting increase in coal production, underground mining will be performed under existing structures. In addition, new structures will be built over areas that were mined previously. Mining induced ground subsidence is estimated to result in damage to residential structures costing between 25 and 35 million dollars each year. This damage is estimated to exceed $1 billion over the final quarter of this century (Gray 1988).

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New construction procedures and techniques are needed to reduce structural damage due to mining induced ground movements. However, the structural response to ground movements is very complex due to interaction effects between the ground, foundation and superstructure. It is essential to understand these interaction mechanisms for the development of new damage mitigation procedures.

Although the subsidence above abandoned room-and-pillar mines is very unpredictable, the ground subsidence induced by a longwall mining is generally predictable both in terms of magnitude and timing. As such, longwall mining subsidence provides a unique opportunity to investigate structural response to subsidence.

A one-story ranch-type house in southern Illinois was monitored during longwall mining subsidence. The monitoring program consisted of survey monuments on the ground surface around the house and at various locations on the structure. Structural tilt, soil strain and soil pressures were also recorded. This paper presents some results from the monitoring program and discusses the interaction process between the ground, foundation and

superstructure. Some recommendations are provided in terms of inexpensive modifications to current construction procedures to minimize structural damage during mining induced ground subsidence.

Site and Structure Background Information

Structure

The house was a one-story wood framed ranch-type house constructed over a crawl space. The house has a concrete slab porch at the front and an annex on the back. The main structure has a plan dimension of about 17 m long and 8 m wide. The floor plan and elevation are shown in Figures 1 and 2, respectively. The walls were composed of fiber board and wooden siding over wooden studs. The interior walls were covered with gypsum wallboard. The house appeared to be in relatively good condition prior to the mining subsidence.

The main structure was supported on continuous unreinforced concrete strip footings about 0.6 m wide as shown in Figure 3. Although the footings along the four sides were connected at the corners, they were not at the same elevation. The footings at the south and west sides were poured approximately 20 cm higher than the footings at the north and east sides. A wooden beam, consisting of three nominal 50 by 200 mm boards, ran longitudinally down the center of the house. The beam was supported on six evenly spaced isolated concrete footings of approximately 0.6 m square. Wood joists spanned between the top of the foundation and the main beam. The house was anchored to the foundation with 9 mm L-shaped anchor bolts at 2 m on center. The sill was a 50 mm by 150 mm board with 18 mm holes for the anchor bolts. There was evidence of some rot in the sill.

Site Conditions and Mining Operations

A borehole was drilled to investigate the subsurface materials above the mine seam. It was found that the overburden consisted primarily of shale and siltstone, with some interbedded sandstone and three thin coal seams (Mehnert et al. 1992). The surface soils were loess deposits consisting primarily of low plasticity silts and clay. The soil layer was approximately 7 m thick with the soil having a plasticity index of 10, angle of internal friction

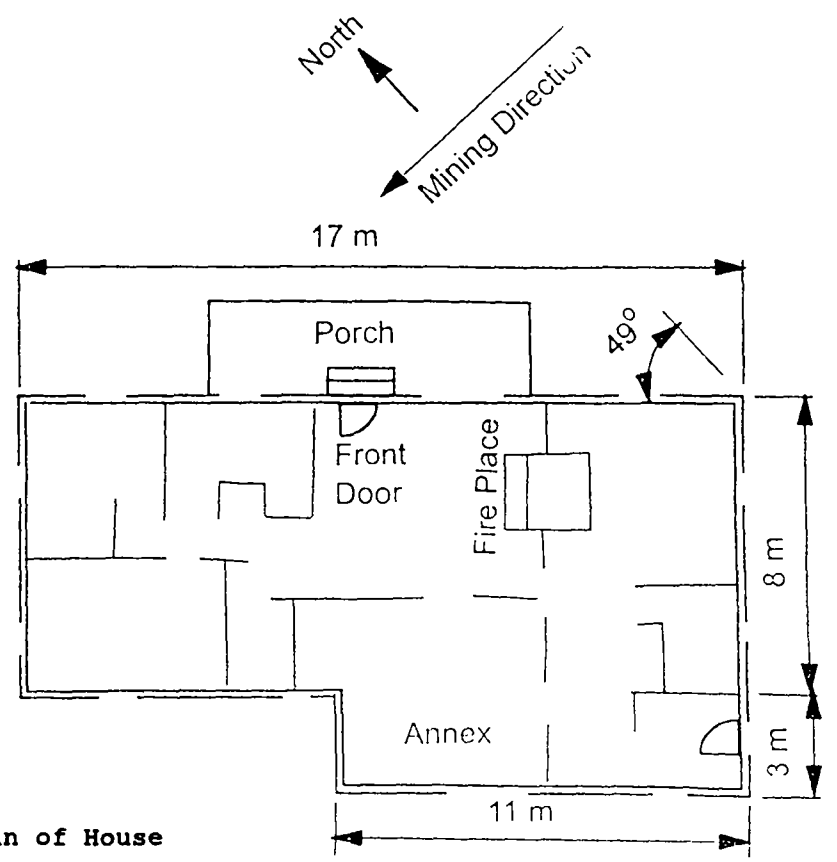
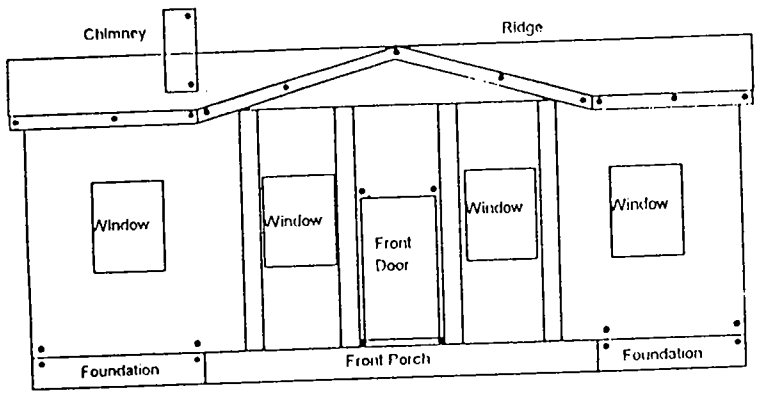
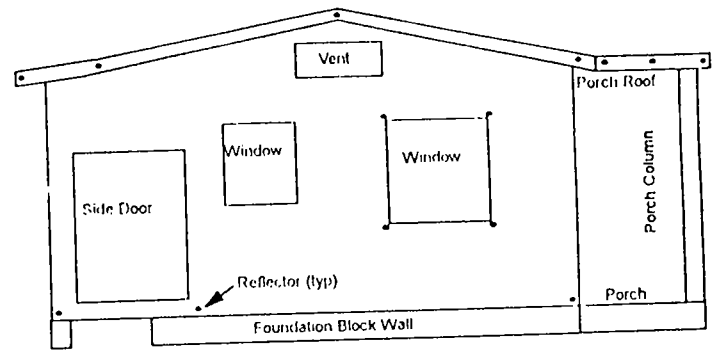


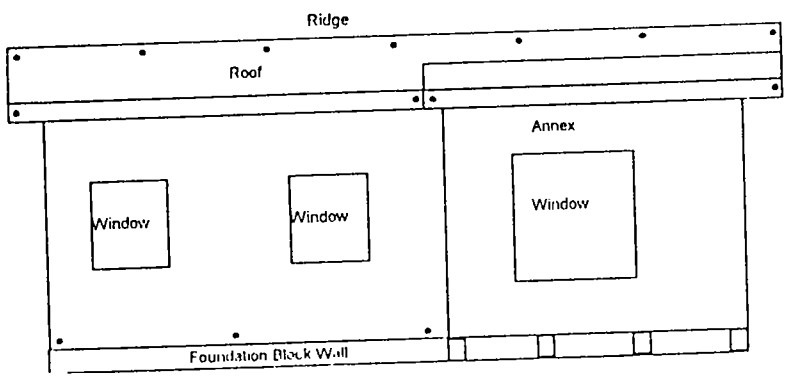
Figure 1 Floor Plan of House



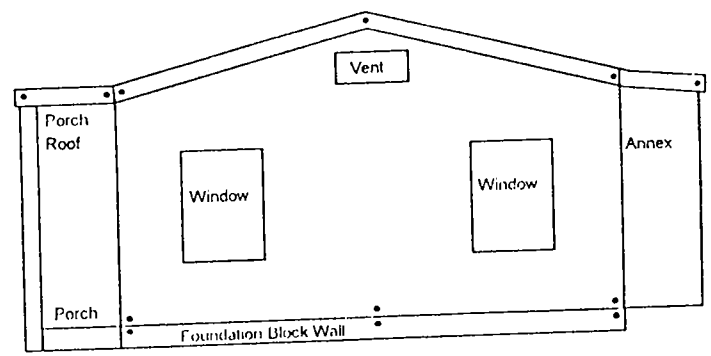
North Elevation



East Elevation



South Elevation



West Elevation

Figure 2 Elevation Views of House

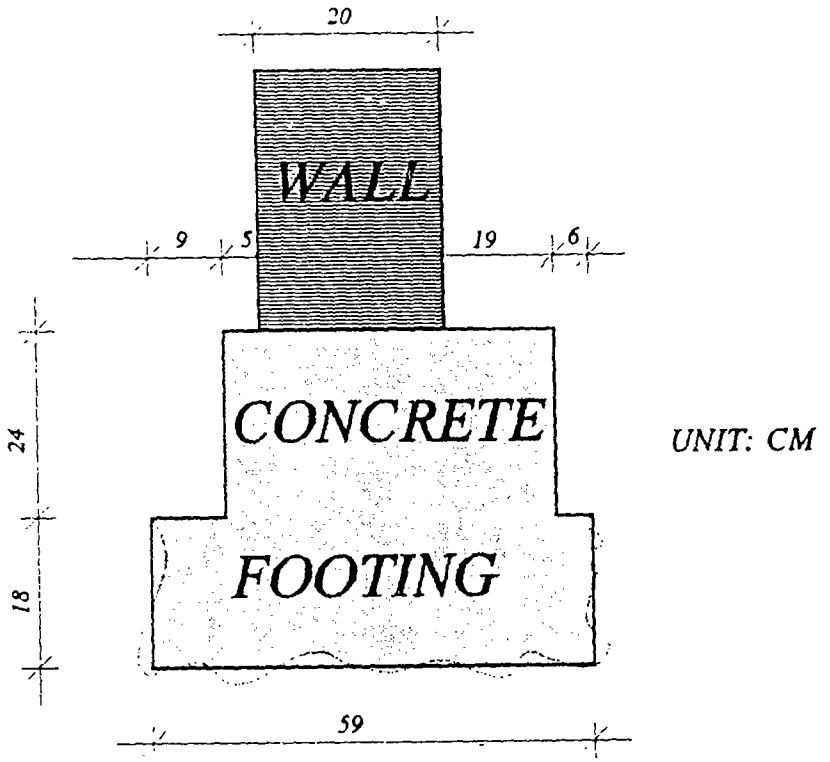


Figure 3 Typical Section through the Foundation

of 35 degrees, a cohesion value of 4.1 kPa and an elastic modulus of 44 MPa (Lawrence, 1992).

The longwall coal mine panel was 280 m wide, with the house being near the center of the panel. The house was oriented at 40.8 degrees with respect to the mining direction (east-west). Therefore, one diagonal of the house was roughly aligned with the mining direction. The panel was supercritical, meaning the subsidence transverse to the mining direction near the house was relatively uniform. The depth of mining was 160 m, with the coal seam being 2.3 m thick. Mining speed averaged 10.7 m/day and mining progressed from east to west. The mining resulted in a moving ground subsidence wave along the surface approximately 100 m wide perpendicular to the centerline of the mining. It was anticipated that the subsidence effects perpendicular to the mining direction would be minimal.

Measured Ground and Structure Movements

Ground Movements

Survey monuments along the access road to the house were monitored during the subsidence event. Figure 4 shows the subsidence profile along the road over a horizontal distance of about 700 m. The relative location of the house is also shown. Figure 4 indicates that a maximum subsidence of about 1.4 meters was measured in the location of the house.

Forty survey monuments were installed on the ground surface around the house. Vertical ground movements were measured with an optical level, and horizontal ground movements measured with an electronic total station survey instrument. The measured vertical ground movements (subsidence) adjacent to the

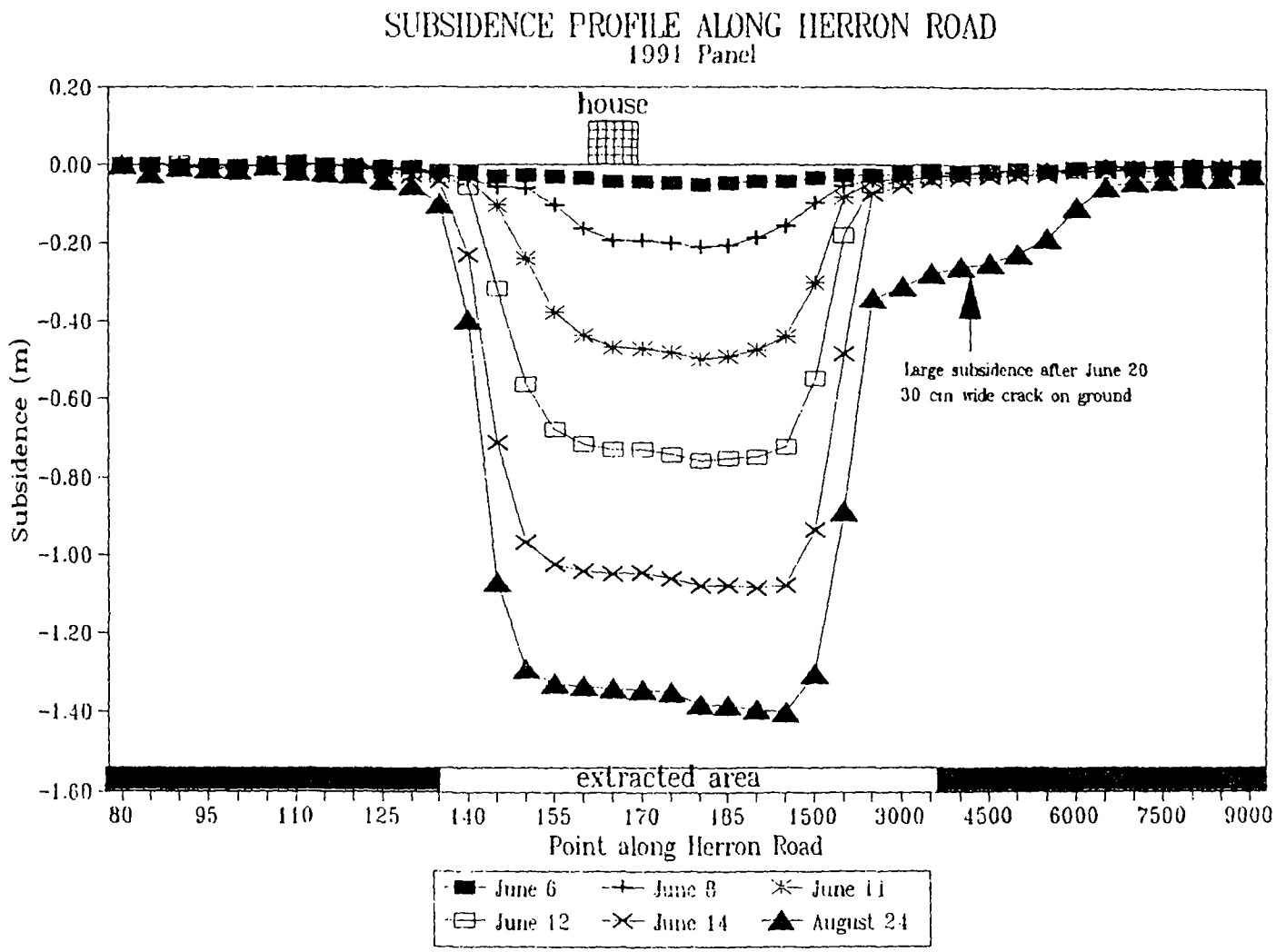


Figure 4 Subsidence Profile Perpendicular to Mining Direction along a 700 m Survey Line

house are presented in Figure 5. The horizontal ground movements adjacent to the house are shown in Figure 6.

Figure 5 depicts ground subsidence along the mining direction and ground deformation at different phases of the subsidence event. The large span in the east half of the curves corresponds to the rigid front porch. The curves indicate that subsidence started at the east end and moved from east to west. The subsidence profiles on June 15 and 16 indicate that the ground subsided considerably more at the east end than at west end, and suggest the east half of the ground was in tension. On June 17, the tension wave on the ground surface reached the western end of the structure, and the eastern half experienced near constant slope. The entire ground around the house was in compression as the subsidence profiles were slightly concave-upward. The ground returned to almost level by June 24, as the mining face moved well beyond the house. The maximum subsidence around the house was approximately 1.2 m, while the the maximum subsidence in the mining panel was approximately 1.4 m. As anticipated, the subsidence around house was caused primarily by the dynamic subsidence moving in the east- west direction; the subsidence perpendicular to the mining direction was negligible.

The horizontal ground movements in Figure 6 were magnified twenty times for clarity. The ground first moved eastward, then moved westward back to the initial position, with the movements in the north-south direction being negligible. Between June 8 and June 16, the ground at the east end moved the maximum amount (0.164 m) while the ground at the west end only moved about one-third of the maximum (0.046 m). This differential movement perpendicular to the mining direction was negligible.

The horizontal ground movements in Figure 6 were magnified twenty times for clarity. The ground first moved eastward, then moved westward back to the initial position, with the movements in the north-south direction being negligible. Between June 8 and June 16, the ground at the east end moved the maximum amount (0.164 m) while the ground at the west end only moved about one-third of the maximum (0.046 m). This differential movement of 0.118 m subjected the house to tension. On June 17, the east end remained almost in the same position while the west end moved to almost the maximum amount (0.139 mm). On June 18 and 19, the east half of the ground started to move back westward while the west part continued to move slightly eastward, subjecting the house to compression.

House Movements

The horizontal and vertical movements of the house were measured with a total station survey instrument by monitoring the position of seventy reflector monuments mounted on the structure (Lin 1993). In addition, a series of structural tilt measurements were obtained with tiltplates, and soil strain and lateral soil pressures were measured. The locations of the tilt plates, soil strain gages, and soil pressure gages are shown on a plan view of the house in Figure 7.

In general, the house experienced the same movement trends as the ground, being subjected to tension (convex-upward bending), compression (concave-upward bending), and returning to a final state with a small amount of residual tension. However, the magnitudes of deformation were different. The magnitude of the vertical movements of the house were very close to the ground subsidence at the adjacent location, but the horizontal movements of the house were typically smaller than the adjacent ground surface. This difference is a result of the interaction between ground, foundation and superstructure.

Due to the orientation of the house relative to the direction of mining, one diagonal of the house was roughly in the mining direction. The subsidence wave traveled from the northeast corner to the southwest corner. The average horizontal strain of the ground, foundation, wall and roof were calculated along the diagonal to demonstrate the transmission of horizontal strain from the ground to the structure. The results in Figure 8 show discrete tension and compression phases. The maximum tensile strain and compression strain are tabulated in Table 1. Table 1 indicates that the maximum tensile strain of the ground was about 1.6 times the maximum compressive strain of the ground. More tensile strain was

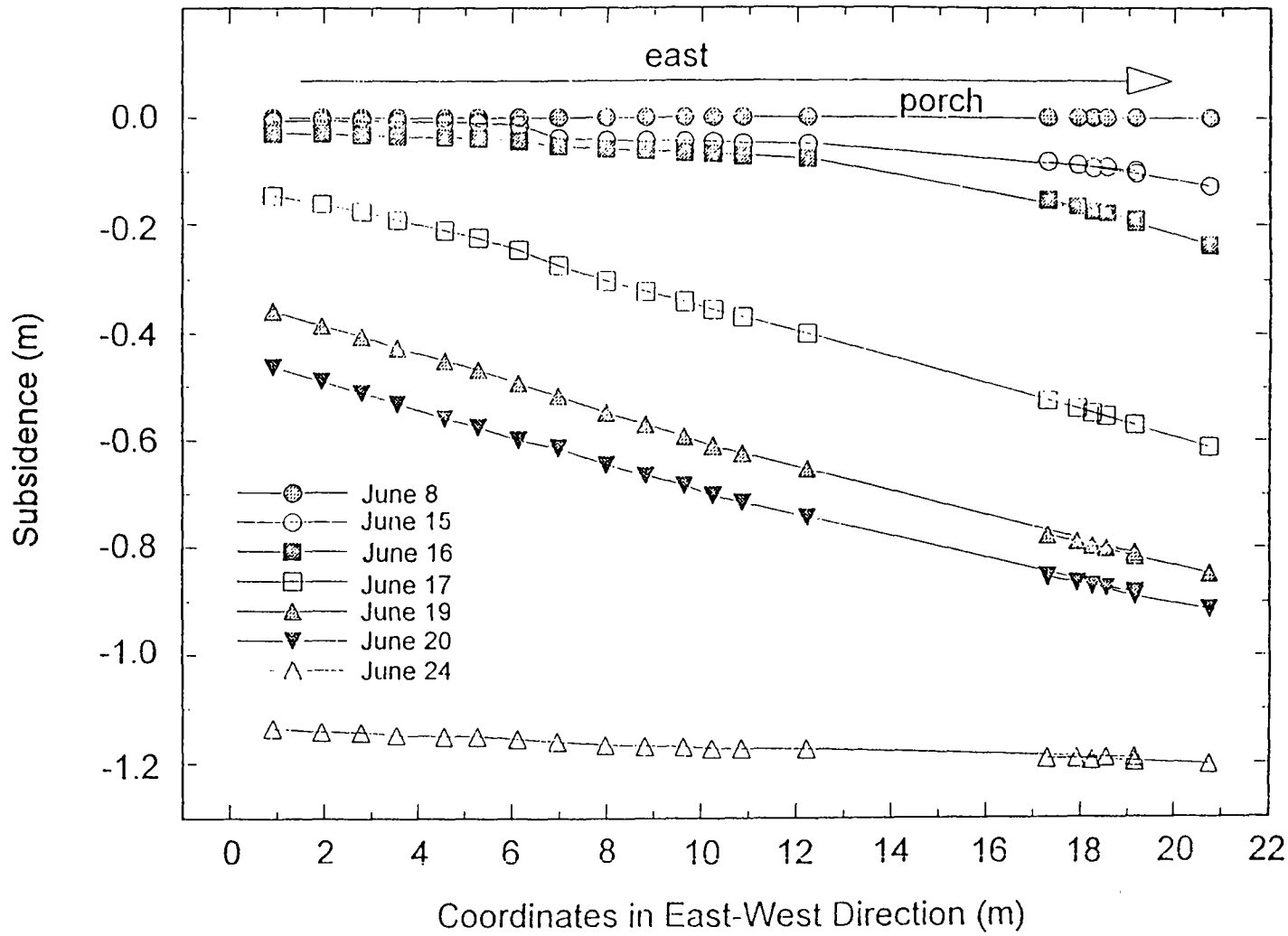


Figure 5 Subsidence of the Ground near the House

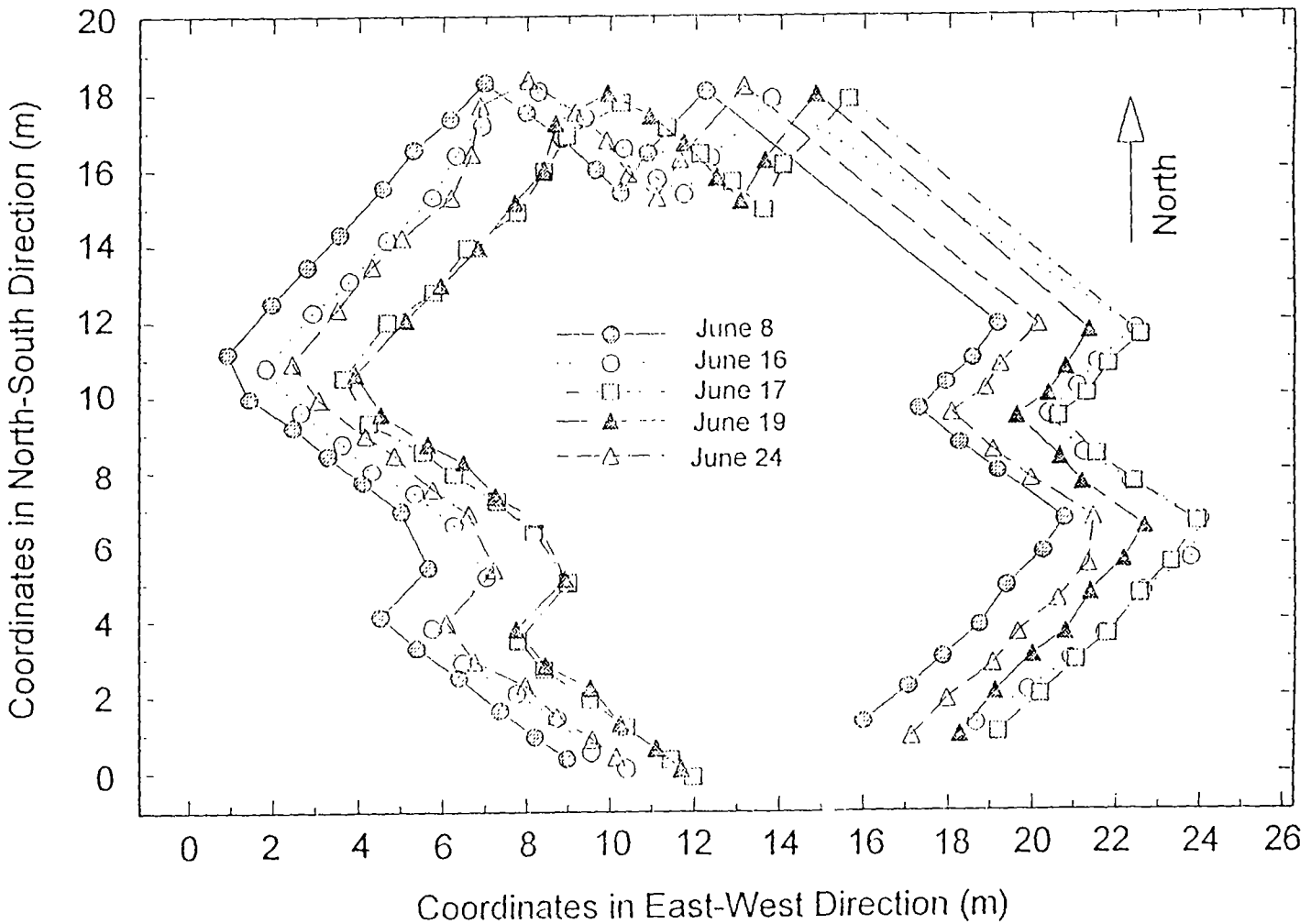


Figure 6 Horizontal movement of the Ground around the House

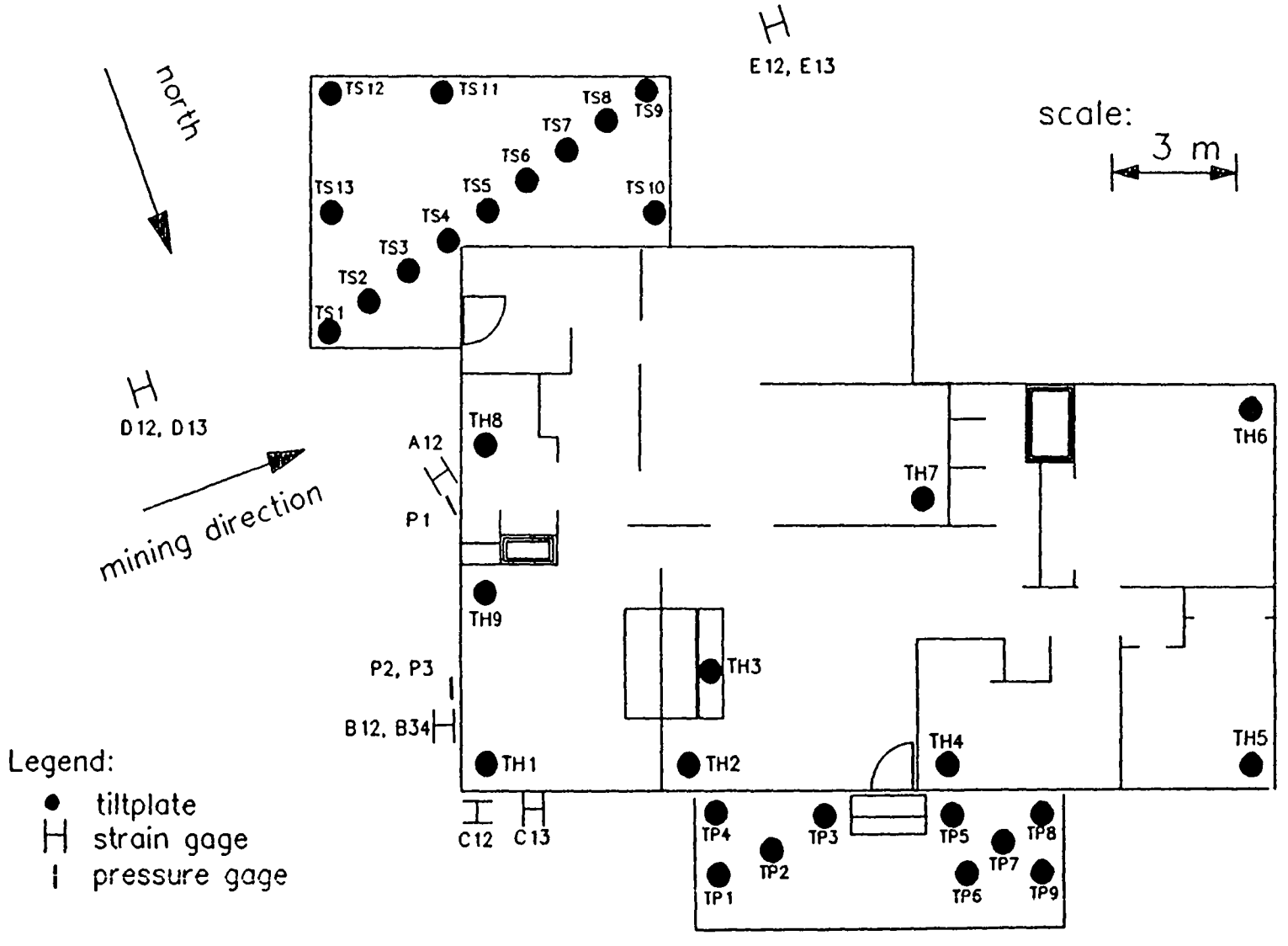


Figure 7 Instrument Locations on the House

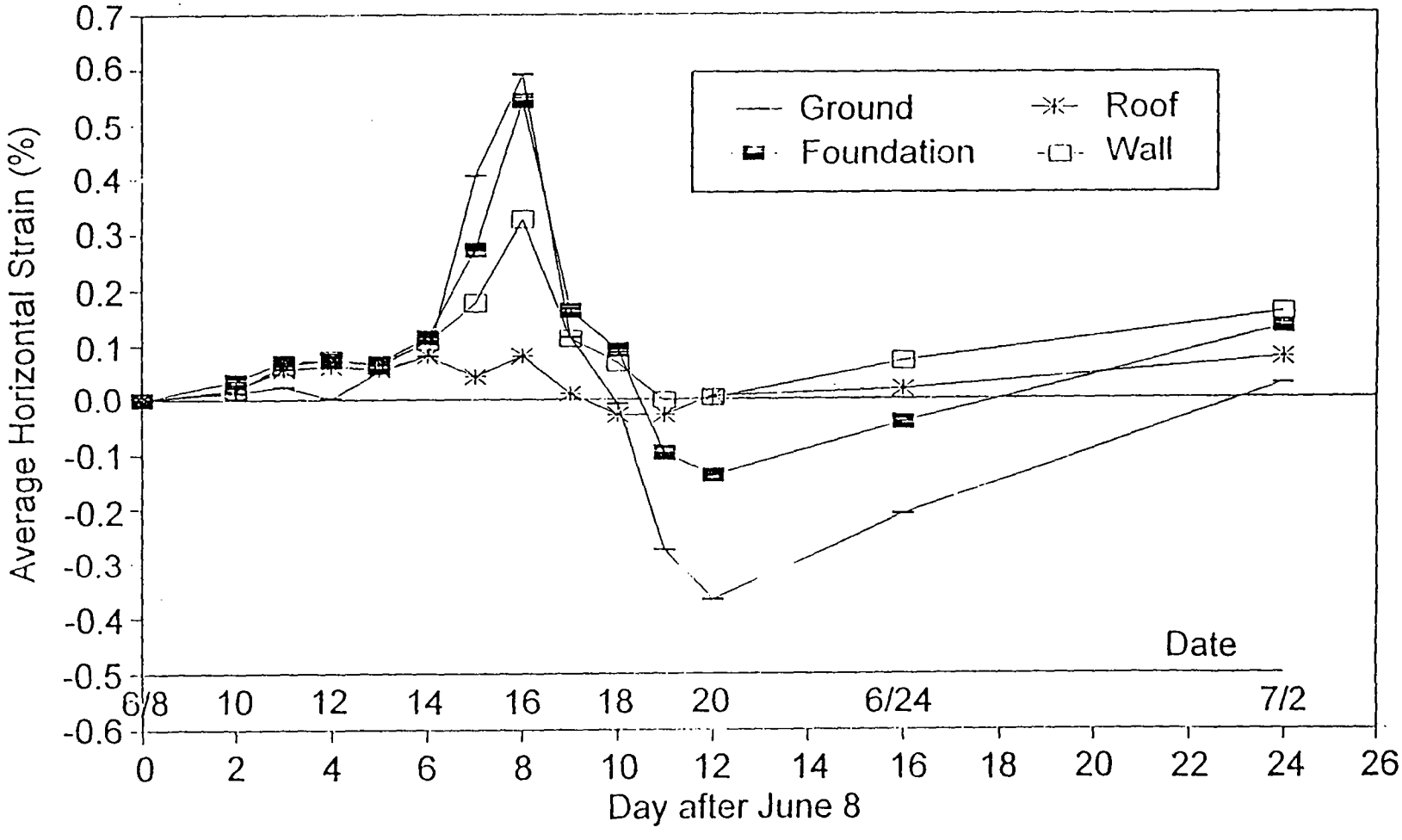


Figure 8 Average Horizontal Strain of the Ground, Foundation, Wall and Roof

transmitted to the superstructure components than compressive strain. The maximum tensile strain of the foundation was very close to the tensile strain of the ground, while the maximum compressive strain of the foundation was only about one-third of the maximum compressive strain of the ground. Both tensile and compressive strains decreased substantially from the foundation to the roof. Both the wall and the roof sustained virtually no compressive strain along the mining direction, which suggests there were relative movements between the foundation and the superstructure.

Since the longitudinal direction of the house was not aligned with the mining direction, the house was distorted during the passing of the subsidence wave. This distortion consisted of both in-plane and out-of-plane deformation of the structural members. The distortion of the house was measured by the angle formed by the north and west foundation walls as listed in Table 2. The wall angle was observed to increase and then decrease, and be left with an overall decrease in wall angle after subsidence.

Structural Tilt

Tiltplates were installed at numerous locations around the floor ofthe house. The tiltplates (SINCO 1985) permit the measurement of structural tilt in two orthogonal directions. Figure 9 illustrates the East-West tilt measured at two locations (tiltplates TH1 and TH6) over a period of 40 days. It is noted that the maximum tilt occurred about June 18 at the eastern most plate, TH1, while the maximum tilt occurred about 2 days later on the western tiltplate, TH6. There was very little tilt measured at either location in the North-South direction perpendicular to mining.

The gradient of the tilt with respect to horizontal distance is the curvature, which is useful for the comparison of structural damage. A linear regression of the tiltplate data was performed to determine the curvature along the diagonal of the structure. Figure 10 shows the curvature of the house floor over time. During the tension phase (about June 17) a convex-upward or negative curvature of 0.0003 m^{-1} was recorded. On June 21, a concave-upward or positive curvature of 0.00015 m^{-1} was recorded. A second negative curvature of about the same magnitude as the first was recorded on July 2.

Table 1 Measured Ground and Structural Strain

	Maximum Tensile Strain	Maximum Compressive Strain
Ground	0.59%	0.37%
Foundation	0.54%	0.14%
Wall	0.31%	<0.1%
Roof	<0.1%	<0.1%

Table 2. Measured Distortion of the House

Distortion of House as Measured by Angle of the north and west foundation wall					
Date	June 8	June 16	June 17	June 18	June 20
Angle (°)	90.01	90.22	90.02	89.93	89.28

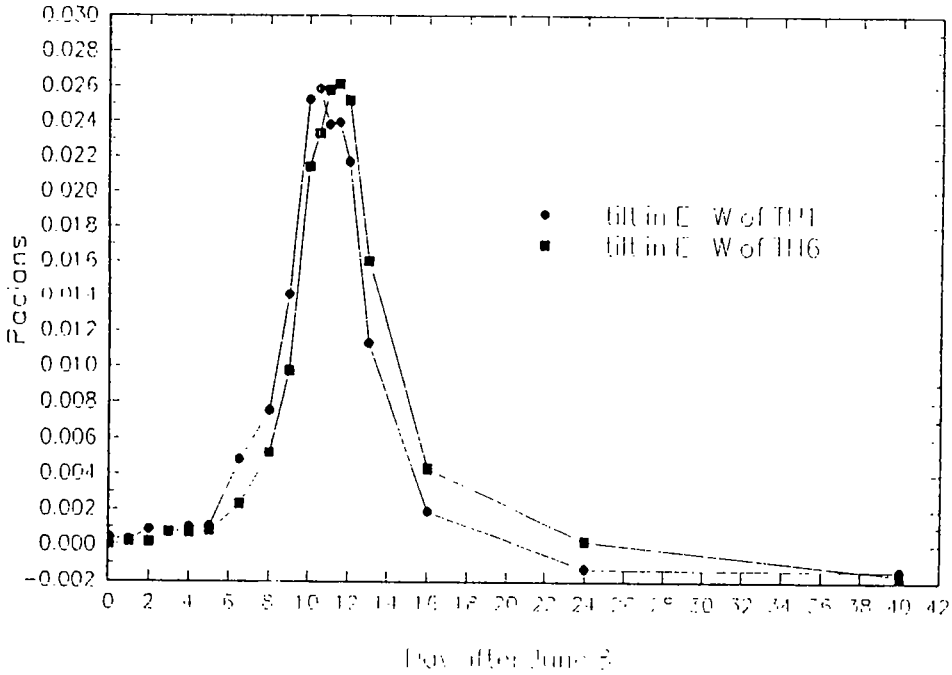


Figure 9 Measured Tilt of the East and West Corners of the House

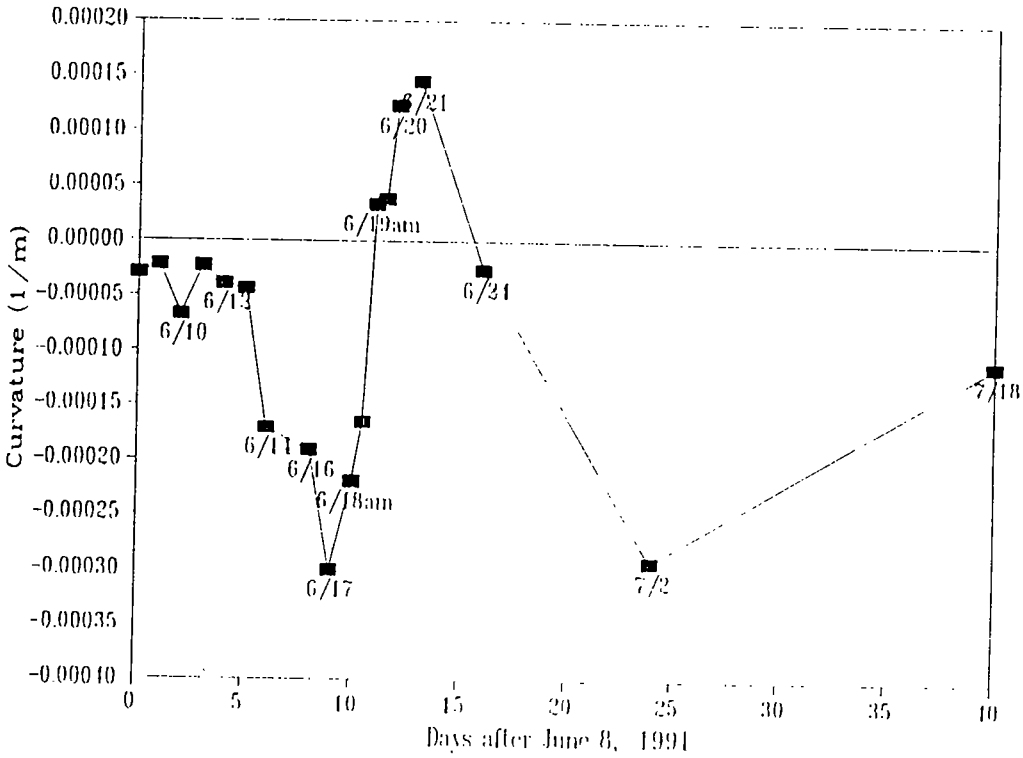


Figure 10 Calculated Curvatures along the House Diagonal

Soil Strain

Bison soil strain gages (Selig 1975; Bison Instruments 1989) were installed at several locations to record the soil strain. The soil strain in the horizontal direction is shown in Figure 11 for four locations. Gages A12, located next to the structure, and gage D12 located away from the structure, were oriented in the direction of mining. Gages B34 and C12 were located adjacent to the structure but oriented relative to the structure as shown in Figure 8. Gage A12 was installed at a depth of 0.6 m, while B34, C12, and D12 were installed at a depth of 0.4 m. The overall soil strain response is similar to the average soil strain over the length of the structure shown in Figure 7, with clear periods of tension (positive strain), followed by compression (negative strain). However, the strain magnitudes are different, and the soil strain gages located adjacent to the foundation were left with residual compressive strain after the subsidence wave passed. The residual compressive strain may reflect the plastic deformation that takes place near the footing during the compression phase. This plastic deformation of the soil is thought to cause the residual tension in the footing after the subsidence is complete. The gage length of the soil strain gage measurements was typically about 100 mm, which is much smaller than the gage length of nearly 20 m used to calculate the soil strain in Figure 7. Therefore, differences in the measured strains are not unexpected.

Soil Pressure

Soil pressure gages were installed to record the lateral earth pressures during the subsidence event. Lateral earth pressures can cause significant damage, especially to structures with basement walls. The soil pressure gages were located between the foundation wall and the surrounding soil. As shown in Figure 8, the pressure gages were oriented with the Bison soil strain gages. The recorded change in soil pressure during the subsidence event is shown in Figure 12. Although the magnitude of the compressive pressure differs from gage to gage, the trend of the pressure variation is similar. A slight negative change or reduction in lateral soil pressure was recorded early in the subsidence event, as the tension wave approached the gage (structure). About June 18, a compressive peak in pressure was recorded corresponding the arrival of the compression wave. This peak was followed by a slight tension period before the pressure returned to nearly that which existed prior to subsidence. The magnitude of the peak compressive stress recorded in the three gages ranged from about 10 kPa to nearly 25 kPa. This is likely a result of differences in the manner in which the gages were installed or seated. In all cases however, the measured stresses were well below the theoretical passive earth pressure which is the maximum pressure that the soil could resist assuming that the wall did not fail. For example, the 25 kPa pressure recorded at gage P2 which was located at a depth of 0.55 m, can be compared to a Rankine passive earth pressure of 35 kPa. This suggests that even though the 25 kPa soil pressure is significant, it is well below the theoretical upper limit.

Since soil pressure gage P1 and soil strain gage A12 were installed at nearly the same location, the results can be combined to produce a lateral stress-strain response curve for the soil as shown in Figure 13. In this curve, the initial tensile strain and pressure are shown as positive, with the subsequent compression loading shown as negative. These instruments recorded much larger strain and stress during the compression phase than during the tension phase.

Structural Damage

As a result of the ground movements and deformation, the house suffered two types of structural damage: cracking of the concrete footings and block walls, and distortion of the superstructure. The development of cracks in the block wall was recorded during the subsidence. The cracks on the concrete footings were mapped after the subsidence event. The superstructure damage is quantitatively described in terms of strain, tilt and angular distortion.

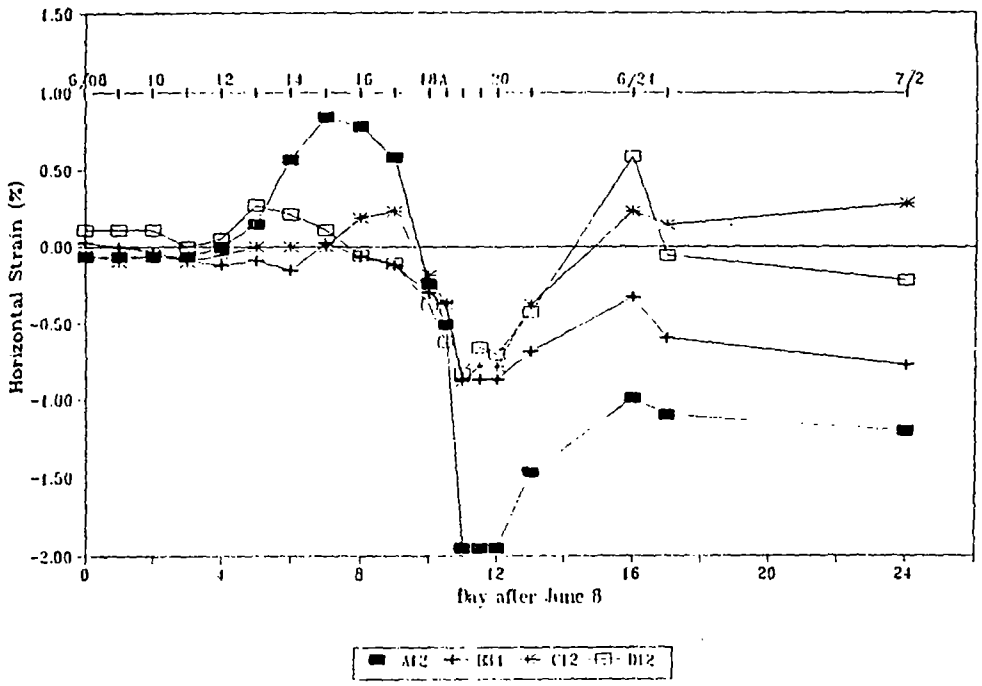


Figure 11 Measured Soil Strain over Time

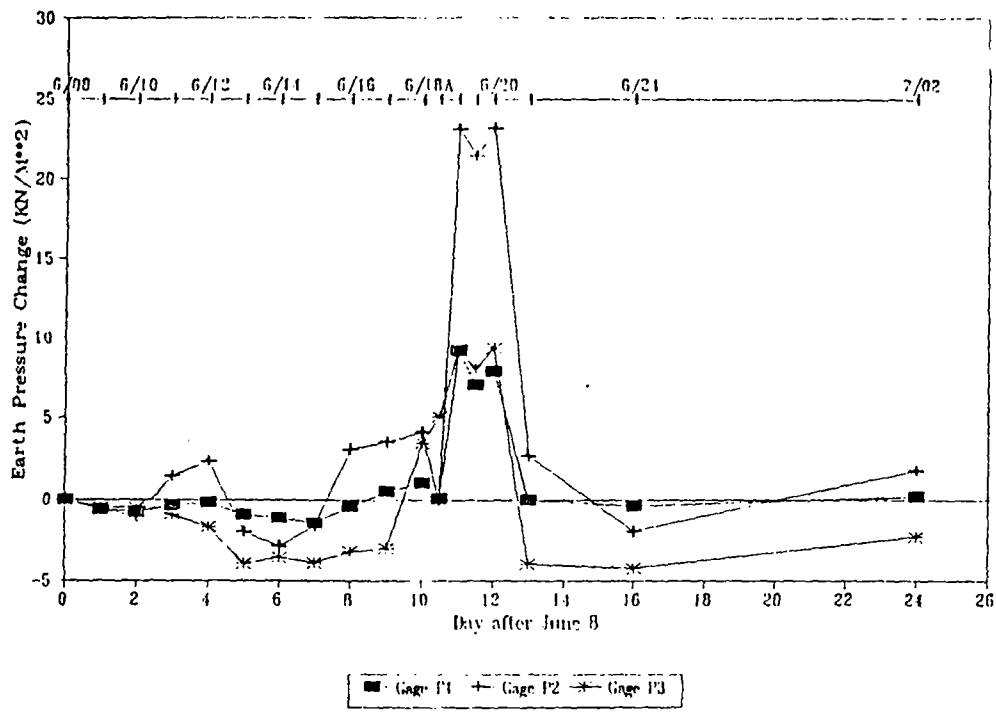


Figure 12 Change in Lateral Soil Pressure over Time

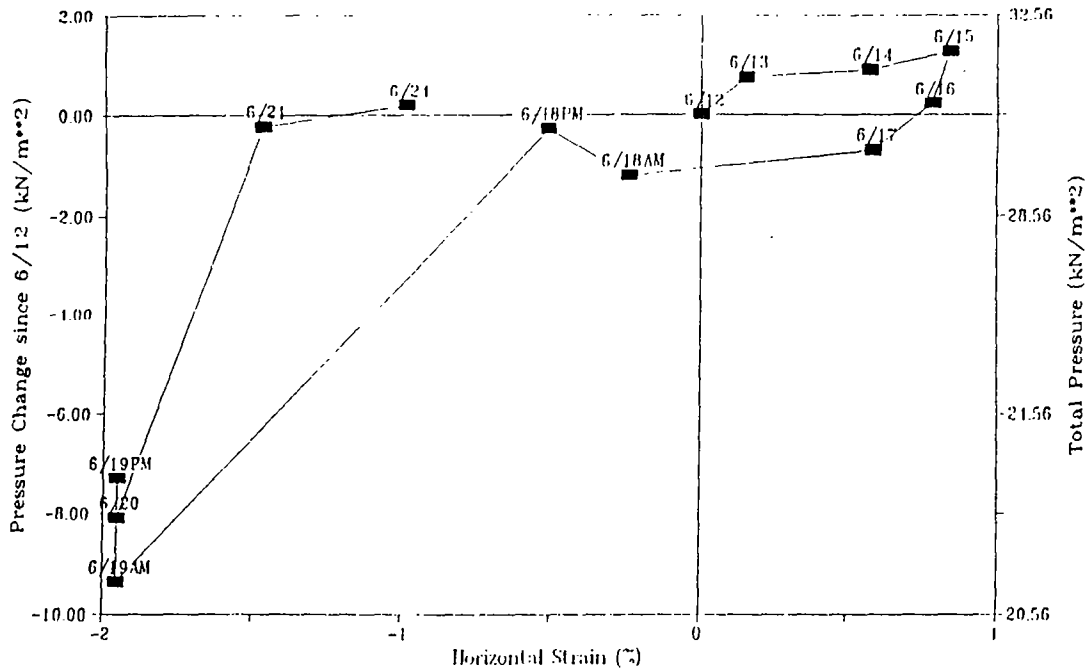


Figure 13 Soil Stress-Strain response during Subsidence

Figures 14 and 15 show the location and residual crack width in the concrete footings and block walls. The residual crack width was considerably smaller than the maximum crack openings developed during the tension phase of the subsidence event, and ranged from 3 mm to 10 mm. The cracks typically extended up through the masonry wall, generally in a stair step fashion and slightly increasing in width. No cracks were observed in the east side footing, but a large portion of the footing was covered by a utility box. Cracks were first found on the east side of the masonry wall on June 15, and by late afternoon six cracks were found in the masonry wall.

The development of the crack on the east section of the front wall, Figure 14, can be used to illustrate the effects of subsidence on the house. This crack started on June 15 and by the morning of June 16, the blocks on the sides of the crack moved apart about 25 mm both in the direction of the wall plane and perpendicular to the wall plane. On the morning of June 17, the cracks closed to about 10 mm in both the in-plane and out-of-plane directions. The crack decreased to about 2 mm in both directions on June 18. Subsequently, the crack reopened in the wall plane and had a residual width of about 10 mm. This process of tension, compression and a second tension has been described previously (Geddes 1977) and was also observed in a nearby series of test foundations (Lin et al. 1994). Due to the high compressive stiffness of the structure and yielding of the soil, the full compressive deformations from the soil
are not induced into the structure. As the ground returned to a state of zero tension, it induced a second tension in the structure.

Additional damage to the masonry wall was observed where the center beam rested on the wall. During the compressive phase of the subsidence, the beam pushed two blocks out about 10 mm on both the east and west sides, causing some local crushing of the block. This is indicative of the relative movements between the foundation and superstructure.

Cracks were also observed on gypsum wallboard inside the house. These cracks typically developed at an about 45° angle from the corner of openings such as windows and doors. These cracks were

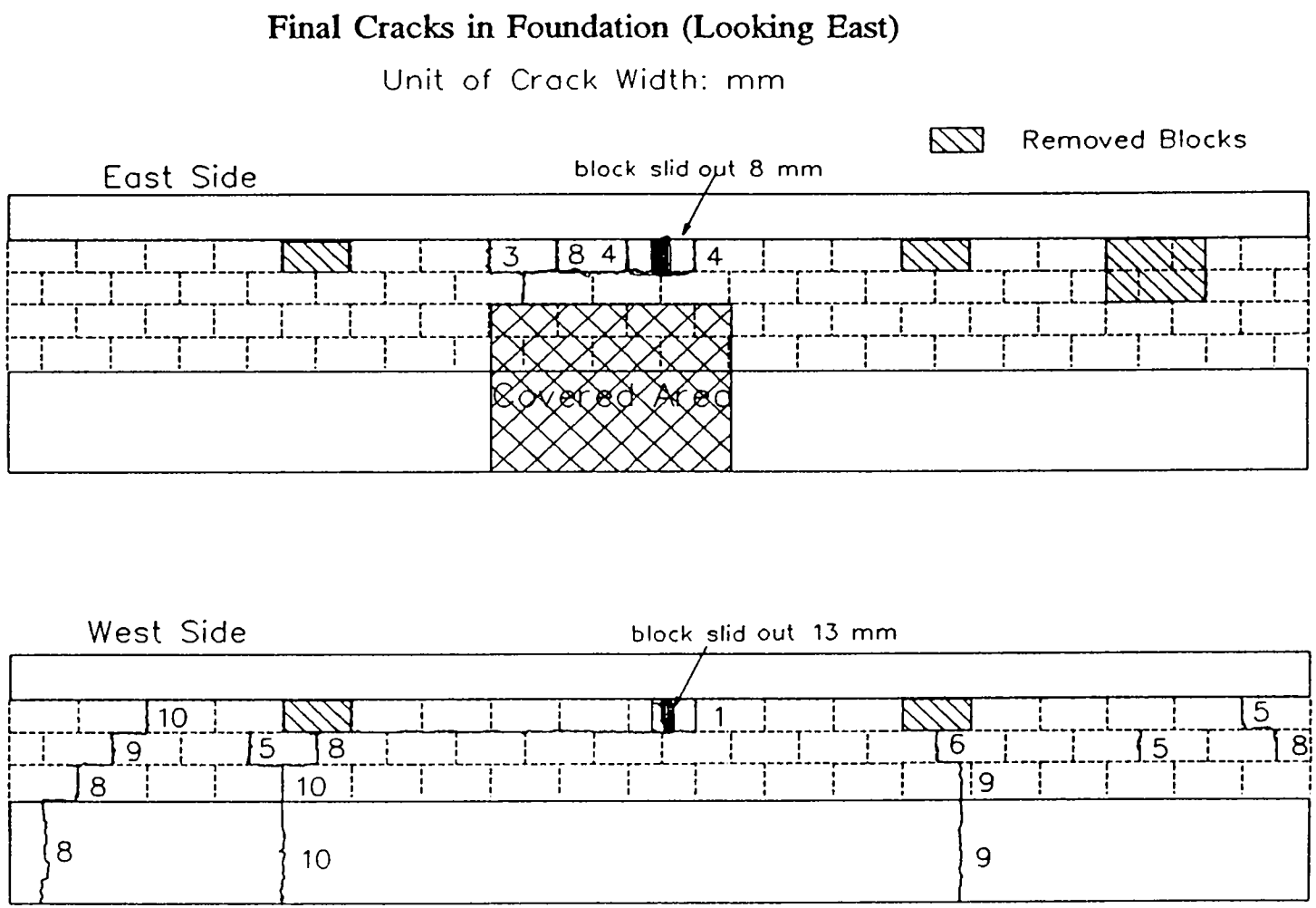
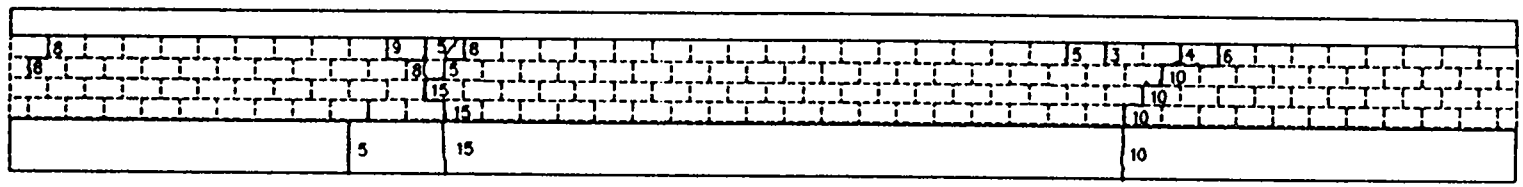


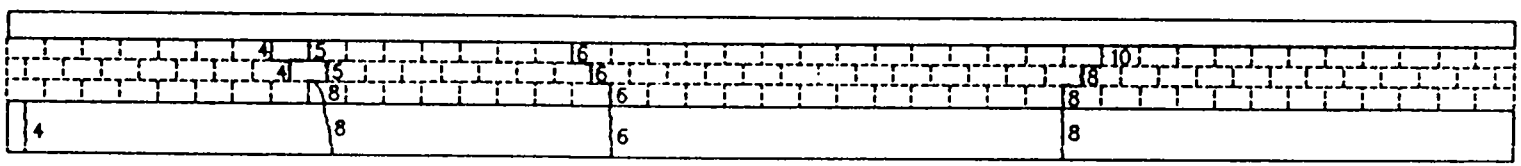
Figure 14 Location and Widths of Final Cracks (Front and Rear of House)

Final Cracks in Foundation (Looking North)

Unit of Crack Width: mm



Front Side



Rear Side

Figure 15 Location and Widths of Final Cracks (East and West Sides of House)

typically very small in width and extended less than 300 mm.

It appears that the superstructure damage was primarily due to differential vertical movements, and secondarily due to horizontal movements. The structural deformation and damage were quantitatively described in terms of angular distortion. Angular distortion is the change of slope between two points divided by the horizontal distance between the two points. Figure 16 shows the angular distortion of the ground, foundation wall, and house wall immediately above the foundation wall. The angular distortion decreases moving from the ground up to the foundation, and continues decreasing with elevation into the structure. The change in angular distortion is indicative of interaction between ground, foundation and the superstructure.

The calculated angular distortions are relatively consistent with the limits of angular distortion suggested by others corresponding to certain levels of structural damage. Bjerrum (1963) suggested an angular distortion of 1/500 as a safe limit for no cracking. Cracking was observed in the house foundation when this limit was exceeded. Marino (1985) suggested that a house with a crawl space that had an angular distortion exceeding 1/208 would require foundation repair. Once the house foundation cracked, it essentially followed the ground movements, resulting in high angular distortion of much greater than 1/208 and complete failure. Wahls (1994) indicates that cracking for plywood or fiberboard on wood frames occurs at an angular distortion between 1/60 and 1/170. Angular distortions greater than 1/170 were observed during the investigation, but only minimal cracking was noticed. However, serviceability problems such as inoperable doors were experienced.

Conclusions and Recommendations

The foundation of the monitored house cracked in numerous places during the mining subsidence. Therefore, extensive repair and replacement work would be required before the house could be returned to normal use. Despite the extensive damage to the foundation, the superstructure of the house suffered minimal damage. The superstructure damage was primarily serviceability issues such as sticking doors and

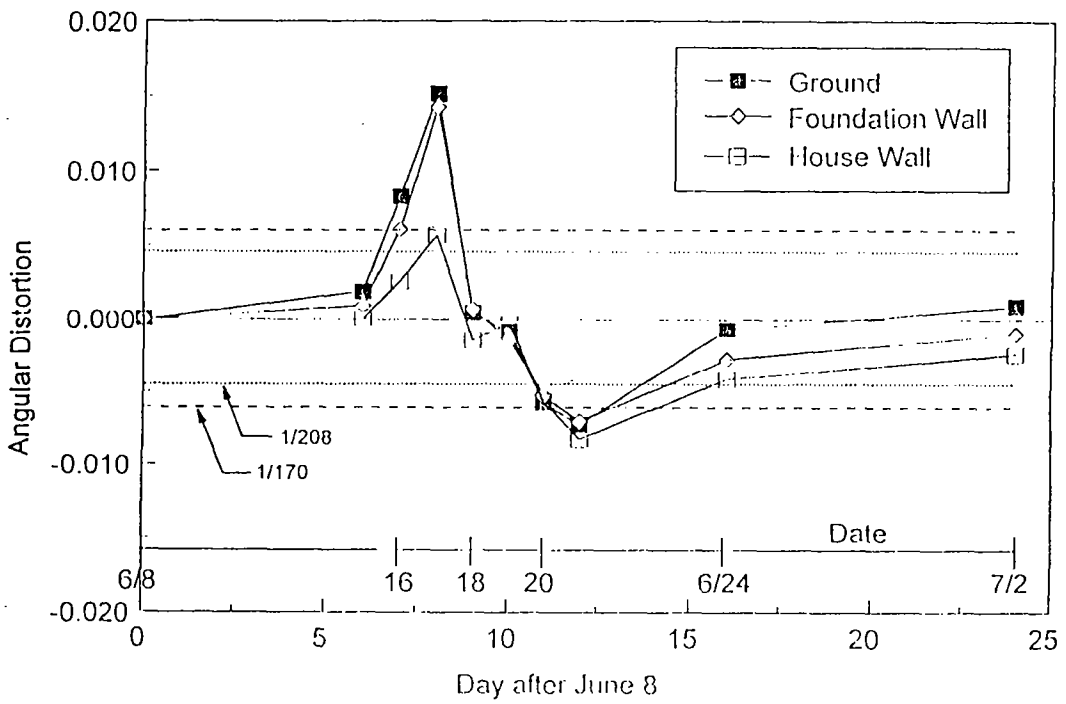


Figure 16 Angular Distortion of Ground, Foundation Wall, and House Wall

aesthetic problems such as cracking of the interior gypsum-board wall. The wall deformations were mainly in-plane shear and out-of-plane movements resulting from twisting of the house.

The house approximately followed the nearby ground subsidence in the vertical direction. Measurements of strain over the length of the house suggest that the foundation experienced tensile strains close to those in the ground during tensile period, resulting in cracking of the foundation and wall. However, during the compression period, the foundation cracks closed and the house frame distorted, resulting in foundation strain that was only about one-third of the strain in the ground. The foundations and walls experienced both in-plane and out-of-plane deformation but the walls appeared more vulnerable to the out-of-plane deformation. The deformation in the foundation was transferred to the superstructure primarily through the mechanism of shearing, and its magnitude decreased from foundation to roof.

The minimal damage to the house superstructure can be attributed to the lack of any significant connection between the superstructure and the foundation. Although this lack of connectivity between the structure and foundation tended to reduce the damage, this cannot be recommended as an damage mitigation technique. Good construction practice dictates anchoring the structure to the foundation to resist lateral loads due to wind and earthquakes, even if the anticipated lateral loads are small.

The purpose of a foundation is to transfer structural load to the ground and to minimize settlements. In an area subject to mining-induced settlements, there are several other considerations during foundation design, most at very little additional construction cost. Damage mitigation schemes for mining subsidence or other similar types of ground movements should be directed towards strengthening the foundation and improving the connection between the foundation and the superstructure. Reinforcing steel and post-tensioning were proven to be effective in minimizing foundation cracking or controlling crack widths in a series of test foundations constructed over a similar mining panel (Lin et al. 1995). By reinforcing and tieing the basement/crawl space walls and possibly the superstructure walls to the foundation so that they act compositely, a stiff and strong beam is formed to resist the ground movements. Additional research is being conducted to further develop this concept.

Mine subsidence results in a dynamic subsidence wave that moves along the ground surface. This wave produces three phases deformation for a structure on the surface: a) convex-upward bending or tension phase, b) concave-upward bending or compression phase, and finally c) a second tension or return to near horizontal configuration. In general, the ground returns to a near zero state of stress in the third phase, but the structure may be left with some residual tension.

When analyzing structures for subsidence deformation, it is important to keep in mind that due to soil-structure interaction effects, the ground deformations in the vicinity of the structure are smaller than those away from the structure. Therefore, a structure need not resist the full magnitude of ground deformation measured during subsidence event. The degree of soil-structure interaction depends upon the relative stiffness of the soil and foundation. The vertical deformations are believed to cause the structural damage, with horizontal deformations serving to open up any cracks that form due the vertical deformation.

Acknowledgments

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USING GIS AND NUMERICAL MODELING TO ASSESS SUBSIDENCE OVER ABANDONED MINES

by

Kevin M. O'Connor, John A. Siekmeier, and Jan Stache

Abstract. The U. S. Bureau of Mines (USBM) has been developing techniques to assess surface subsidence over abandoned coal mines. One technique has involved estimation of stress acting on every pillar of an abandoned mine then comparing it to pillar strength and floor bearing capacity. This required computations for several thousand pillars for one mine. Mine maps are digitized and saved as a computer drawing file. Then the tributary area loading each pillar was determined graphically and outlined. Geographic Information System (GIS) software was used to compute the ratio of pillar area to tributary area for each pillar and then divide the average overburden stress by this ratio to compute an estimated pillar stress. Numerical modeling was then used to analyze a two-dimensional cross section of the overburden and mine, and provide an independent estimate of stresses. Based on published data for floor bearing capacity and pillar load capacity, GIS was used to perform a mine-wide classification of pillars according to stress level. An example of this analysis and classification is presented in this paper for an abandoned coal mine in the Illinois Basin. The mine had been operated in the Herrin No. 6 Seam at a depth of 60 m with an average overburden stress of 1.4 MPa. It was found that pillars with estimated stresses greater than 5 MPa correlated with historical subsidence events. Due to the greater detail considered in this approach, it provides a fundamental basis for the assessment of subsidence risk since it incorporates the geometry of mine pillars and entries as well as the ultimate strength of the pillars and floor.

Additional Key Words: pillars, stress, animation, CAD

Introduction

Coal mining was active in Collinsville, Illinois from 1870 to 1964, and the area is underlain by a network of mine openings. Support for the overlying rock is provided by stable pillars and blocks of coal as well as those that may be failing or possibly punching into the underlying claystone. As a result of failures within the abandoned mines, there have been localized mine subsidence occurrences throughout the city. Movement of the overlying rock, and ultimately the surface, has subjected structures, streets, and utilities to strains and stresses that have caused damage. Occurrences of subsidence and damage have been regarded as random events. Current research indicates that this may not be the case and rational assessment of subsidence potential may be possible.

Subsidence characterization over abandoned room-and-pillar mines remains relatively undeveloped due to a lack of understanding about time-dependent rock mass behavior at mine level and within the overburden. The effectiveness of early warning technology inherently requires that the timing and extent of a subsidence event can be anticipated. In the case of important structures located over abandoned mines, it is not adequate to say that subsidence may occur. A means must be provided to indicate if strata movements are occurring (and the rate at which they are occurring)

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[1]Paper presented at the 13th Annual National Meeting of the American Society for Surface Mining and Reclamation, Knoxville, Tennessee, May 18-23,1996.

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beneath these structures so that appropriate measures can be taken to mitigate damage. One component of research being conducted by the USBM has involved monitoring of subsurface movements which are precursors of surface subsidence. This paper concentrates on another component involving the use of historical subsidence data and mine maps to identify areas over abandoned mines with a high likelihood of subsidence occurrence.

The USBM in cooperation with the Illinois Mine Subsidence Insurance Fund (IMSIF), Illinois Abandoned Mined Land Reclamation Council (AMLRC), and Collinsville Community Unit School District No. 10 (CUSD #10), has implemented these techniques to allow proactive land use planning over abandoned mines. This joint project was initiated because of subsidence damage to one school, the Dorris Elementary School (figure 1 and figure 2). It will be replaced by a new building and two sites are being considered. One site is an athletic field adjacent to the existing school. The alternative site is an athletic field adjacent to the Lincoln Elementary School (figure 2). Both sites are undermined so it is a situation in which the school district must assess the relative risk of future subsidence.

The USBM conducted a program of site characterization involving core drilling, downhole camera inspection of mine conditions, monitoring of subsurface movements, and data analysis. This paper focuses on efforts to estimate pillar stresses and correlate occurrences of historical subsidence with pillars that are highly stressed. This effort was undertaken using two independent approaches. The first approach involved GIS analysis of the geometry of pillars and entries to estimate stresses assuming tributary loading on pillars (Whittaker and Reddish, 1989). The second approach involved numerical modeling of a two-dimensional cross section of the abandoned mine and overburden to estimate the distribution of stress.

Geometric and Subsidence Data Collection

The project sites are located in Collinsville (figures 1 and 2) which has been extensively undermined. Historically, the old high school and Dorris Elementary School have been damaged by subsidence. There has been no evidence of subsidence at the Lincoln Elementary School.

Available data which were collected for this projected included: 1) U.S. Geological Survey (USGS) 7.5-minute Collinsville quadrangle map, 2) mine maps, 3) subsidence survey data, 4) drilling logs, 5) structural damage reports, and 6) Madison County plat map and orthophoto overlays. Maps were assembled and digitized for the following mines (ISGS, 1991):
Bullock Mine, operated 1899 - 1946;
Heinz Bluff Mine, operated 1883 - 1907;
Huntley Bluff Mine, operated 1886 - 1887;
Lumaghi Mine No. 1, operated 1883 - 1903;
Donk Bros. Mine No. 1, operated 1900 -1922.

Surface subsidence has been monitored by the IMSIF and the AMLRC (Gibson and Schottel, 1990) in response to occurrences of subsidence damage. Collectively, ninety-eight benchmarks in the area of Dorris School and the old high school have been monitored at various times during the period from February 1984 to August 1994. This monitoring has been performed in response to claims of subsidence damage so it was started after "events" had affected the ground surface. Furthermore, data were collected as a series of nets with local reference systems that were not tied to a common origin.

All surface feature and mine maps were digitized and assembled as a drawing file using computer-aided drafting (CAD) software. This allowed all spatial data to be assembled, viewed, and plotted with a common origin, reference system, and graphical scale. With the survey monuments now tied to a common reference, it was possible to assign global coordinates to each point. A drawing of the Dorris Elementary School with surveyed benchmark locations was obtained from the AMLRC. Drawings and survey elevation measurements of monitored benchmarks and structures in the surrounding area were obtained from the IMSIF. The mine maps and drawings were scanned using DeskScan II, then digitized using AutoCAD and assigned to different drawing layers. The historical survey data from both AMLRC and IMSIF were entered into a spreadsheet.

The digitized base map was created using section corners and road intersections as control points for positional accuracy of the mine and subdivision drawing layers. This made it possible to register the map with USGS 7.5 minute Collinsville quadrangle digital line graph (DLG) features and the Universal Transverse Mercator (UTM) projection coordinate system. Road alignments, property boundaries, and the addition of cultural features (baseball field, power poles, fences) for the two school sites were entered from overlay maps obtained

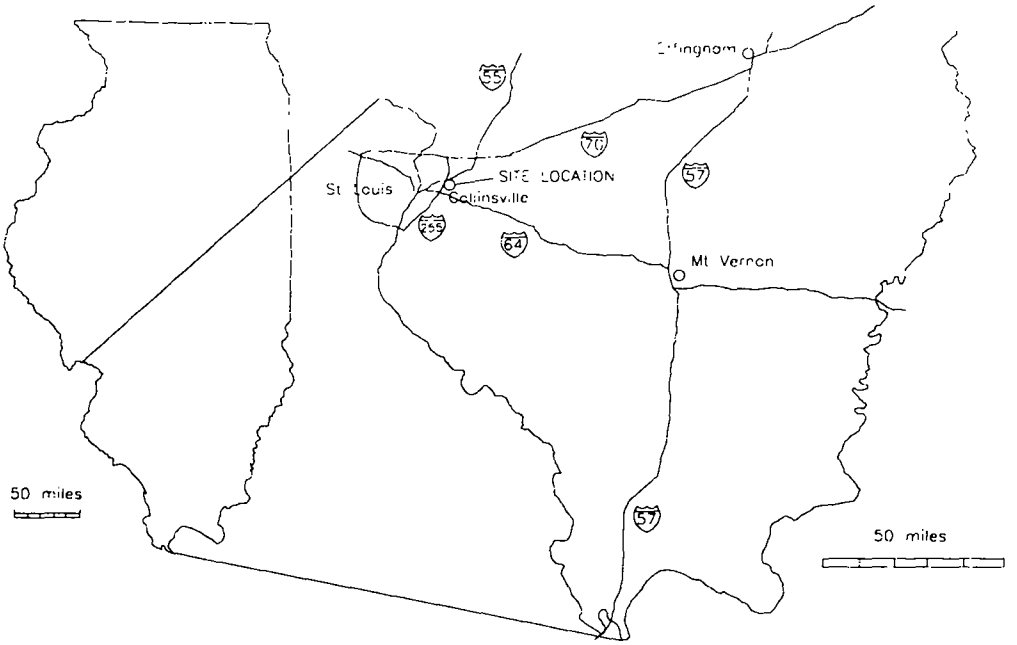

Figure 1.-Site location in Illinois.

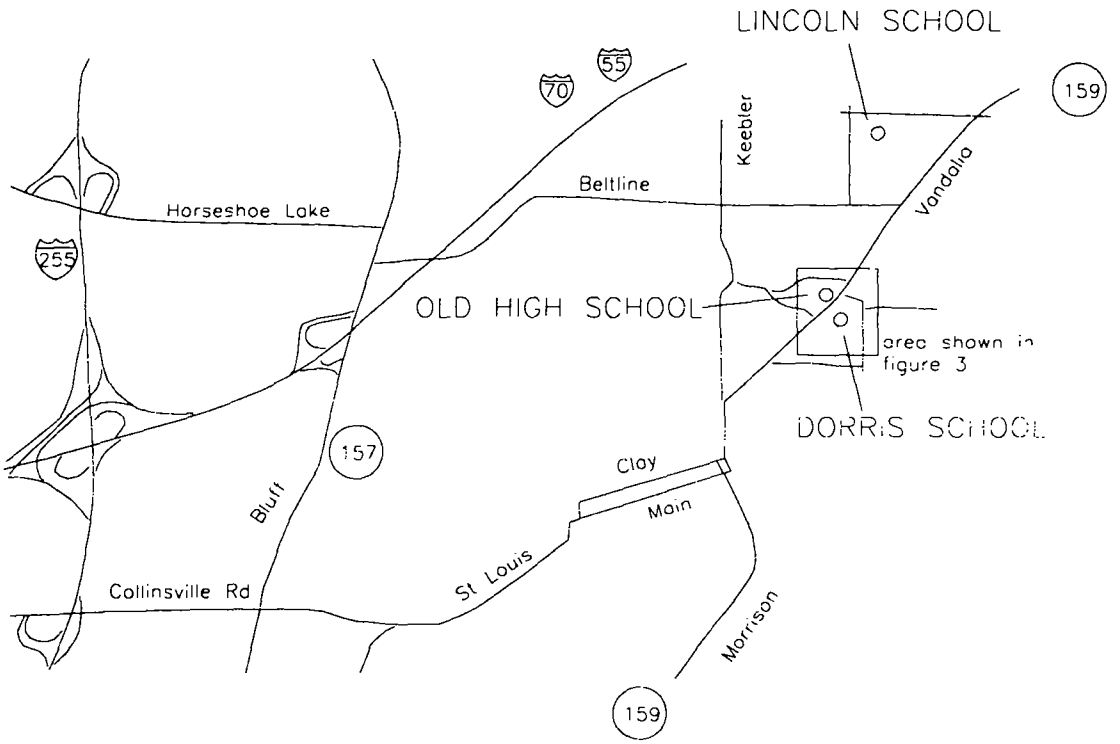

Figure 2.-Site location in Collinsville, Illinois.

from the Madison County Maps and Plats Department. These maps display property boundaries as an overlay with aerial photography at a scale of 1:1200 (1 in.= 100 ft).

Map scale errors may have been introduced or compounded as a result of scanning or digitizing copies of the maps used for data entry. USBM researchers recognize that accuracy of the base maps is an approximation and therefore use the maps to assess general tendency correlations between structural damage and ground surface curvatures (Triplett et al, 1992), and as a tool for display of spatial analyses. However, the elevations of subsidence monitoring points used to build the research database were measured using modern survey techniques with a precision of ±1.5 mm (±0.005 ft).

Initial survey data were used to establish baseline elevations and subsequent surveys were used to calculate vertical displacement. Subsidence contours for each date were generated using a statistical technique known as kriging available within SURFER for Windows. The subsidence contour lines were then imported into AutoCAD and contours for each date were placed on separate layers. Figure 3 displays the last date of survey measurements available overlaid on the mine map. The first contour represents 0.05 m (0.16 ft) of subsidence and the contour interval is 0.025 m (0.08 ft).

Analysis of Subsidence Occurrences

By selectively activating different drawing layers, it was possible to generate a series of slides showing subsidence contours for each date that measurements were made overlaid with the mine geometry. The slides were compiled as an animation to enhance the changes in subsidence contours as a function of time. The animations allowed an assessment of subsidence activity and its relation to underlying mine geometry. This was critical since mine level changes are the cause of subsidence and time is a major factor when considering subsidence over abandoned mines. With this visual aid it was possible to qualitatively assess: 1) the rate of surface subsidence, 2) the rate of lateral progression, and 3) the correlation with mine geometry.

Based on the computed subsidence contours, it was hypothesized that the extent of pillars associated with subsidence for each date must be within the limits of the contours. This hypothesis is based on experience (Hunt, 1980) and the conventionally accepted geometrical relationship between subsurface openings and surface subsidence (Whittaker and Reddish, 1989; Kratzch, 1983). The expansion of surface subsidence bowls can be attributed to increasing mine level span widths. The plot in figure 3 illustrates how the bowl width is controlled by the position of pillars at mine level. However, the bowl depth (i.e., maximum subsidence) is controlled by several factors including the magnitude of convergence at mine level and the overburden stratigraphy.

Estimation of Pillar Stresses

In figure 3, it can be seen that the length and width of pillars are on the order of 20 m (64 ft) by 6 m (18 ft) beneath the Dorris School (center bowl). By contrast, they are 17 m (57 ft) by 13 m (43 ft) beneath the old high school (northernmost bowl). Also note the entry widths are different so the pillar stresses must be different. Assume that each pillar supports a column of overburden extending from the mine to the surface and that the column boundaries are the midpoint of entries between adjacent pillars (table 1) (Whittaker and Reddish, 1989). The weight of this column can be computed by assuming that the overburden has a unit weight of 20.3 KN/m^3 (144 lbs/ft^3).

It is estimated that the pillars beneath Dorris were subjected to a stress of 5.8 MPa (840 psi) compared with 2.4 MPa (345 psi) beneath the high school. The capacity of pillars to support this stress is not only controlled by the pillar strength (e.g., 10 MPa) but also floor bearing capacity (e.g., 5 MPa) (Pula et al, 1990). In the case of pillar crushing, convergence could occur rapidly at mine level with rapid subsidence at the surface. In the case of pillars punching into the floor, convergence may be long term. In summary, the difference in maximum subsidence for the three bowls in figure 3 can be attributed to different stress conditions, support capacities, and failure mechanisms.

GIS Analysis of Pillar Stresses

Since the hypothesis of tributary area loading on pillars is based almost exclusively on geometry, this type of analysis lends itself to the use of GIS software to estimate the distribution of pillar stress on a mine-wide scale. The AutoCAD drawing

Figure 3.-Estimated subsidence contours with hypothetical associated pillars shaded, minimum contour is 0.05 m, contour interval is 0.025 m; *see figure 4 for location.*

Table 1.-Estimation of Pillar Stresses

	Old High School	Dorris School
Pillar Dimensions		
length.....	17 m	20 m
width.....	13 m	6 m
area.....	221 m^2	120 m^2
Overburden Supported		
length.....	22 m	26 m
width.....	20 m	22 m
area.....	440 m^2	572 m^2
volume.....	26,400 m^3	34,320 m^3
weight.....	537 MN	698 MN
Pillar stress	2.4 MPa	5.8 MPa

Table 2.-Material Properties

	Density kg/m3	Bulk Modulus MPa	Shear Modulus MPa	Constitutive Law	Friction Angle Degrees	Cohesive Strength MPa	Tensile Strength MPa
Soil	2090	330	200	Mohr-Coulomb	30	0.03	0
Rock Mass	2410	630 to 7370	380 to 4420	Mohr-Coulomb	31	2.6	1.0

Table 3.- Horizontal Joint Properties

Normal Stiffness MPa/m	Shear Stiffness MPa/m	Slip Criterion	Friction Angle Degrees	Cohesive Strength MPa
30000	1500	Coulomb	15	0

was converted to GIS coverages using ArcCAD. ArcCAD is a bridge between CAD and GIS integrating the editing and graphical tools from AutoCAD with the topological and analytical tools of a GIS program. A user can toggle between AutoCAD and ArcCAD user interfaces, as well as being able to call upon third-party extensions available to AutoCAD. ArcCAD allows simple conversion of the graphical "entities" of the AutoCAD drawings to GIS "features" without requiring additional data input. ArcCAD is a vector-based GIS that enables the coordinates of the CAD graphical representation to be related to real-world coordinates and map projections. A GIS allows integration of data and maps of disparate scales and sources into one set of geographically referenced data files and establishes topology (relationships) between the geographically referenced features.

The tributary area polygon surrounding each pillar was defined by the centerline of the entries surrounding each pillar. This tributary area network was created using AutoCAD then converted to a GIS POLYGON coverage. Since POLYGON coverages automatically calculate area measurements and are accessible in the database, a ratio between the tributary area and pillar area was calculated by merging the two POLYGON coverages. The tributary area/pillar area ratio was multiplied by an estimated overburden unit weight (20.3 KN/m^3 = 144 lbs/ft^3) to compute pillar stress. Computations were required for several hundred to several thousand pillars for each of the four mines shown in figure 4 which encompassed the entire study area. GIS was then used to classify the mine pillars into four levels of stress:

Class	Stress Level
1	< 3 MPa
2	3 - 4.99 MPa
3	5 - 5.99 MPa
4	>6 MPa

Figure 5 displays pillars below Dorris School. The class 3 stress level is consistent with statistical values for bearing capacity of weak floor strata in the Illinois Basin (Pula et al, 1990).

Numerical Modeling of Pillar Stresses

Numerical modeling was done to provide an independent assessment of the GIS analysis. The methodology was patterned after guidelines outlined by Starfield and Cundall (1988) who encouraged researchers to remember that the focus of numerical analysis should be on gaining an understanding of the mechanisms that characterize the system and not on obtaining unique numeric values for specific parameters. Hart and Cundall (1992) go on to explain that this is accomplished by using the computer model as a laboratory to perform experiments on the system. This can result in an improved understanding of mechanisms, knowledge of parameter dependence, and a means by which to check theories or hypotheses. Ultimately, the new knowledge may lead to new theories or simple conceptual models that can be used in design (Hart and Cundall, 1992). In the present study, the numerical analysis increased understanding of stress levels and the distribution of stresses acting on pillars in the abandoned mines.

The Universal Distinct Element Code (UDEC) (Itasca, 1992) was used to perform desk top experiments and identify parameters that significantly influence model behavior. The computer code allows internal deformation of discrete blocks and also allows large displacements and separations along discontinuities. UDEC utilizes the distinct-element method which is a particular type of numerical model that explicitly incorporates contacts between deformable blocks and explicit time-stepping solution of the equations of motion (Cundall and Strack, 1979). Output in the form of displacements, velocities, and stress distributions can be compiled into movies that make it possible to visualize the influence of various parameters on rock mass response.

Stratigraphy

The distinct element model is composed of deformable blocks separated by discontinuities. Since it is hypothesized that overburden response is affected by the location and character of discontinuities within the rock mass, Bureau researchers used the modified Rock Mass Rating (RMR) system (Bieniawski, 1989) to systematically define the location of horizontal discontinuities in the model. A commercially-available spreadsheet program was used to calculate a RMR for each lithologic bed based on drill core logs and engineering property tests. The RMR was then used to calculate a deformation modulus and bending stiffness for each lithologic bed (figure 6). Large contrasts in the bending stiffness between adjacent lithologic beds have been shown to correlate with measured horizontal shear displacements (Siekmeier et al, 1992). Thus the locations of these large contrasts in bending stiffness were used to define

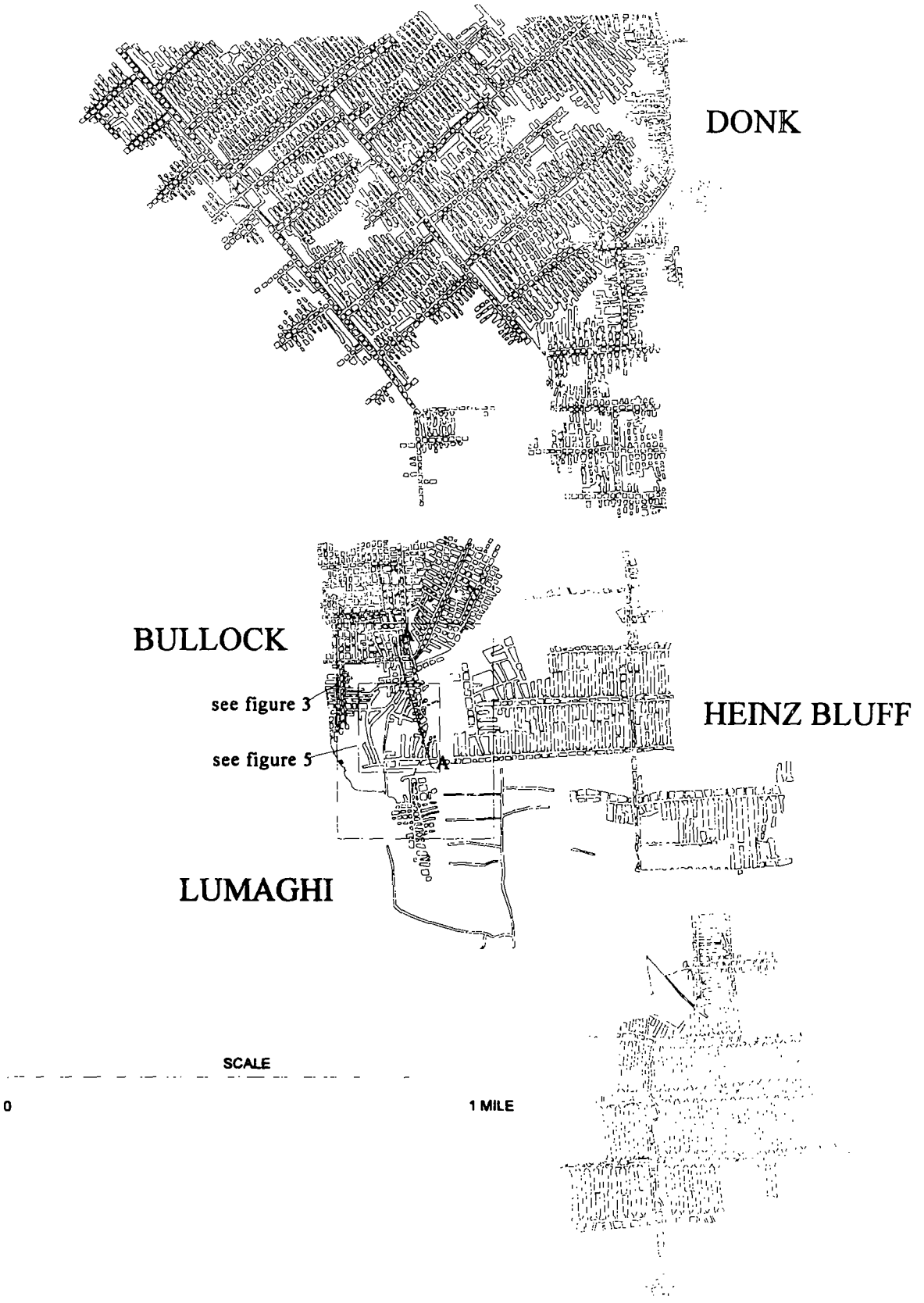

Figure 4.-Display of the four mines in the research area.

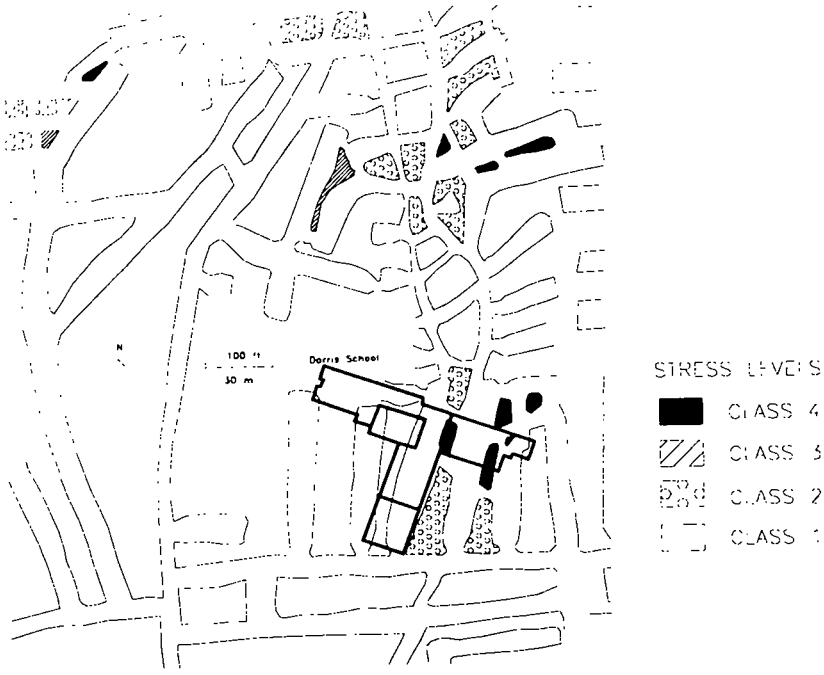

Figure 5.-Dorris Elementary School with pillars from Bullock Mine shaded according to stress level, class 4 is the highest stress; see figure 4 for location.

horizontal discontinuities in the distinct element model (horizontal lines on the right side of figure 6) .

The mines are approximately 60 m (200 ft) below the surface, overlaid by 30 to 34 m (100 to 110 ft) of glacial material and 27 to 30 m (90 to 100 ft) of Pennsylvanian age rock The glacial material consists of 9 to 12 m (30 to 40 ft) of loess overlying silty clay till and sandy stream deposits. The topmost rock stratum is a claystone that has altered to a silty clay of high plasticity. This altered zone is approximately 8 m (25 ft) thick and washed out easily during drilling. The significant features are the two stiffer limestone strata of the mine roof. The upper one is about 1.2 m (4 ft) thick, and the lower one is 6 to 8 m (20 to 25 ft) thick. This lower unit forms the immediate mine roof.

Based on observations made with a downhole camera as well as water level measurements, the mines are flooded and the water is pressurized. An attempt was made to obtain a sample of the mine floor underclay and assess its thickness. The stratum is a highly plastic gray clay at least 1.2 m (4 ft) thick. This thickness was estimated by allowing the drill string to penetrate under its own weight.

Boundary Conditions and Material Properties Used in Numerical Model

A two-dimensional, plane-strain, distinct element model was used to simulate the overburden and abandoned mine along line AA shown in figure 3. The horizontally bedded rock mass and overlying glacial materials were modeled as eleven deformable blocks stacked as a layer cake (figure 6). The blocks are composed of finite-difference triangles and represent layers defined by horizontal discontinuities between strata with significant changes in stiffness.

The deformable block properties are listed in Table 2 and properties of the horizontal discontinuities are listed in Table 3. These values were considered to be reasonable approximations. In order to simulate tributary loading over entries adjacent to the pillars but outside the cross sectional plane, overburden density was doubled. Once again, the objective was to determine if the pillar stresses estimated using the assumption of tributary area loading was reasonable so that any correlation with historical surface subsidence could be considered valid.

Results of Computer Simulations

To simulate abandoned coal mines, blocks representing entries were deleted and gravity loading was applied for 15000 calculation cycles. Support provided by water pressure was not considered and the resulting distribution of vertical stress at two depths is shown in figure 7. The lower line graph is the computed vertical stress acting on the pillars and the upper line graph is the computed vertical stress within the rock mass above the mine roof. Note that the larger stresses occur over pillars adjacent to wider mine openings which is consistent with the tributary area assumption. The maximum value of stress computed for the mine pillars (the lower line graph) is 8.0 MPa which is consistent with maximum stresses on class 4 pillars in figure 5. These results lend validity to the assumption of tributary loading on pillars.

Comparison Between Pillar Stress Conditions and Subsidence Occurrence

While numerical modeling provided an independent check of the tributary loading assumption made in the GIS analysis, the objective was to correlate pillar stress conditions with occurrences of subsidence. Compare the Class 2, Class 3, and Class 4 pillars in figure 5 with the shaded pillars in the center of figure 3. There is good correlation between the subsidence bowl which impacted Dorris Elementary School and the level of pillar stress. This would appear to imply that the GIS analysis assuming tributary loading can provide a basis for assessing the likelihood of subsidence occurrence.

This hypothesis does not appear to be valid for pillars in the top of figure 5. Although there are pillars rated at stress levels of Class 2 and greater, subsidence has not been measured in this area. This does not invalidate the hypothesis but it does demonstrate difficulties with proving that the hypothesis is valid. It is likely that subsidence will occur over highly stressed pillars, but pillar stress analysis cannot predict when subsidence will occur. Furthermore, subsidence may have occurred but it was not measured since there are no structures in this area which could have suffered damage.

Another compounding factor is the influence of mine flooding. Based on downhole camera inspection of mine conditions and water level

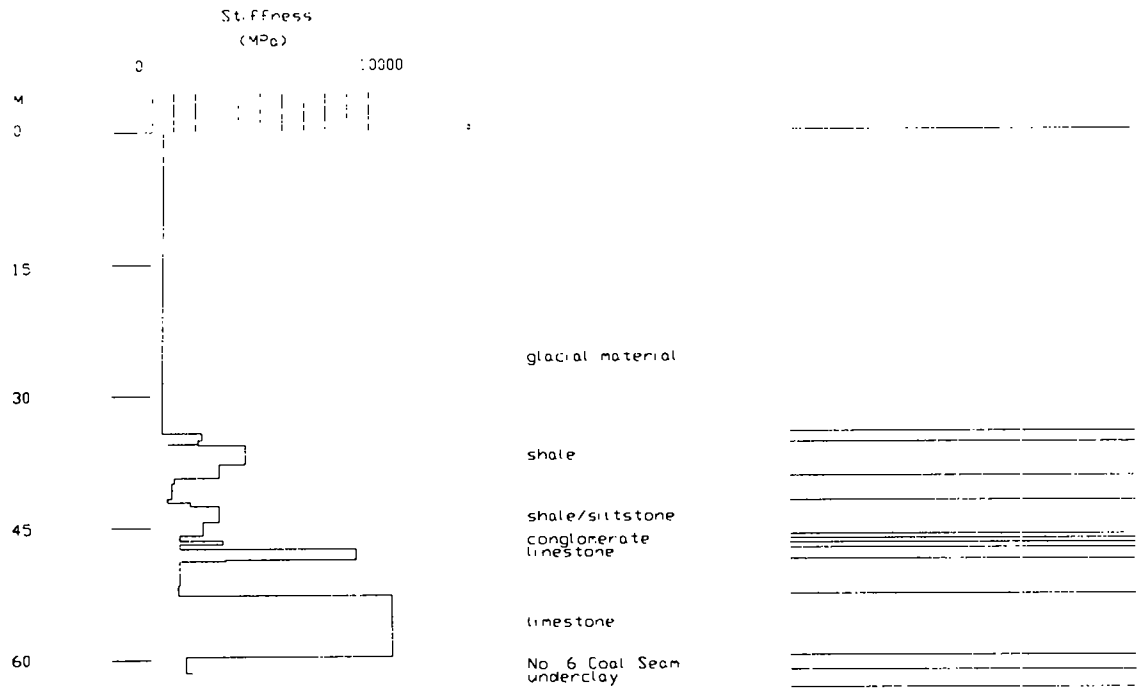

Figure 6.-Stiffness histogram and geologic model used for numerical simulation.

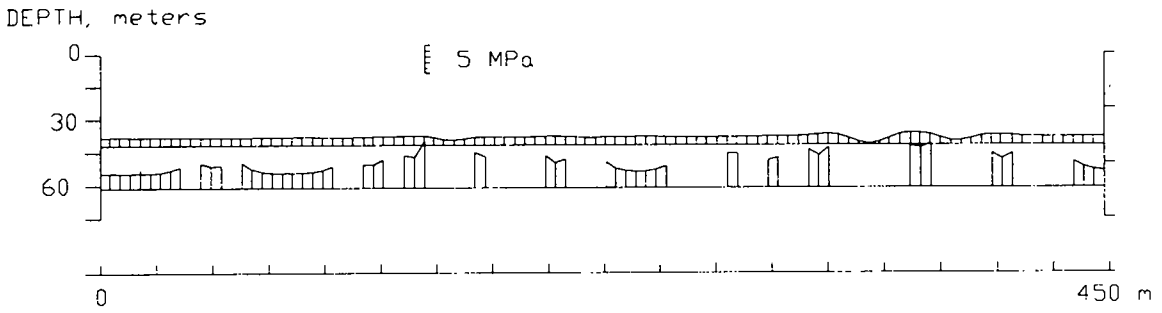

Figure 7.-Distribution of computed vertical stress acting on pillars and along top of main roof stratum; cross section is along line AA in figure 3.

measurements, it is known that the mine is flooded and the water is pressurized. Support for the mine roof is provided simultaneously by pillars and water pressure. Consequently, the actual pillar stress is less than that computed by tributary area loading. If the water pressure dissipates then pillar stresses will increase and the likelihood of subsidence occurrence will increase (USEPA, 1981).

Summary

This paper presents a study which was conducted to assess the correlation between pillar stress and subsidence occurrence over abandoned coal mines. Historical data were assembled and subsidence contours were generated for each date that measurements were made. These contours were overlaid on a CAD base map which included the digitized mine pillars and entries. The CAD layers were selectively activated and a time lapse animation was created to observe the areal progression of subsidence occurrences. Assuming tributary loading on pillars, stresses were computed for several thousand pillars using GIS software. An independent check of the tributary loading assumption was made by numerical modeling of the overburden and abandoned mine. Pillars were then classified by stress level and this classification map was overlaid with the subsidence contours.

Pillar stresses estimated using the hypothesis of tributary loading are consistent with those estimated using the numerical model. Subsidence occurrences are not random events and have developed in a manner which is physically consistent with the layout and stress level of mine pillars. The combination of CAD techniques and GIS analysis offers a valuable tool for subsidence risk assessment.

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MINE DESIGN: LONG TERM EFFECTS OF HIGH EXTRACTION MINING[1]

by

Paul W. Jeran[2]

Abstract. A consideration when designing a high extraction coal mine is the effects that mining will have on the ground above the mine. This becomes particularly important when the surface has been improved or is inhabited. Surface owners are concerned about; when the effects will begin?, how large will they be?, and how long they will last?. Each of these should be addressed by the designer.

For more than a decade, the U.S. Bureau of Mines (USBM) has been monitoring subsidence at various sites. Based upon the data gathered, some inferences may be made regarding the above stated questions. Essentially surface movement begins with undermining. The magnitude of the movements are proportional to the thickness extracted and the width of the mined area, and inversely proportional to the depth of the mine below surface. The duration of the subsidence process in the northern Appalachian Basin is approximately one year.

The USBM has developed a computer model which predicts the final subsidence profile across a longwall panel in the northern Appalachian Coal Basin. USBM studies on the dynamic development of subsidence have shown that the magnitude of the deformations developed during the subsidence process never exceed those exhibited in the final subsidence profile. Use of the model will provide engineers with a starting point in the design process.

INTRODUCTION

The mining of a large rectangular block of coal by the longwall method results in the development of a trough-shaped depression of the surface above the extracted area. The process of subsidence is dynamic as surface movement begins with undermining and continues until some maximum displacement has occurred: the magnitude of which is controlled primarily by the extracted thickness, the width of the panel, and the overburden thickness and geology. At the end of the subsidence process, equilibrium is achieved and the resulting surface deformations become static. Any subsequent movement of the surface does not result from subsidence but from the altering of the conditions affecting the rock strata disturbed by subsidence (i.e., injection or removal of fluids, erection of structures).

During any discussion of subsidence with longwall mine operators and particularly surface property owners, several questions always arise: How much subsidence will occur?; What will be affected?; When will subsidence begin?; and How long will the process of subsidence last? The first two questions can be estimated using the geometry of mining as input into the U.S. Bureau of Mines' (USBM) subsidence

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prediction model for the northern Appalachian Coal Field (Jeran 1986). The latter two are the subject of this paper.

Subsidence monitoring is typically carried out in two directions: along the centerline of the panel to obtain data which will show the maximum amount of subsidence and the duration of the dynamic phase of movement; and, across the panel (perpendicular to the centerline) to evaluate: the lateral extent of surface deformations, the final cross-section of the developed subsidence trough, and the distribution of final deformations. This is the static result of the subsidence process and any further movement must be attributed to some other cause. Subsidence prediction is typically limited to the final shape of the subsidence trough. Practically all predictive models address this aspect of mining-induced ground movement.

Study of subsidence data gathered along the centerlines of a number of longwall panels in the northern Appalachian Coal Basin, by Adamek and Jeran (1992), has shown that the subsidence process starts with the undermining of a surface point and is 90 percent completed by the time the longwall face has been advanced the thickness of the overburden beyond that point. In this study, it was determined that: the speed with which the face is advanced has no effect on the magnitude of surface deformations; the final movement is usually achieved with mining of the adjacent panel; and, the magnitude of surface deformations that occur during dynamic subsidence are always less than the static case (Adamek 1992)

Examination of manmade features on the surface has shown that they are affected by the forces applied to them throughout the subsidence process. Therefore, the final static shape of the subsidence trough cannot be used to fully explain their degradation or how they came to their final condition. It has also been observed that the degree of degradation is dependent upon location within the subsidence trough (Walker 1990).

Field Studies

As has been noted above, the description of dynamic subsidence has been limited to movement along the centerline. However, does this behavior occur equally across the developing subsidence trough? In an effort to obtain some insight to this question, long term subsidence data were needed. At one site, in northern West Virginia, the USBM monitored a series of longwall panels, in the Pittsburgh Coalbed, remote from previous high extraction mining. Monitoring was conducted for over two years during which time four adjacent longwall panels were mined. Surface monuments were installed over the first three panels (figure 1) and monitoring was conducted over the portions of the array that were actively moving due to undermining. Periodic measurements were also made of the previously undermined portions of the array to determine final movement. Above the first two panels there was no further vertical movement detected after the subsequent adjacent panel was mined (Jeran 1988).

To simplify the analysis, four monuments across the first panel, as shown in figure 1, were selected: (A) above the centerline, (B) 30.5 m (100 ft) from the centerline, (C) 30.5 m (100 ft) inside the rib abutting the barrier pillar, and (D) above the rib abutting the barrier pillar. Figure 2 shows a plot of the subsidence of these four monuments verses time. Also included are the face positions showing the progress of mining for the studied panels. The last measurement was made after mining of the fourth panel was completed. Each face position line is marked to indicate the location of the

face when the four monuments were either undermined or when the longwall face passed by them. From this, we can see that each monument experienced the majority of its movement with the mining of Panel 1 and had completed movement with the mining of Panel 2. Figure 3 shows the first 60 days of the study. From this, it can be observed that the monuments were undermined on the 25th day and some minor movement had been detected as much as 10 days earlier. Furthermore, most movement had occurred within 30 days after undermining.

Since each monument subsided a different amount, the percentage of final movement for each measurement was computed to provide some basis of comparison. Please note that negative values of time and face position, where they occur on graphs, indicate data acquired prior to undermining of the monuments. When the percentages of final subsidence are plotted against face position (figure 4) it can be seen that the three interior monuments (A, B, and C) subsided more than 85 percent of the final movement by the time the face was 245.4 m (800 ft) past the monument line. Movement of the monument over the rib (D) was only 70 percent complete at this time. Plotting the percentage of final subsidence against time (figure 5) shows that all of the monuments behaved similarly for the first 10 days after undermining. After that period, the two central monuments (A and B) behaved similarly and the monument 30.5 m (100 ft) inside the rib (C) lagged slightly behind. The rib monument (D) behaved differently.

From the above, it can be concluded that subsidence of the monuments within the panel limits is governed by face position and time. While subsidence of the monument over the rib, after its initial movement, is more governed by time.

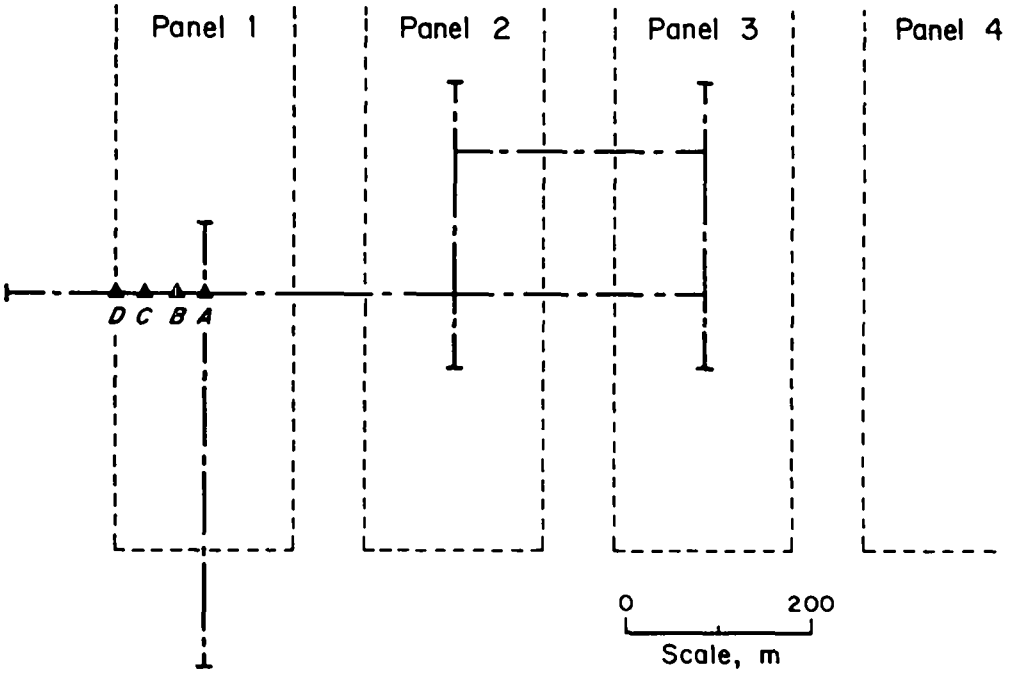

Figure 1. Monitoring Array

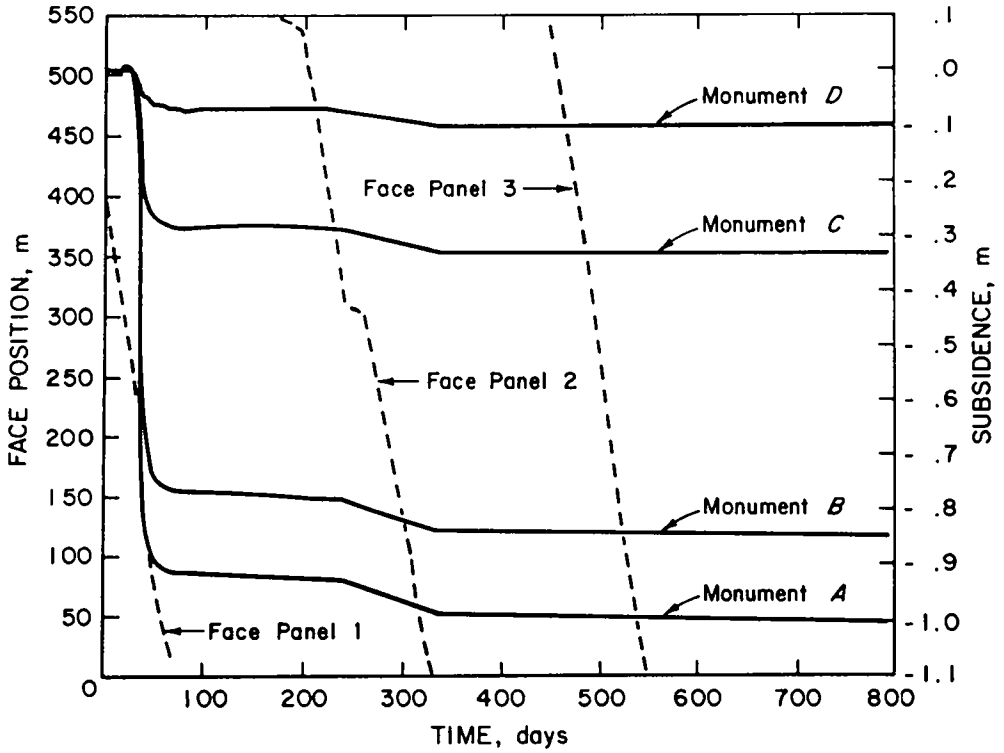

Figure 2. Subsidence during the study versus time

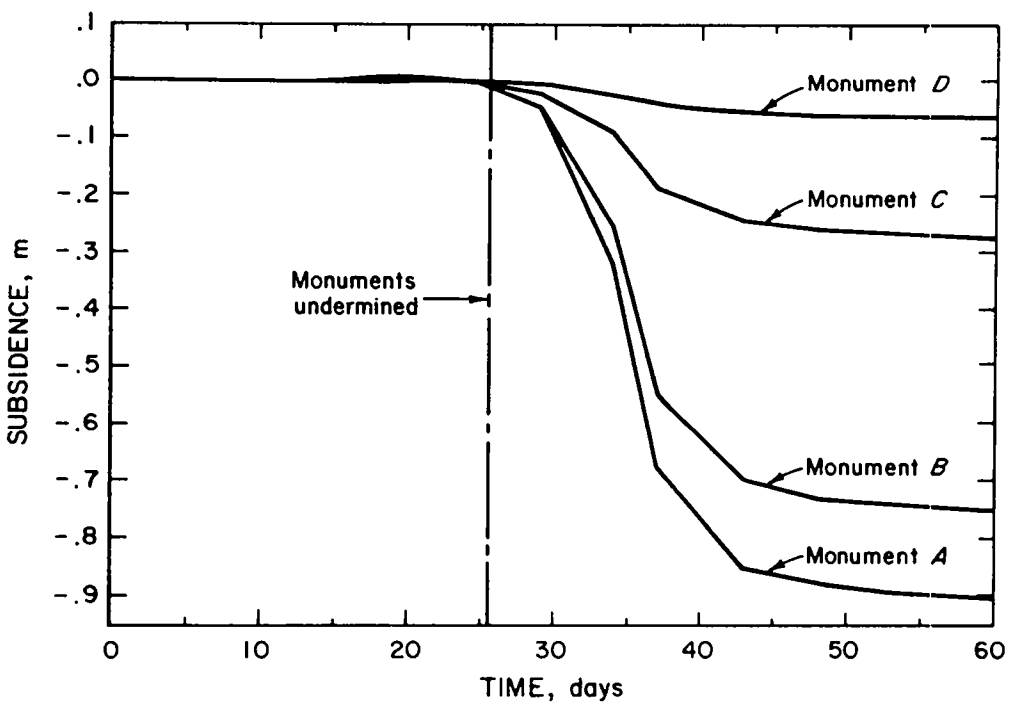

Figure 3. Subsidence during first 60 days

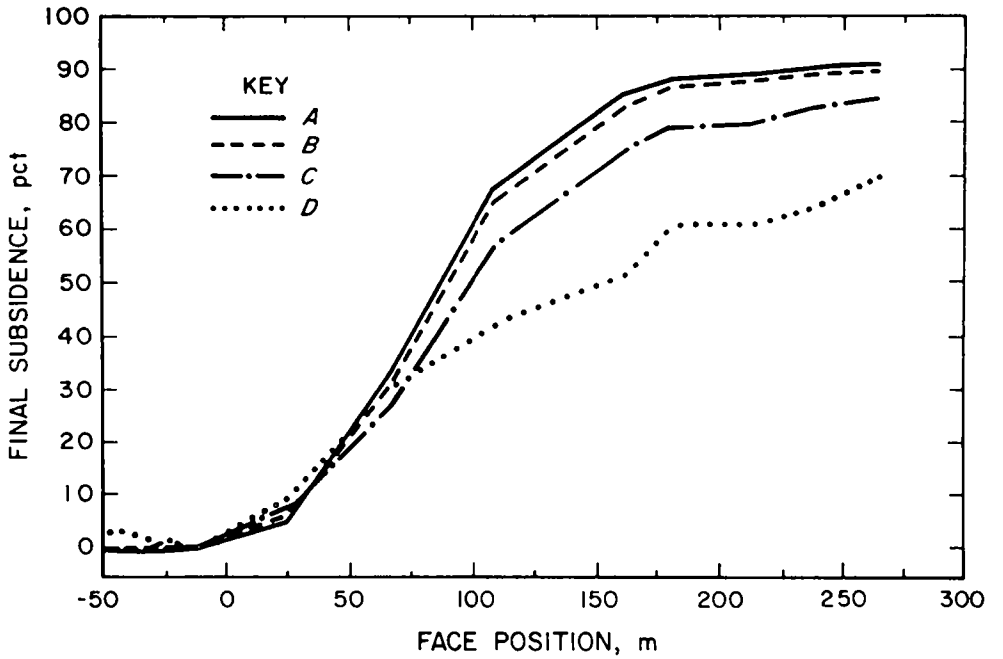

Figure 4. Percent final subsidence versus face position

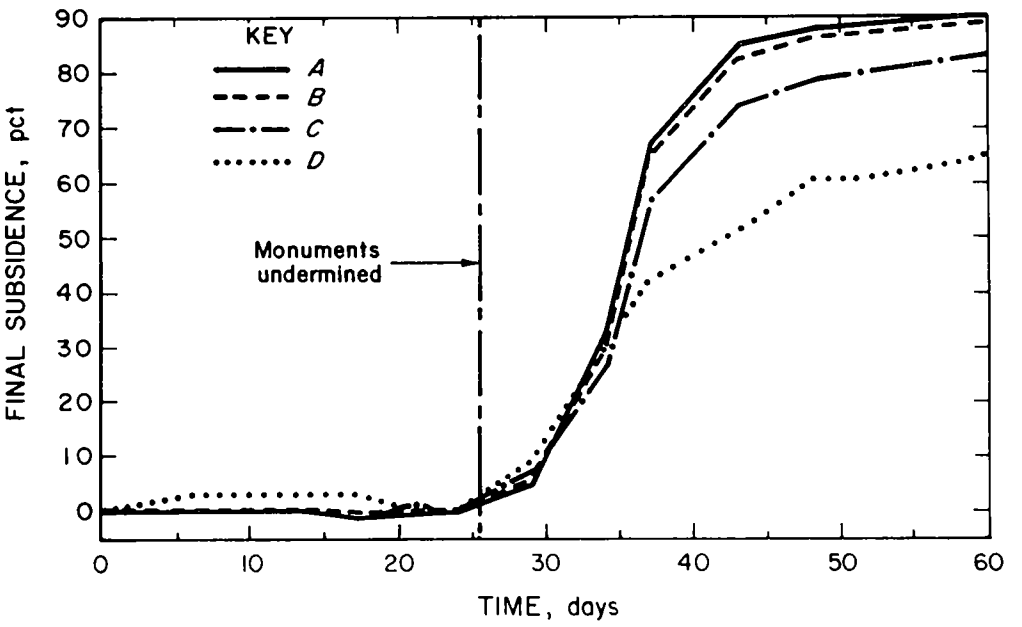

Figure 5. Percent final subsidence versus time

Since these conclusions only represent information from one site, other sites, with shorter duration of study, were investigated to substantiate our observations. Mine sites were selected, one in each of three different coal beds: Pittsburgh, Lower Kittanning, and Freeport. Figure 6 shows the locations of these sites.

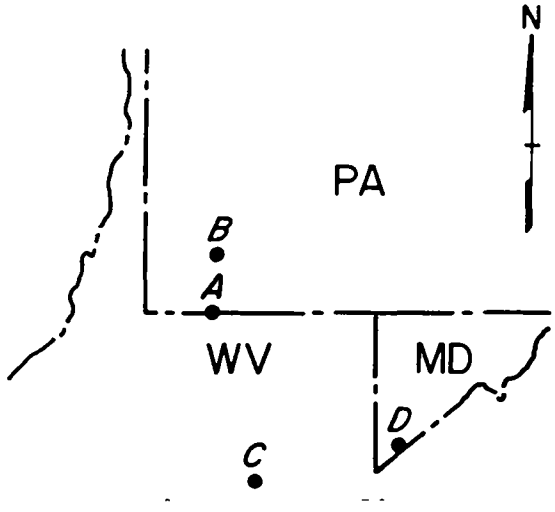

Figure 6. Location of Mines

Mine A is the original site. The Pittsburgh Coalbed site was chosen to determine if the observations were limited to operations in this coalbed (the original site was in the Pittsburgh Coalbed). Overburdens, extracted thicknesses, rates of face advance and geometries varied among the sites. Figure 7 is a plot of face advance for each of the sites. The average rates of mining ranged from 3.1 m/day (10.0 ft/day)at Mine B to 11.8 m/day (38.8 ft/day) at Mine D. Table 1 contains data describing overburden thickness, panel width, average rates of face advance, and coalbed for the sites.

In all cases, the monuments chosen were from the profile line extending from the centerline outward away from subsequently mined panels. Since the extracted thicknesses were different and resulted in differing magnitudes of subsidence, the percentage of the final movement of each monument was calculated and used in the comparisons. It should be noted that the process of subsidence at all sites was completed within one year of undermining.

Data from the centerline monuments at each of the 4 sites plotted against time (figure 8) show that subsidence at mines C and D generally agreed with that at Mine A, while Mine B was much slower. Mine B had the slowest face advance and greatest overburden thickness. Considering that dynamic subsidence is proportional to face advance and inversely proportional to overburden thickness (Adamek 1992) this could be the expected result. When the percentage of final subsidence was plotted against face position, (figure 9) Mines A and B (Pittsburgh Coalbed) are in agreement with Mines C and D differing. It should be noted however, that Mines C and D had lesser overburden thicknesses. When plotted against face position in terms of overburden thickness (figure 10) all sites are in closer agreement. Note, in figure 10, for the centerline monuments, subsidence at all sites was about 90 percent complete when the face had advanced the thickness of the overburden beyond the monument location. These observations agree with the findings of the earlier mentioned study by Adamek and Jeran (1992).

The same plots were made for the monuments 30.5 m (100 ft) from the centerline for the 4 sites under study (figures 11-13). Again, as for the centerline points, the plot versus face position in terms of overburden thickness shows the closest agreement and again, over 90% of the final subsidence was completed when the face

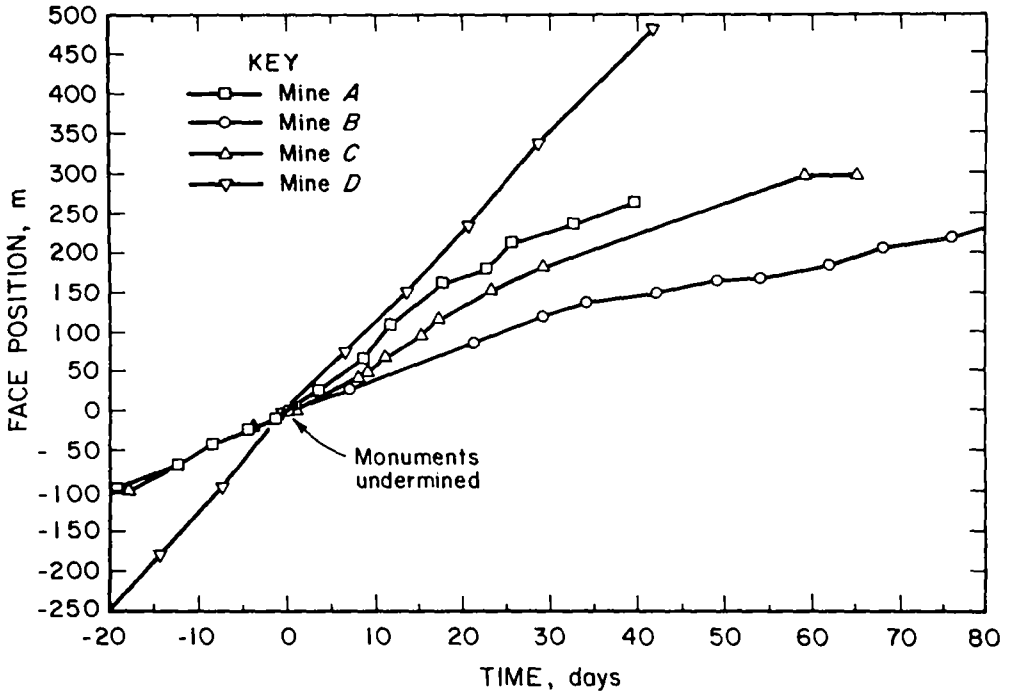

Figure 7. Face advance for each site

Table 1. Study Sites Information

Mine	Panel width, m(ft)	Overburden, m (ft)	Average face advance m/day (ft/day)	Coalbed
A	190.5 (625)	210.3-213.4 (690-700)	6.55 (21.5)	Pittsburgh
B	182.9 (600)	277.4-298.7 (910-980)	3.05 (10.0)	Pittsburgh
C	289.6 (950)	118.9-126.5 (390-415)	4.94 (16.2)	Kittanning
D	182.9 (600)	143.3-155.4 (470-510)	11.83 (38.8)	Freeport

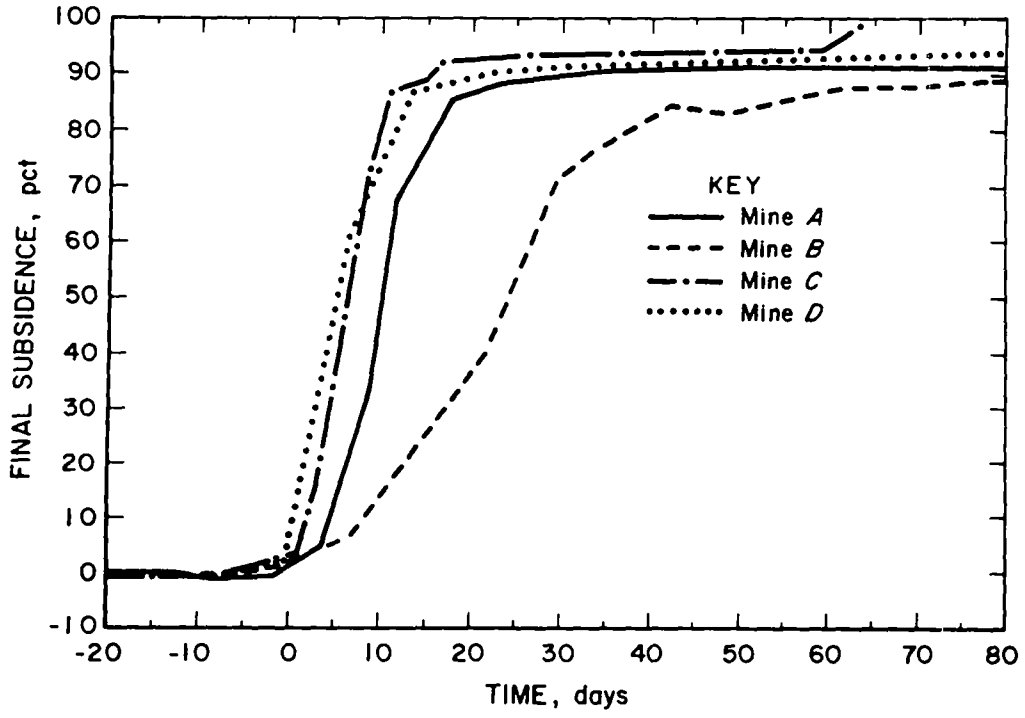

Figure 8. Centerline poinst versus time

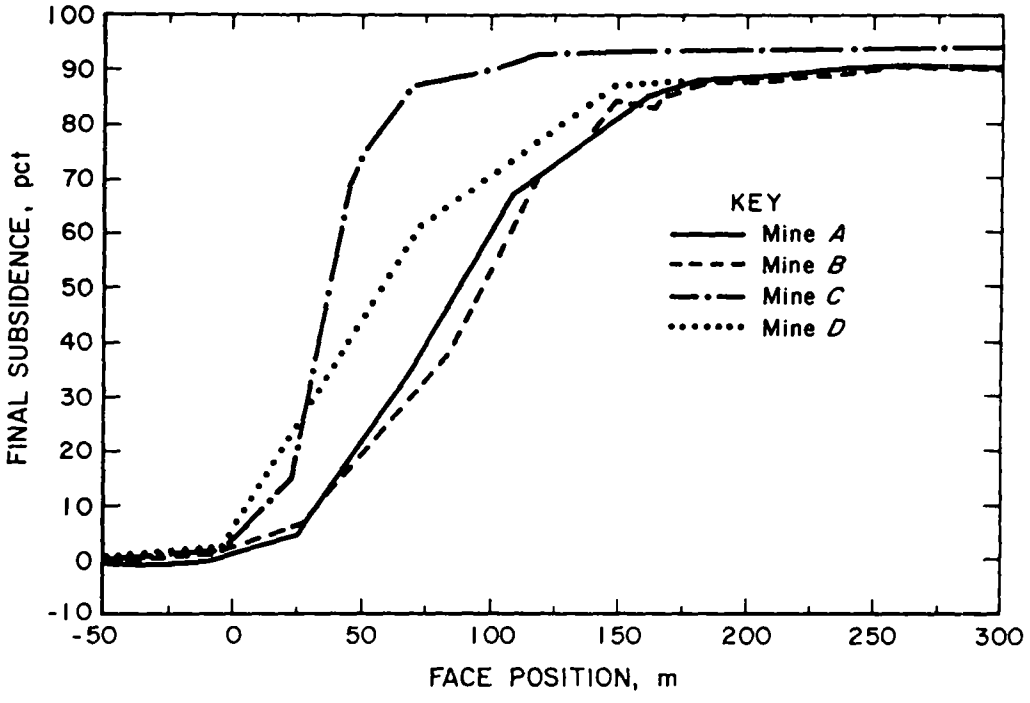

Figure 9. Centerline points versus face position

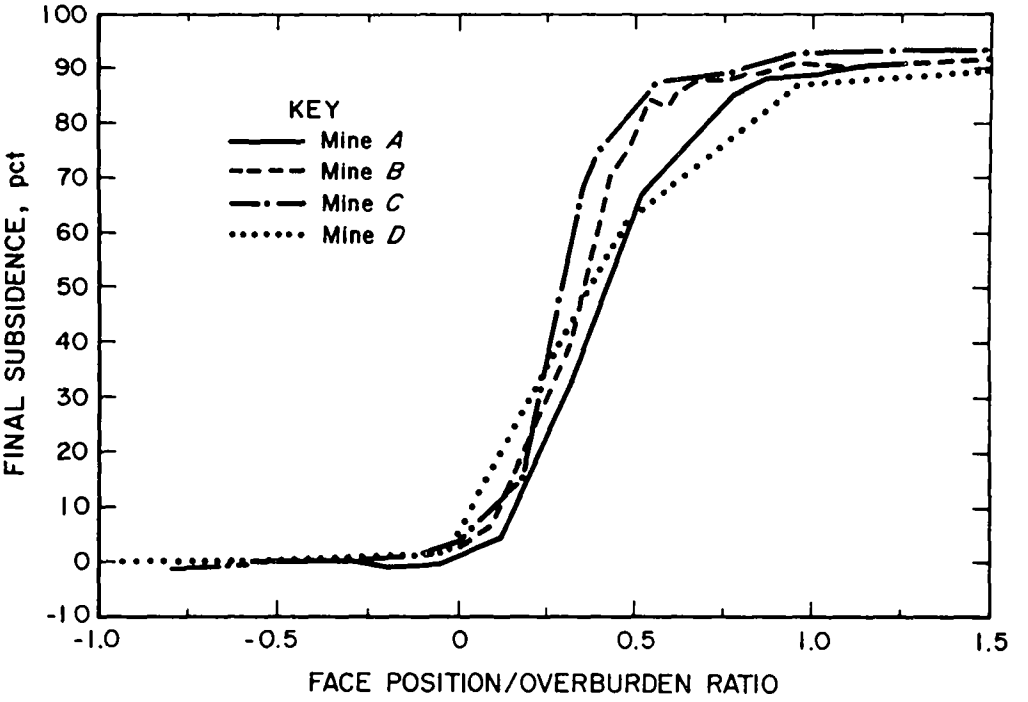

Figure 10. Centerline points versus face position/overburden

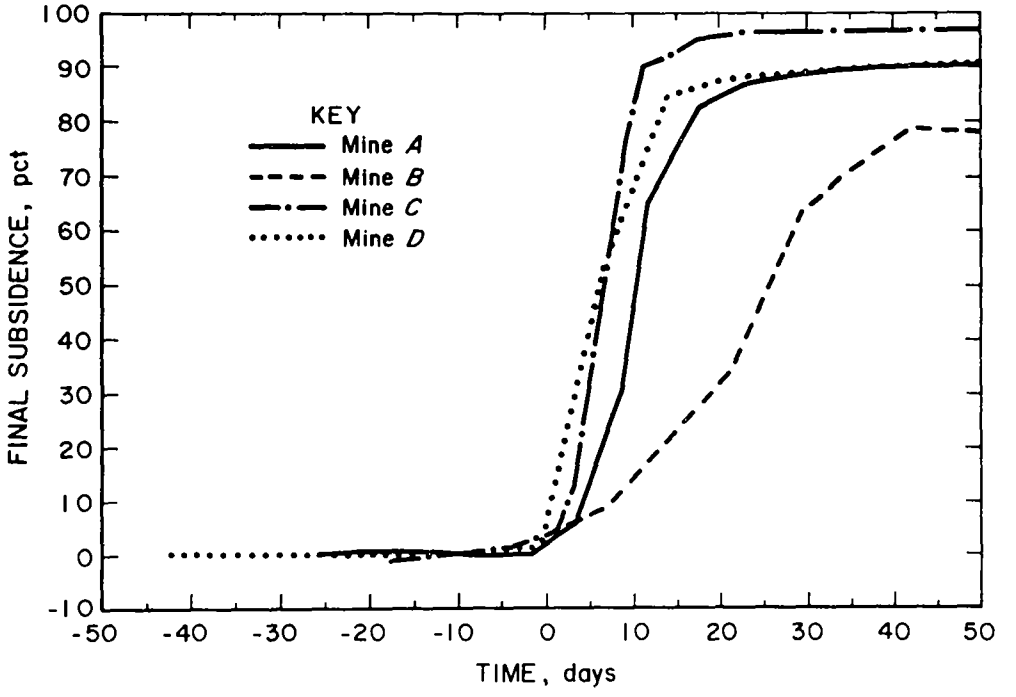

Figure 11. Centerline +100 points versus time

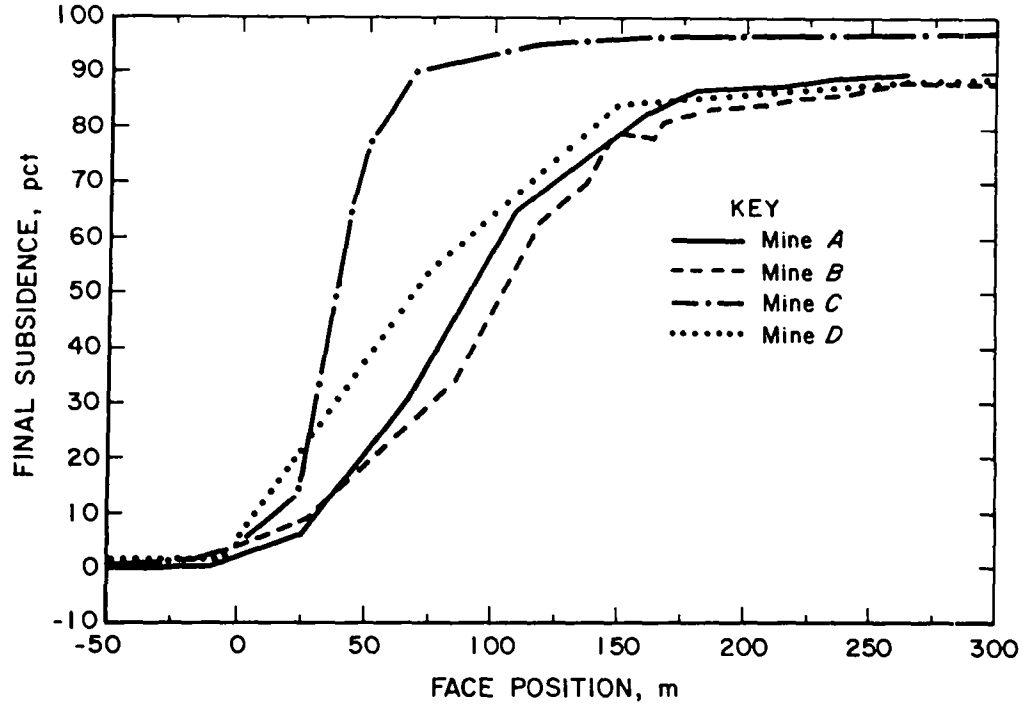

Figure 12. Centerline +100 points versus face position

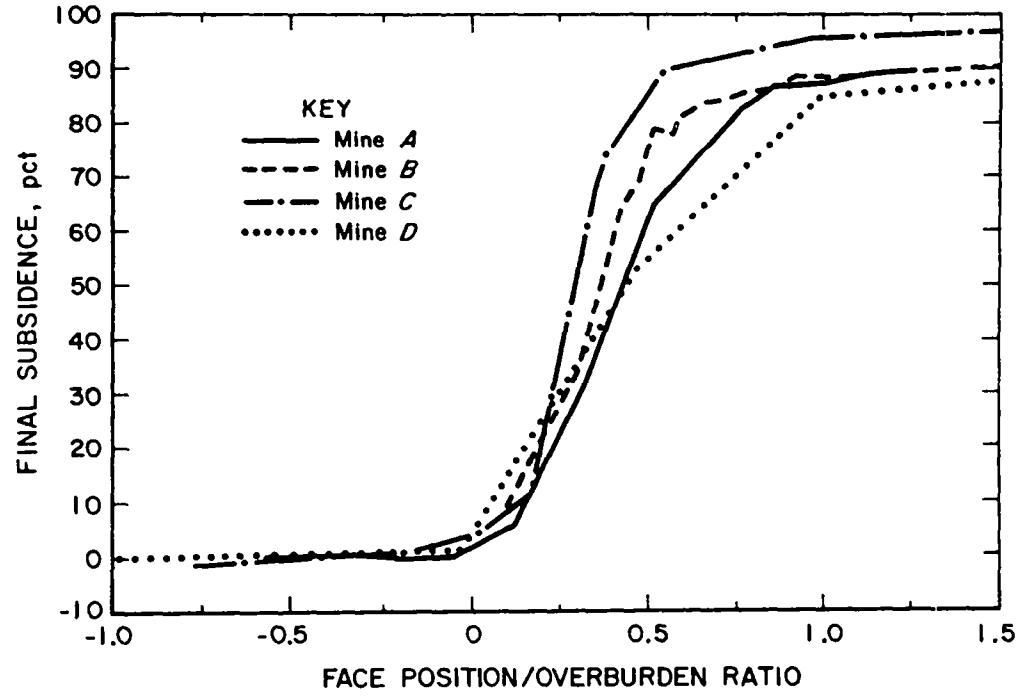

Figure 13. Centerline +100 points versus face position/overburden

had advanced the thickness of the overburden past each point.

Plots for the monuments 30.5 m (100 ft) inside the rib versus time and face position (figures 14 and 15) are similar to those for the more centrally located monuments. However, the plot the percentage of final subsidence versus face position/ overburden ratio (figure 16) shows that this location at Mine D lagged behind the same location at the other 3 mines indicating retardation of the subsidence process. The overburden at this site contains significantly more and thicker sandstone units. These stiffer members bridge further over the gob than the less resistant strata at the other sites. Additional time is needed for the stiffer units to bend into the gob area.

The similar plots for the rib monuments (figures 17-19) show a different characteristic. Plots of the percentage of final subsidence versus face position (figure 18) and face position in terms of overburden thickness (figure 19) do not show the same trends as observed for the other points. The plot of percentage of final subsidence versus time (figure 17), however, shows that within a month of undermining these monuments had completed from 40 to 70 percent of their final subsidence. This indicates that the process of subsidence over the rib is very much site dependent and time appears to play a very significant role.

Deformations

Underlying all discussions of subsidence resulting from longwall mining is the concern for damage. From the above discussion it can be seen that except for the area above the rib most of the surface movement is accomplished by the time that the longwall face has mined past the thickness of the local overburden. It is therefore logical to assume that most of the deformations of the surface are also completed by this time.

The study of dynamic subsidence showed that the magnitude of dynamic deformations (inclination, curvature, and horizontal strain) are always less than the static values (Adamek 1992). Therefore, if the difference between the values of final static deformations and those developed at the time the longwall face has mined one overburden thickness past the profile are small, then the additional deformations from a dynamic point of view should also be small.

The distribution of static inclinations were calculated for each of the profiles used in this study at the point where the longwall face had mined 1 overburden thickness past and again when subsidence had been completed. These are shown in figures 20 - 23. They show that static inclinations increased by less than 3 mm/m. Therefore, dynamic inclinations should be smaller than these values. From these we must conclude that, for these sites, the surface deformations occurred primarily during the mining of the longwall face one overburden thickness past the profile. Additional deformations were small. Since there are no criteria established in this country correlating magnitude of deformation with structural damage it is impossible to state that no additional damages would result from the additional deformation. If criteria developed in Europe (Adamek 1982) are any indication, then these additional deformations should not significantly contribute to surface damages.

Summary

Subsidence data from four sites in the northern Appalachian Coal Basin were studied. These data show that for locations in the central portion of the

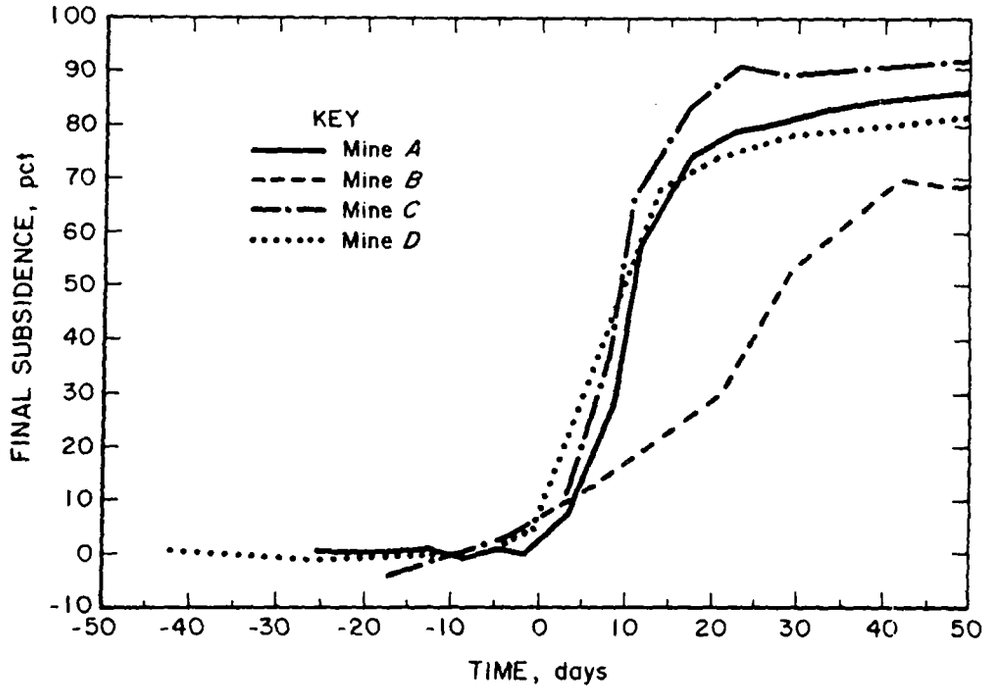

Figure 14. Rib -100 points versus time

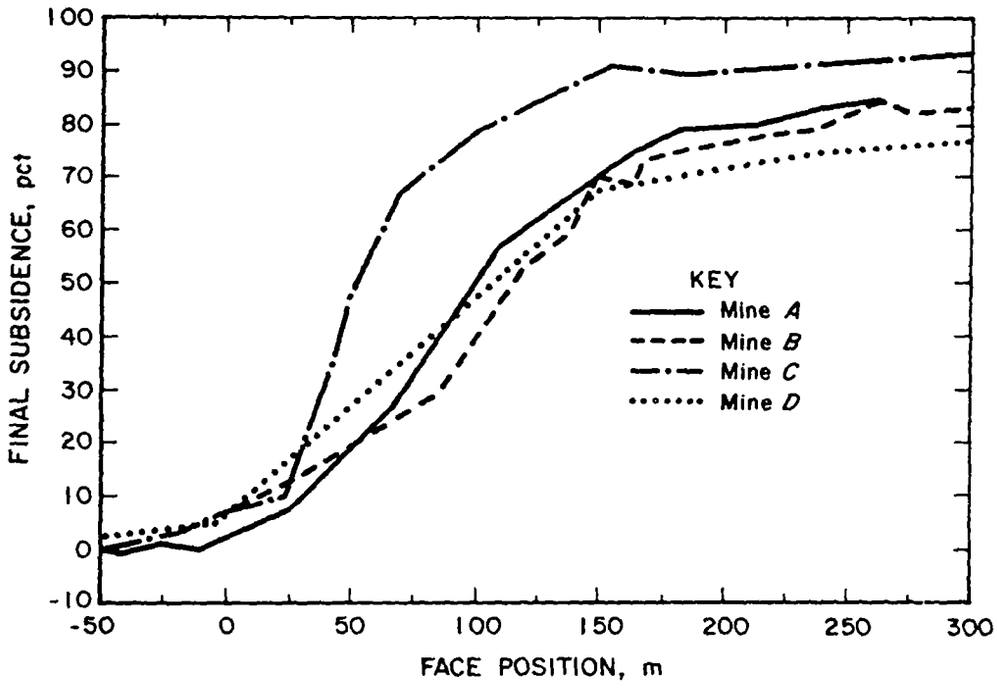

Figure 15. Rib -100 points versus face position

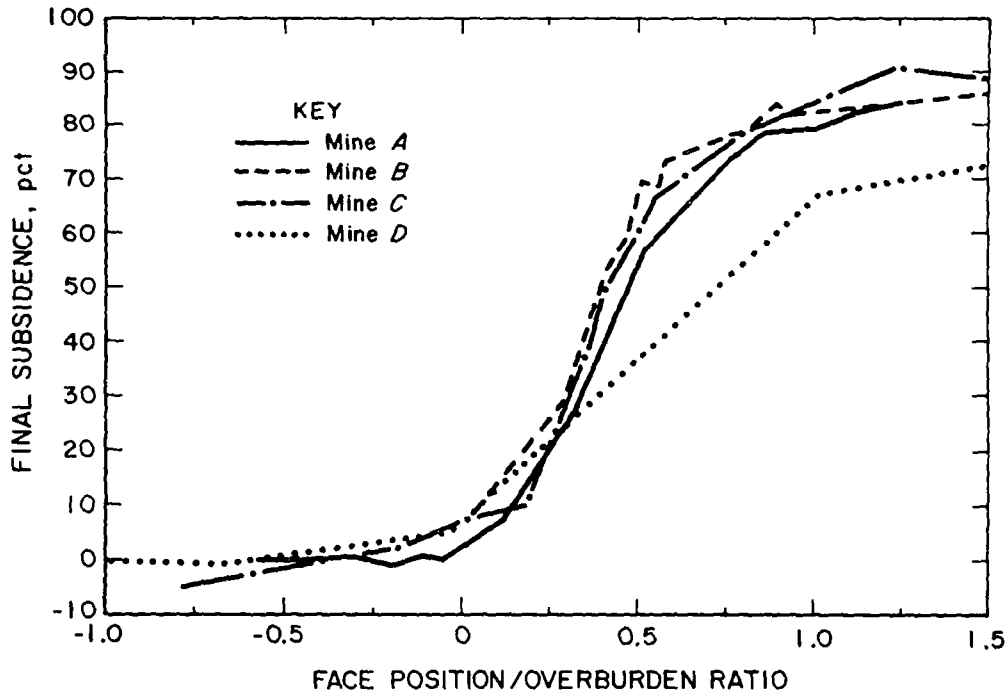

Figure 16. Rib -100 points versus face position/overburden

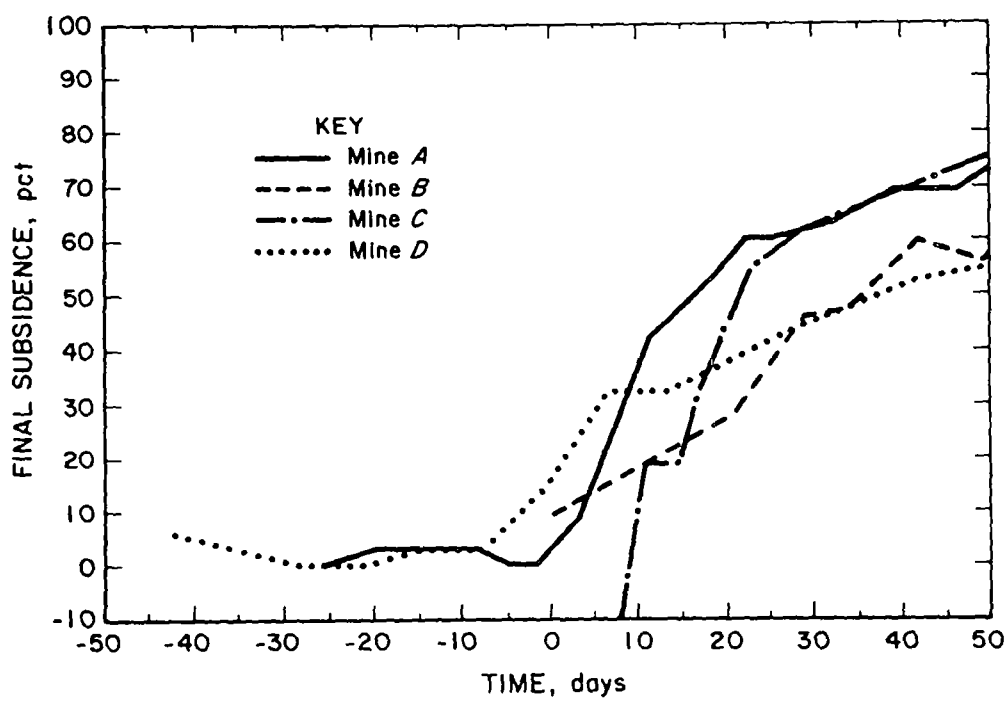

Figure 17. Rib points versus time

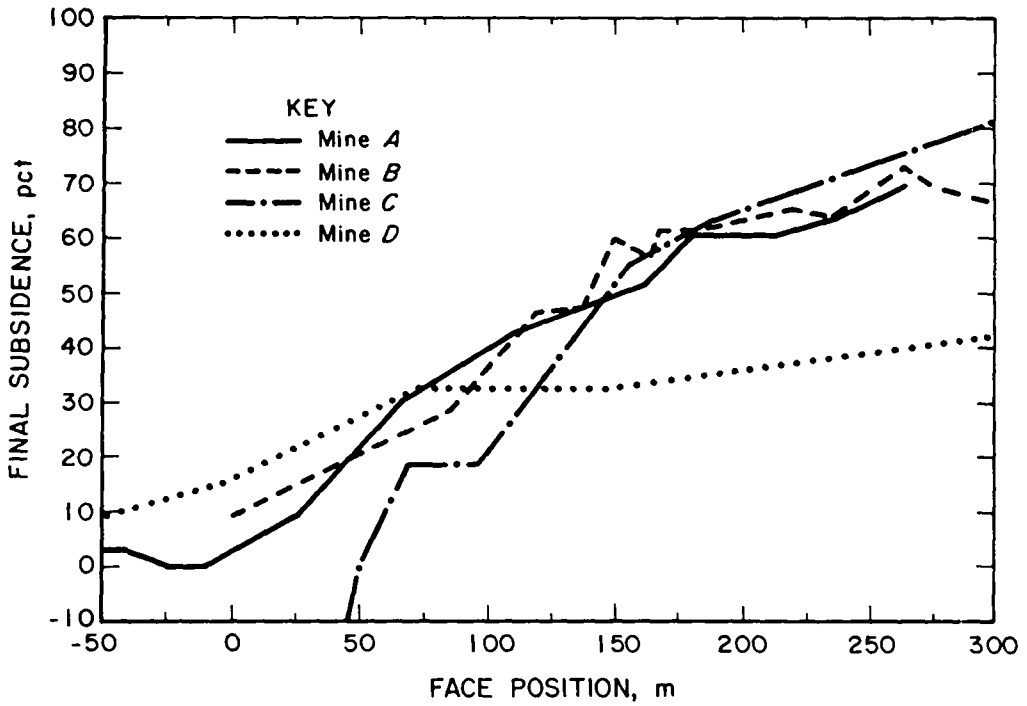

Figure 18. Rib Points versus face position

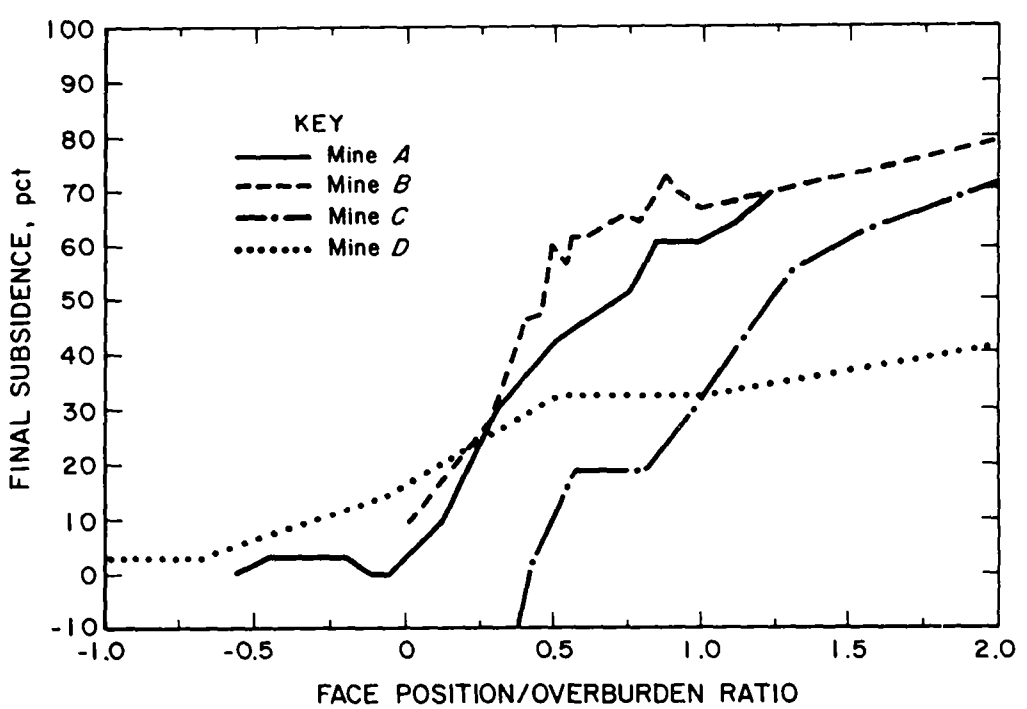

Figure 19. Rib Points versus face position/overburden

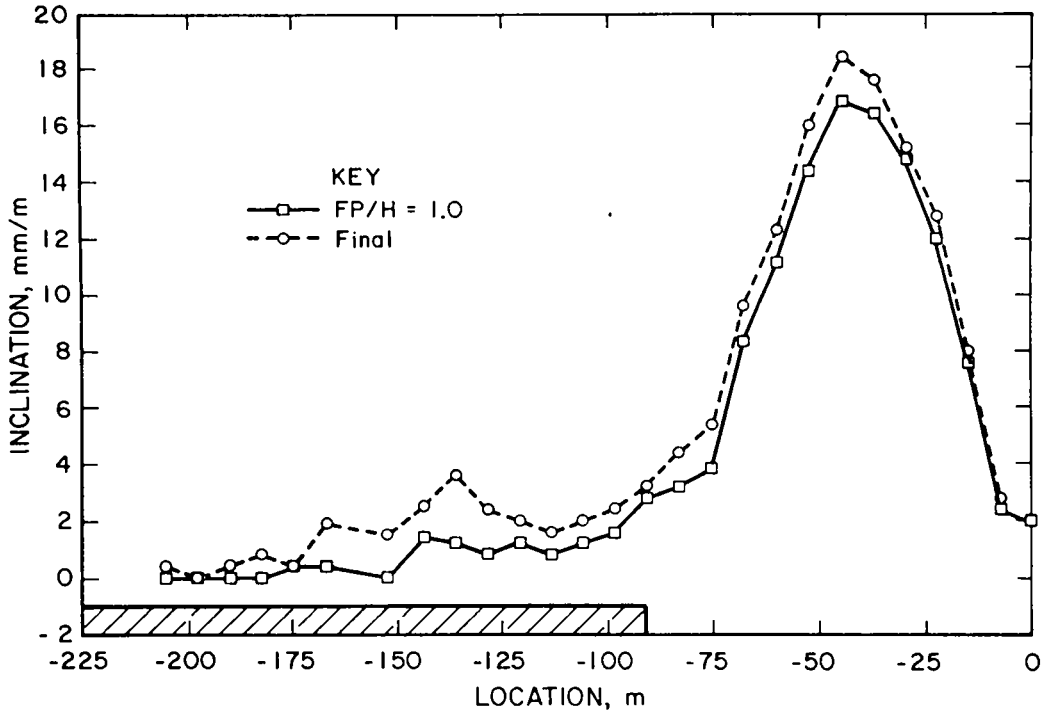

Figure 20. Inclination difference at Mine A

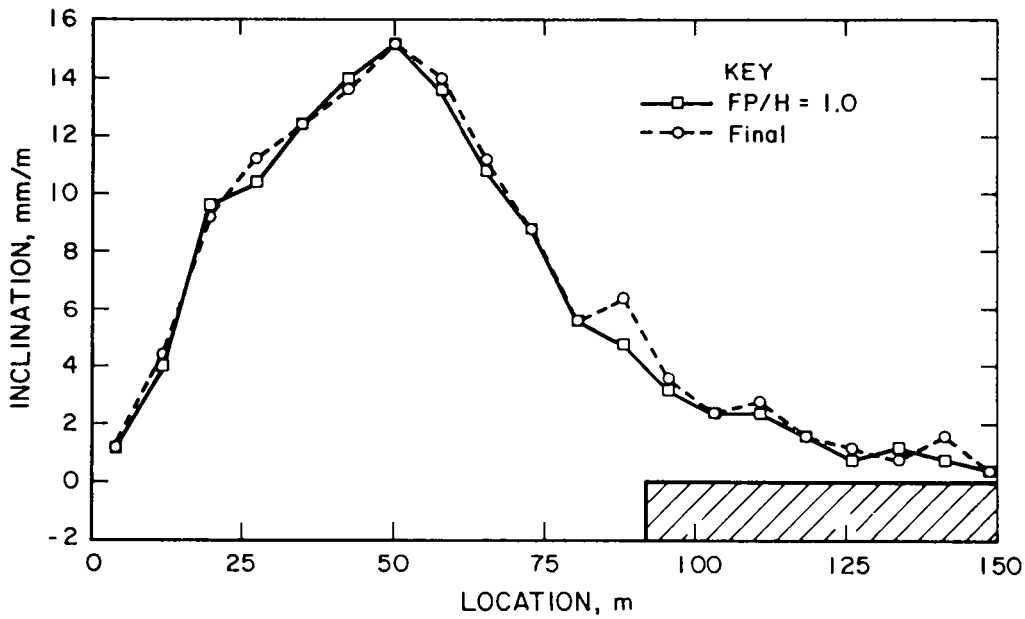

Figure 21. Inclination difference at Mine B

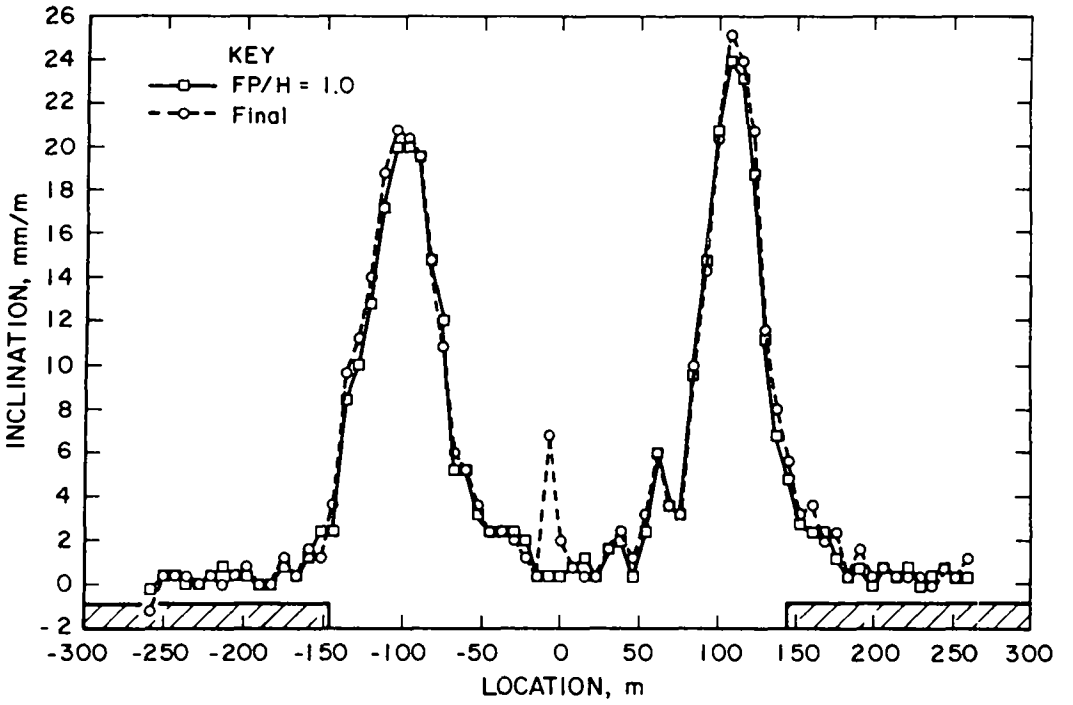

Figure 22. Inclination difference at Mine C

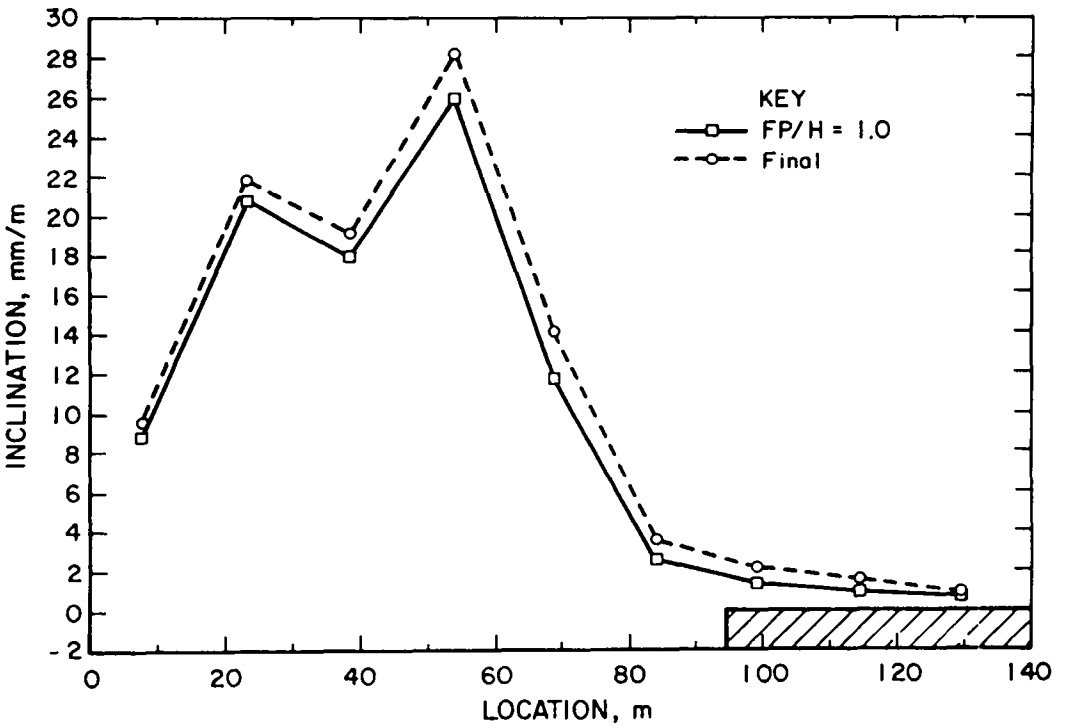

Figure 23. Inclination difference at Mine D

developing subsidence trough (within 30.5 m (100 ft) of the centerline) subsidence develops relative to face position and overburden thickness and is about 90 percent complete when the face has advanced the thickness of the overburden beyond the point in question. Above the rib of the panel the process of subsidence is very much site dependent and time appears to play a significant role, in that, when the face has advanced the thickness of the overburden past a monument only 40 to 70 percent of the movement has taken place. The remainder of the surface movement, irrespective of location above the longwall panel, takes up to a year to complete.

The monuments 30.5 m (100 ft) inside the rib exhibited movements that were characteristically between that of the centerlines and rib areas. With slow to moderate face advance (3.1 to 6.6 m/d), the subsidence process is similar to that over the central portion of the panel with over 80 percent of the final movement completed when the face has advanced the thickness of the overburden past a monument. However, with fast face advance (11.8 m/d) there appears to be some retardation or lag in the subsidence process.

At the sites studied, the preponderance of the surface deformations had occurred by the time that the longwall face had mined one overburden thickness past each profile. The additional deformations were small. All of these factors should be taken into account when designing a high extraction mine to evaluate the potential for damage resulting from mining-induced subsidence.

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ERROR CORRECTION FOR VERTICAL SURVEYS CONDUCTED OVER A SUBSIDING LONGWALL MINING PANEL

by

D. Allen Hughes

Abstract: The difference between a conventional land survey and a survey of subsiding ground is discussed and a correction method was formulated for surveys conducted on subsiding ground. The area over the longwall mining panel subsided detectible amounts during the time required to conduct the survey when subsidence was at its highest rate, which introduces error into the survey. When the ground subsides before the survey is completed, the survey no longer represents the locations of all points at a common point in time, which is a basic assumption of conventional land surveying. Conventional methods of correction average movement of subsiding points and apply those amounts of movement to points which were unaffected by subsidence, a different correction method was needed. A correction method was used which uses multiple surveys to calculate rates of subsidence for each point in the survey. Subsidence rates were used to estimate the location of each point at a common time. Results are presented using the correction for subsiding ground and using no correction. Different results of the same surveys are shown in terms of elevations and curvatures. The significance of the different types of corrections is discussed and the compounding of error is demonstrated when calculating curvatures.

Additional Key Words: Longwall Mining, Error Correction, Subside.

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[3]Publication in this proceedings does not prevent authors from publishing their manuscripts, whole or in part, in other publication outlets."

Introduction

Mine subsidence is the product of a chain of events that begins several hundred feet below the surface of the ground and propagates upward. The end result is a subsidence trough. Strains caused by subsidence in the area of the trough is the direct cause of structural damage. An accurate, time efficient and cost-effective method of measuring the behavior of the trough is critical. Perhaps the most reliable and accepted method of monitoring mine subsidence is by the use of conventional surveying techniques. However, some modifications to the surveying procedure must be made when monitoring mine subsidence.

Survey Setup

Multiple surveys were conducted over a longwall mining panel near West Frankfort in southern Illinois. Monitoring by surveying was started before the area began to subside, conducted at daily intervals during the peak subsidence and continued until subsidence had ceased. The area subsided between the months of May and August of 1990.

Soils of southern Illinois are typically highly expansive. To accurately measure subsidence, it was necessary to avoid measuring movements caused by other sources, such as freezing and thawing of the soil, and shrinking and swelling of the soil caused by changes in the moisture content. These types of movement occur most often near the surface.

Each survey monument had to be installed so that it would remain as stable as possible. A steel bar was isolated from the soil by surrounding it with sand. The sand allowed some movement near the surface while the bar remained in the ground below the freeze line (Figure 1).

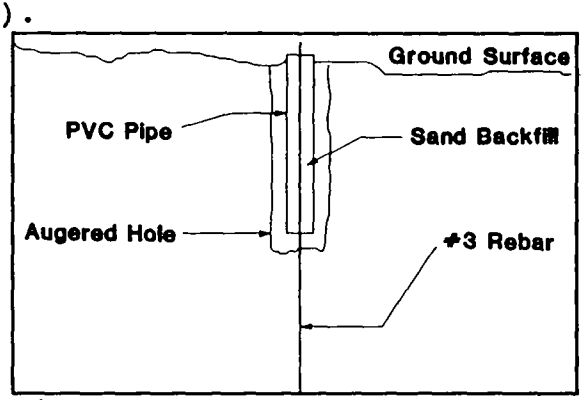

Figure 1. Survey Monument Construction.

The survey monuments were constructed of 1.5 m (5 ft), #3 steel reinforcing bar, 0.075 m (3 in.) O.D. PVC pipe, and sand. A 0.15 m (6 in) diameter hole was augered into the ground to a depth of about 0.5 m (1.5 ft). The reinforcing bar was then driven into the bottom of the hole until about 0.05 m (2 in) of the bar protruded above the surface of the ground. The PVC pipe was then placed in the hole around the bar and back-filled with tamped sand. The hole around the pipe was filled with tamped soil. The PVC pipe was then capped to isolate the sand from moisture. Lines of survey monuments were set up crossing and along the centerline of the subsidence trough. Shorter lines of monuments were also installed adjacent to test footings constructed on the center-line and on the edge of the subsidence trough.

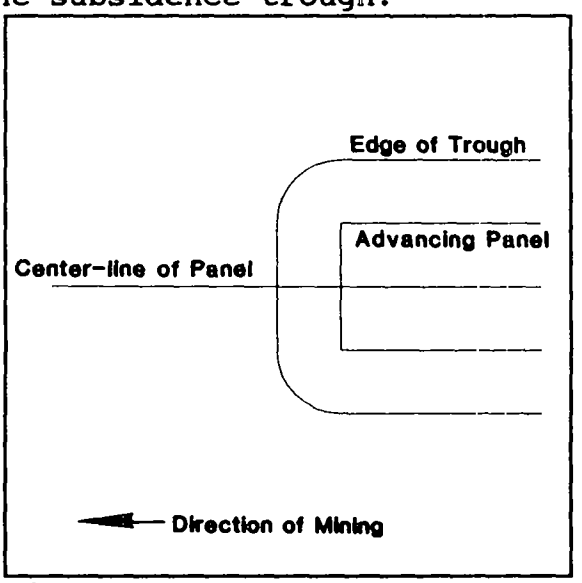

Figure 2. Layout of Survey Monuments.

To demonstrate the correction, a survey of fifteen monuments, numbered 1 through 15, located along the centerline of a subsiding longwall mining panel will be used (Figure 2). Level surveys were conducted on

surveys used to demonstrate the correction method were conducted when the rate of subsidence was at its highest. Two surveys were conducted on the day which the maximum amount of subsidence occurred.

Vertical Survey

To obtain accurate measurements, a Pentax level with a micrometer accurate to 0.0001 m (0.0003 ft) was used with a Nedo temperature invariant leveling rod. The elevation of a point is determined by sighting the leveling rod through the optical level. The reading on the leveling rod indicates the elevation difference between the bottom of the leveling rod and the sights on the level. This is known because the optical level sights the leveling rod on the horizontal plane. Each time the level is set up, the operator back sights on a point of known elevation. Since the difference in elevation of the point of known elevation and the level can be determined, the elevation of any new measurement can be determined by finding the elevation difference between the level and the new point.

The initiation of subsidence was determined to be on May 10, 1990 and the end of subsidence was determined to be on August 8, 1990. The magnitude of subsidence was determined by subtracting the elevation of a given point before subsidence from the elevation of the same point after subsidence. The maximum amount of subsidence observed along the centerline was 1.425 m (4.674 ft). Over fifty percent of the total subsidence occurred in 5 days, when the subsidence rate was also at its greatest (Bennet et al).

Conventional Error Correction

Conventionally, when a survey is conducted, a loop is closed and a closure error is calculated. Closure error for a vertical survey is determined by measuring a series of elevations consecutively to create a loop. At some point, a point in the survey is used to calculate the elevation of a point which has already been surveyed. If the second measurement differs from the first, this is considered to be the amount of error in the survey loop. In a conventional survey, the total amount of error is then distributed evenly among all the points in the loop. For example, if a loop of ten survey points is determined to have a total error of 1 cm then each elevation in the loop would be adjusted by one tenth of a centimeter.

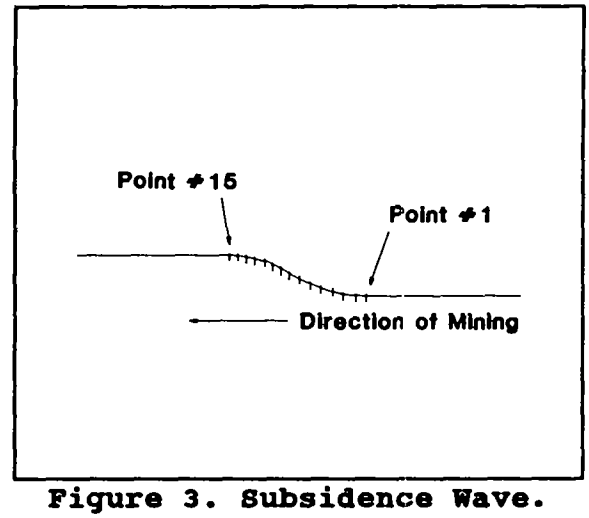

Figure 3. Subsidence Wave.

Typical Subsidence Surveys

One of the basic assumptions of conventional surveying is that all distances

remain constant and any discrepancy is due to error or lack of precision. This is not always the case when surveying mine subsidence. Most published results of subsided mine surveys were begun at day 0 and continued for months or even years after the surface was undermined or collapsed. These surveys were conducted at monthly or weekly intervals. Since most surveys of subsided areas are started after subsidence has ceased, it is correct to use conventional error correction techniques. Surveys of subsided areas commonly apply no correction at all to the survey (Peng, 1992). In this project surveys were conducted either once or twice per day during the time of the maximum rate of subsidence. Subsidence occurred in measurable amounts during the two hours required to conduct the survey. Since the ground subsided before the loop could be closed, adjusting the distances would create more error than would be corrected.

It is possible to correct for subsidence that occurs during the survey, by obtaining rates of subsidence and simulating an instantaneous survey. A correction method was tested over the center line of an advancing longwall mining trough. Subsidence induced by longwall mining is the most convenient type of subsidence to monitor, because it is known when subsidence will begin in the area above the mine.

At its maximum rate, the ground surface over a longwall mining panel can subside at about 1 cm (2.5 in) per hour (Table 1). The rate of subsidence depends on the position of a given point on the subsidence wave (Figure 3).

The problem with calculating closure error of a subsiding surface is that the survey cannot be completed before the ground moves. Closure error could be calculated on a subsiding surface only if an instantaneous survey is conducted. Since this is impractical, if not impossible with current technology, the only logical alternative is to simulate an instantaneous survey.

If a line of points is surveyed, it takes some finite period of time to conduct the survey. That finite period of time is the difference between the initial time , when the first measurement is recorded, and the final time when the final measurement is taken.

In order to know precisely what the profile shape is, one would need to know what the profile is at an instant in time. As the survey is conducted, the survey monuments subside from the initial instantaneous profile to the final instantaneous profile. The problem is that the survey spanned a finite time period and does not represent either of the true subsidence profiles.

An instantaneous profile can be simulated by calculating a rate of subsidence for every point on the survey. After the subsidence rates are calculated, all points can be projected forward or backward in time (Table 1).

Table 1. Comparison of Actual and Corrected Profiles

Point	5-23-90 AM Elev. Drop (m)	AM Time of Measure	5/23/90 PM Elev. Drop (m)	PM Time of Measure	AM/PM Delta Elev. (m)	Subsidence Rate (m/hr)
1	0.737	8:00	0.820	4:00	0.083	0.01038
2	0.667	8:04	0.756	4:04	0.089	0.01113
3	0.598	8:08	0.687	4:08	0.089	0.01113
4	0.571	8:12	0.659	4:12	0.088	0.01110
5	0.541	8:16	0.630	4:16	0.089	0.01113
6	0.515	8:20	0.601	4:20	0.086	0.01075
7	0.487	8:24	0.572	4:24	0.085	0.01063
8	0.458	8:28	0.540	4:28	0.082	0.01025
9	0.435	8:32	0.516	4:32	0.081	0.01013
10	0.409	8:36	0.487	4:36	0.078	0.00975
11	0.383	8:40	0.458	4:40	0.075	0.00938
12	0.361	8:44	0.434	4:44	0.073	0.00913
13	0.334	8:48	0.404	4:48	0.070	0.00875
14	0.278	8:52	0.340	4:52	0.062	0.00775
15	0.221	8:56	0.273	4:56	0.052	0.00650

One foreseeable source of error would be a change in subsidence rate during the time taken to conduct the survey. For this reason loops should be arranged so that completion can be achieved within as short a time span as possible.

To calculate subsidence rates in the field, the elevation of each point must be measured twice. Two surveys should be conducted on each loop with the time of each measurement also recorded. The measurement of the first point is taken at the initial time, t_1, the second at t_2, the third at t_3, etc. With two different elevations for each point at two different times, the data can be used to calculate the subsidence rate for each point. Figure 4 plots projected data points shown with the two sets of actual data used to calculate the subsidence rates. The surveys were conducted approximately eight hours apart. An eight-hour interval was used because this was the smallest interval available. Ideally one would want two surveys conducted about an hour apart to insure that the subsidence rates were relatively the same.

Figure 5 compares the curvature of the actual profiles of points 1 to 15 with the corrected profiles of the same points.

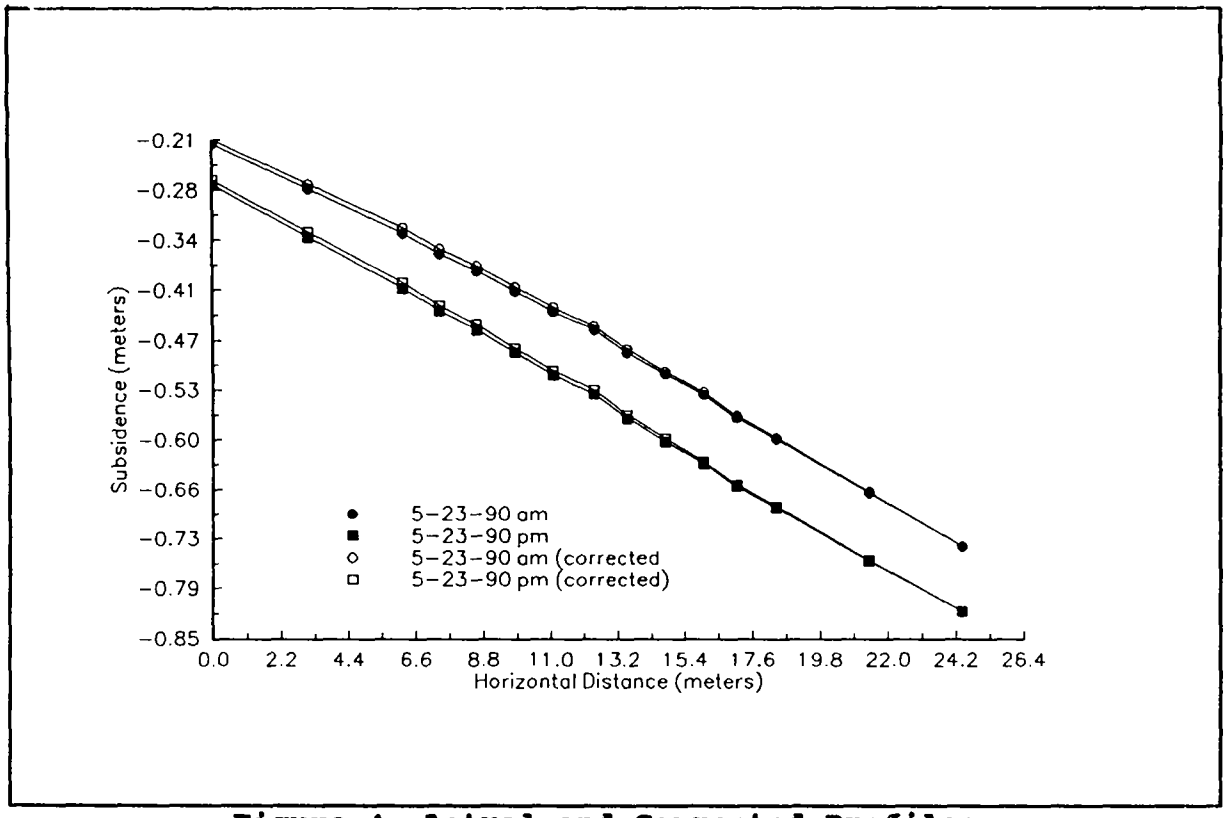

Figure 4. Actual and Corrected Profiles

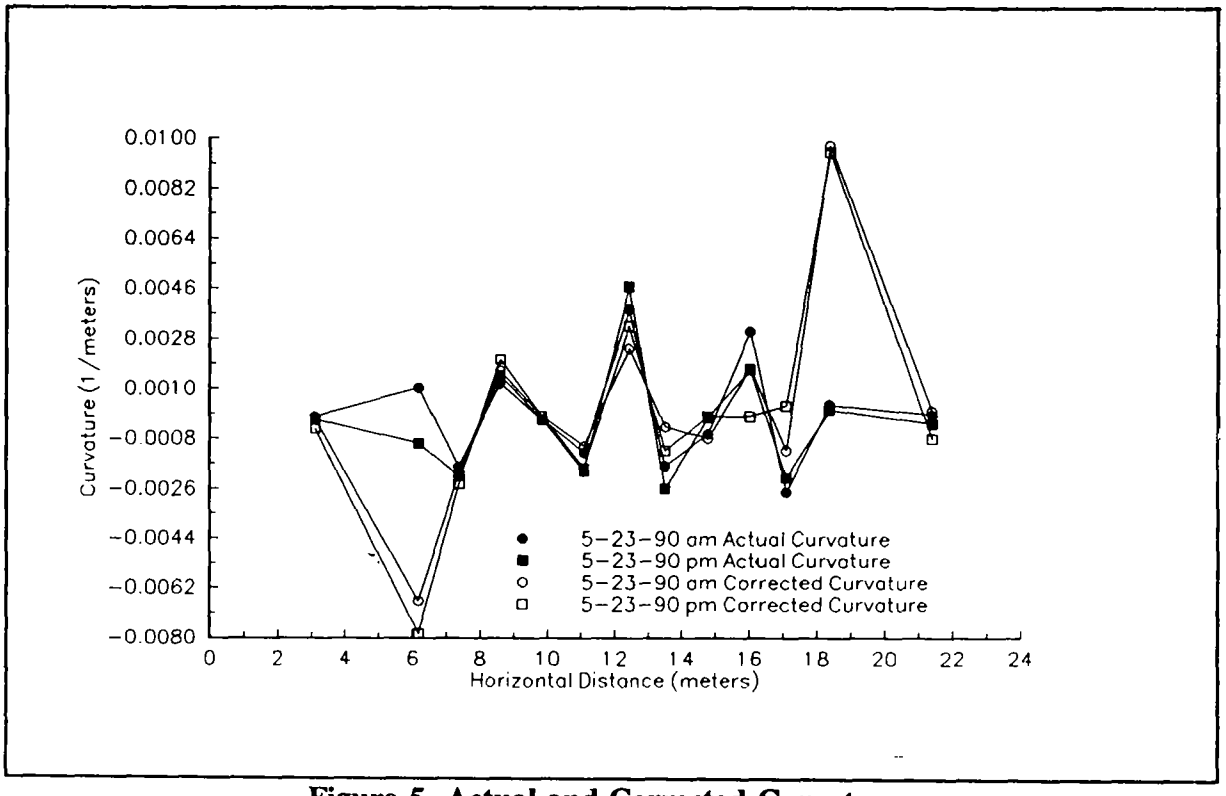

Figure 5. Actual and Corrected Curvatures

Table 2. Comparison of Actual and Corrected Curvatures

Point	AM/PM Correction	AM Corrected Elevation (m)	PM Corrected Elevation (m)	AM Curve 1/(m)	PM Curve 1/(m)
1	0.0000	0.7370	0.8200		
2	0.0008	0.6662	0.7552	-0.0002	-0.0008
3	0.0015	0.5965	0.6855	-0.0007	-0.0095
4	0.0022	0.5688	0.6568	-0.00012	0.0004
5	0.0030	0.5380	0.6270	0.00017	0.0000
6	0.0036	0.5114	0.5974	-0.0008	0.0000
7	0.0042	0.4828	0.5678	-0.0004	-0.0012
8	0.0048	0.4532	0.5352	0.0024	0.0033
9	0.0055	0.4295	0.5105	-0.0011	-0.0019
10	0.0058	0.4032	0.4812	0.0000	0.0000
11	0.0062	0.3768	0.4518	0.0016	0.0020
12	0.0067	0.3543	0.4273	-0.0020	-0.0024
13	0.0070	0.3270	0.3970	-0.0067	-0.0079
14	0.0067	0.2713	0.3333	-0.0001	-0.0005
15	0.0061	0.2149	0.2669		

The differences in curvature are substantial. Since the correction adjusts each point according to the time that elapses after the first measurement is taken, one end of the survey line is adjusted more than the other (Figure 4). The first point required no adjustment. This is the most significant aspect of this correction method. If the correction only shifted the whole line some uniform distance as done with closure correction, then it would be trivial. Since the correction is nonuniform, it deserves more attention.

Conclusions

It appears that the calculation of closure error is of little significance compared to the correction from projecting the points to time t_1. A plot of the points after calculation of closure error is indistinguishable from that of the projected points. Since the rate of subsidence can vary significantly over short horizontal distances, i.e. distances of 5 m (16 ft) or less, the survey should be conducted as quickly as possible, regardless of the correction method used.

In most cases, the most precise methods and equipment for surveying are used to monitor mine subsidence. Greater than normal care is taken to conduct the surveys required to monitor mine subsidence. A monitoring method which produces more accurate results would be beneficial. Since the profile which most nearly represents an instantaneous survey of the subsiding area is desired, this correction method should de used for surveys which have detectable amounts of movement taking place during the survey.

This method would be useful for measurement of any subsiding area or structure which subsides at a rate which would introduce measurable amounts of subsidence before the survey could be conducted. When the survey data is intended to be used for calculation of curvatures or other properties which involve the multiplication of error, the correction has the greatest significance.

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A HISTORY OF METAL SHAFT/PORTAL CLOSURES IN UTAH[1]

by

Louis A. Amodt[2]

Abstract. Over the last twelve years the Utah Abandoned Mine Reclamation Program has utilized metal shaft and portal closures when physical constraints required an alternative method of closure to backfill or masonry seals. Eight types of metal closures have been installed with varying degrees of success. Relative advantages/disadvantages of these closures are discussed. The eight closure types are 1) A-frame "bird cages", 2) large diameter cyclone fence-type grid, 3) bar grates, 4) rebar grates, 5) cable nets, 6) steel doors, 7) angle iron bat grates, and 8) "jail bar" steel bat grates. The primary application of the fabricated metal closures has been in non-coal mines. Only the angle iron bat grate has been installed in abandoned coal mines to date. Bird cages placed in avalanche zones failed and the shafts were later backfilled. Large cyclone fence-type grids were placed in the Wasatch and Tintic mountains in 1985. Rebar grates have been used in the Wasatch and Tintic mountains as an alternative to backfill. Bar grating was used in the Tintic mountains when the aesthetics of an historical headframe dictated a less visible closure. Cable nets have been installed in Canyonlands National Park utilizing a design pioneered in Death Valley National Monument. Steel doors have been utilized where the landowner requests access or a need to access the adit exists. Two types of bat grates have been utilized in both coal and non-coal mines where sensitive or endangered bat species have been identified. The bird cage design has not been as effective as the others. Indications are the rebar grates, angle iron bar grates, and steel doors excel in giving long term protection to the public, with the second generation bat grate giving the maximum protection to the public.

Additional Key Words: Shaft and Adit Reclamation, Reclamation Costs, Metal Closures, Bat Protection, Hazard Mitigation

Introduction

During the last twelve years the Utah Abandoned Mine Reclamation Program (AMRP) has been utilizing a variety of metal closures to address the problem of effectively sealing abandoned shafts and adits when standard techniques are unsuitable. Securing dangerous abandoned mines with a less than permanent seal is affected by the location and ease of accessibility to the site. The easier the accessibility, the higher the visitation and possible vandalism. Thus, the closer and more accessible the site, the stronger the design must be to deter vandalism. Sites located on steep slopes at high elevations may experience not only vandalism problems but also snow loads and possible avalanche damage.

Eight designs of fabricated steel closures have been employed by the Utah AMRP with varying degrees of success in both longevity and durability. The eight closure types are 1) A-frame "bird cages", 2) large diameter cyclone fence-type grid, 3) bar grates, 4) rebar grates, 5) cable nets, 6) steel doors, 7) angle iron steel bat grates, and 8) "jail bar" steel bat grates. The primary application of the metal closures has been in non-coal mines. Only the first generation (angle iron) bat grate has been installed in abandoned

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Publication in this proceedings does not prevent author from publishing the manuscript whole or in part, in other publication outlets.

coal mines in Utah to date.

Costs for the fabricated metal closures range from a high of $2,675.37/m² ($247.69/ft²) for a one time use bar grate closure to a low of $72.06/m² ($6.68/ft²) for installation of large rebar grate closures. A cost analysis was performed and shows a comparable cost for the metal closure when compared to the volume of backfill that would be required for filling the shaft or adit. Table 1 summarizes the costs of each closure completed by the Utah AMRP over the last twelve years.

A-Frame "Bird Cage"

The A-frame bird cages were first used in the Wasatch Mountains above Salt Lake City in 1982 and 1983 (Figure 1). The design consists of a 95 mm (3/8-inch) square mesh on 25 cm (10-inch) centers bolted to a 5 cm (2-inch) by 8 cm (3-inch) by 48 mm (3/16-inch) tubing frame on 61 cm (2-foot). The square mesh was secured by u-bolts to the framing. The bird cages were adjacent to and within the Wasatch-Cache National Forest near four ski resorts. Visitor use days within the Wasatch-Cache National Forest were in excess of 2,000,000 user days in the 1990-91 ski season alone. (Fjeldsted and Hachman 1991) The 2,000,000+ figure does not include the hiker traffic during the summer months. Many shafts in the area extend over 300 meters (1,000 feet) in depth and may be used as recharge or ventilation of the aquifer. Backfill material is limited or nonexistent at the sites.

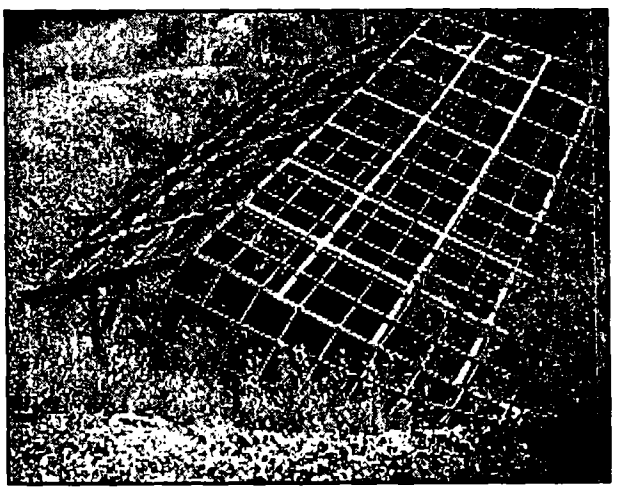

Figure 1. A-Frame "Bird Cage" Grate closure in Alta-Brighton Project, 1984.

The bird cages were designed to protect the public from entering the shafts and performed well to restrict the recreational skier or hiker. The more inquisitive public found that the square mesh could be bent back and the framing used to anchor ropes for descending down the shafts. Sloughing of the shaft collar was a problem with a loss of support for the concrete pillars or epoxy roof bolts near the edge of the collar. The bird cages were transported to the sites by ski, snow cat, and helicopter. Three of the bird cages collapsed due to heavy snow loads from avalanches. The location of the shaft in relation to potential avalanches and competence of the collar material must be evaluated when any type of closure other than backfill is utilized.

The costs for this type of closure range from $1,200 for a 1 m x 1.4 m (3.5' x 4.5') cage to $6,640 for a 13 m x 7 m (42' x 23' m) cage with an average size of 5 m x 4 m (15' x 12') cage at an average cost of $3,512 or $483.76/m² ($46.63/ft²) including the fabrication of the custom fit cages.

To date eight of 15 bird cages are still secure in the Wasatch Range. Five shafts with failed bird cages ranging in depth from 6 meters (20 feet) to 212 meters (400 feet) were later backfilled by hand. One bird cage was repaired by expanding the size of the 95 mm (3/8-inch) mesh by 3 m (10 feet).

Large Cyclone Fencing Grid

The large cyclone fencing grid was designed to withstand the heavy snow loads encountered high in the Wasatch Range, and in response to the failures seen with the bird cages under heavy loading (Figure 2). Based on the known capability of cyclone fencing to distribute the load and flex, a design consisting of 95 mm (3/8") diameter hot dipped galvanized wire twisted into a cyclone fence on 15 cm (6-inch) centers was fabricated. Agutter Engineering of Salt Lake City designed a jig to turn the 95 mm (3/8") diameter bar for the cyclone fencing in 4 m (12') lengths which were threaded together for the required length of "fencing". Where the length exceeded 4 m (12'), the sections

Table 1 - Metal Closure Costs by Site

SITE	SIZE (FEET)	SIZE (METERS)	TYPE	COST
AB-1	3.5' x 4.5'	1.07 m x 1.37 m	Bird Cage	$1,200
AB-2	6' x 8'	1.83 m x 2.44 m	"	$3,575
AB-3	6' x 10'	1.83 m x 3.05 m	"	$2,950
AB-4	6' x 5'	1.83 m x 1.52 m	"	$2,750
AB-5	10' x 10'	3.05 m x 3.05 m	"	$3,000
AB-6	42' x 23'	12.8 m x 7.01 m	"	$6,640
AB-7	17' x 14'	5.18 m x 4.27 m	"	$4,000
AB-13	26' x 8'	7.92 m x 2.44 m	"	$5,575
AB-15	13' x 17'	3.96 m x 5.18 m	"	$3,650
AB-18	15' x 15'	4.57 m x 4.57 m	"	$2,600
AB-22	5' x 8'	1.52 m x 2.44 m	"	$2,450
AB-38	6' X 8'	1.83 m x 2.44 m	"	$3,750
BB-BB	5' x 11'	1.52 m x 3.35 m	Bar Grate[3]	13,623
BB-EH	16' x 32'	4.88 m x 9.75 m	Cyclone Fence[4]	$6,245
BB-GE	24' x 34'	7.32 m x 10.36 m	"	13,007
BB-RV	44' x 36'	13.41 m x 10.97 m	"	13,781
BB-SA	32' x 38'	9.75 m x 11.58 m	"	11,136
A-17	12' x 16'	3.66 m x 4.88 m	"	$4,375
W-2	4' x 8'	6.10 m x 6.10 m	Rebar Grate[5]	$ 384
W-35	4' x 6'	1.22 m x 1.83 m	"	$ 792
W-119	4' x 4'	1.22 m x 1.22 m	"	$ 500
W-312	12' x 15'	3.66 m x 4.57 m	Rebar Grate[6]	$1,485
W-324	18' x 24'	5.49 m x 7.32 m	"	$2,376
W-350	8' x 16'	2.44 m x 4.88 m	"	$3,200
T-3	20' x 20'	6.10 m x 6.10 m	Rebar Grate[7]	$2,300
T-7	36' x 36'	10.97 m x 10.97 m	"	$8,976
T-8	13' x 16'	3.96 m x 4.88 m	"	$1,700
T-13	35' x 40'	10.67 m x 12.19 m	"	$5,400
T-14	15' x 18'	4.57 m x 5.49 m	"	$2,000
T-15	9' x 13'	2.74 m x 3.96 m	"	$1,000
T-16	24' x 24'	7.32 m x 7.32 m	"	$3,400
T-17	34' x 35'	10.36 m x 10.67 m	"	$7,100
T-19	24' x 25'	7.32 m x 7.62 m	"	$3,500
T-21	12' x 14'	3.66 m x 4.27 m	"	$2,100
T-22	20' x 25'	6.10 m x 7.62 m	"	$3,000
T-27	12' x 16'	3.66 m x 4.88 m	"	$1,092
T-28	14' x 14'	4.27 m x 4.27 m	"	$1,100
T-30	18' x 22'	5.49 m x 6.71 m	"	$2,300
T-36	32' x 35'	9.75 m x 10.67 m	"	$6,632
T-38	10' x 10'	3.05 m x 3.05 m	"	$4,000
L-H1b	7' x 6'	2.13 m x 1.83 m	Cable Net	$ 882
L-H3b	11' x 6'	3.35 m x 1.83 m	"	$1,386
L-H4a	19' x 7'	5.79 m x 2.13 m	"	$3,234
L-H4c	10' x 7'	3.05 m x 2.13 m	"	$1,995
L-H4d	7.5' x 7'	2.29 m x 2.13 m	"	$1,113
L-H4e	7.5' x 6.5'	1.98 m x 1.98 m	"	$ 903
L-HO5	10' x 8'	3.05 m x 2.44 m	"	$1,680
L-HO6	7' x 9'	2.13 m x 2.74 m	"	$1,323
L-HO7	9.5' x 7'	2.90 m x 2.13 m	"	$1,407
L-HO8	7' x 8.5'	2.13 m x 2.59 m	"	$1,260
W-67	6' x 6'	1.83 m x 1.83 m	Steel Door[8]	$ 965
W-89	4' x 4'	1.22 m x 1.22 m	"	$ 975
W-103	6' x 6'	1.83 m x 1.83 m	"	$ 975

FN-B1	10' x 6'	2.90 m x 2.44 m	Angle Iron	$4,500
FN-B3	12' x 6'	3.66 m x 1.83 m	Steel Bat	$5,400
FN-B4	10' x 6'	3.05 m x 1.83 m	Grate	$4,500
FN-F1	6' x 6'	1.83 m x 1.83 m	"	$3,000
FN-K1	13' x 8'	3.96 m x 2.44 m	"	$8,150
O-P1	9.5' x 8'	2.90 m x 2.44 m	"	$2,160
O-P2	6' x 9'	1.83 m x 2.74 m	"	$2,835
O-P3	8' x 6'	2.44 m x 1.83 m	"	$2,160
O-P4	9' x 7'	2.74 m x 2.13 m	"	$2,160
O-P5	6' x 8'	1.83 m x 2.44 m	"	$3,040
W-77[9]	5' x 7'	1.52 m x 2.13 m	"	$2,725
W-291	7' x 9'	2.13 m x 2.74 m	"	$4,975
S-3	3' x 2'	0.91 m x 0.61 m	Jail Bar Steel	$1,500
S-4	7' x 6'	2.13 m x 1.83 m	Bat Grate	$5,250
S-38	8' x 6'	2.44 m x 1.83 m	"	$7,280
S-42	3' x 6'	0.91 m x 1.83 m	"	$2,250
S-43	4' x 6'	1.22 m x 1.83 m	"	$3,000
S-46	5' x 3'	1.52 m x 0.91 m	"	$3,640
S-52	5' x 5'	1.52 m x 1.52 m	"	$3,125
S-68	5' x 4'	1.52 m x 1.22 m	"	$3,125
S-73	6' x 5'	1.83 m x 1.52 m	"	$3,750
S-78	8' x 5'	2.44 m x 1.52 m	"	$5,000
S-79	5' x 6'	1.52 m x 1.83 m	"	$4,000
S-80	4' x 6'	1.22 m x 1.83 m	"	$3,120
S-84	4' x 2'	1.22 m x 0.61 m	"	$5,000
S-85	5' x 8'	1.52 m x 2.44 m	"	$5,970
S-105	6' x 6'	1.83 m x 1.83 m	"	$4,500
S-107a	4' x 5'	1.22 m x 1.52 m	"	$2,700
S-107b	7' x 6'	2.13 m x 1.83 m	"	$5,460

[3] Cost includes concrete stabilization of collar area and repair of headframe.
[4] Cost includes fabrication and installation without cost of access improvements or site grading.
[5] Rebar grate consists of #5 rebar on 6" centers, rebar drilled into collar without the placement of a concrete grade beam.
[6] Rebar grate consists of #5 rebar on 6" centers with the placement of a concrete grade beam.
[7] Rebar grate consists of #6 rebar on 8" centers with the placement of a concrete grade beam.
[8] Steel Door closure includes concrete block placed around the steel structure to secure the door.
[9] Angle Iron Steel Bat Grate closure includes mild steel frame allowing bat grate to be opened like a gate.

Figure 2. Cyclone Fence Grid closure in Bullion-Beck Project, 1985.

were bolted together with 12 mm (½") bolts. The design was made to allow the grid to be installed over shafts with uneven collar configurations. However, more grooming of the collar was required than had been anticipated. The first cyclone grid was installed in the Alta area above Salt Lake City in 1985. The grid overlapped the collar 90 cm to 120 cm (3 to 4 feet) with roof bolts installed a minimum of 30 cm (1-foot) to anchor the grid into the bedrock. A 95 mm x 5 mm (2" x 3/8") bar strap was threaded around the perimeter of the grid for attachment of the anchor.

Later in 1985 the same type of grid was used in the Bullion Beck Project with one alteration to the design. A S12 x 22.5 I-beam was installed around the perimeter and

anchored to the dump material utilizing a 61 cm (24") anchor disk as a deadman placed 3 m to 5 m (10' to 15') back from the collar on all four sides of the grid. The anchor disks were secured to the grid with 12 mm (½") galvanized steel cable. The anchor disks were used to allow for possible failure of the collar without compromising the integrity of the closure. Site preparation for the grid required a relatively flat surface with the anchors placed in the dump material surrounding the collar. The perimeter I-beam was covered with dump material to blend the closure with the surrounding terrain. This design allowed for placement around the base of headframes without disturbing the integrity of the structures. None of the grids placed in 1985 have been compromised by collar failure or vandalism. Equipment was utilized to place the grid, I-beam, and excavate for the deadman anchors. Equipment access is required for this type of installation.

The grid size ranged from 4 m x 5 m (12' x 16') in the Alta Project to a maximum size of 14 m x 11 m (44' x 36') at the Bullion Beck Project. The average cost was $148.01/m² ($13.81/ft²) including the fabrication. The cost of site grading and covering of the perimeter of the grate with soil materials is not included in the above costs.

Bar Grate

A bar grate design was developed as a specialized application for the Bullion Beck Project in 1985 when the cyclone fence grid design was determined to be obtrusive to the visual appearance of a historic headframe (Figure 3). The wooden collar lining of the shaft was deteriorating near the main support for a 2-post type headframe. A concrete collar support was installed with bar grating sized to approximate the original dimensions of the collar of the shaft. The final design with the skip guides attached to the wooden beams around the collar gives the appearance of the original structure while securing the collar from safety hazards. The bar grate is constructed of 12 mm (½") diameter bar on 5 cm (2") centers with 25 mm (1") square bar on 61 cm (2') centers supporting the bars.

The cost of the 1.5 m by 3 m

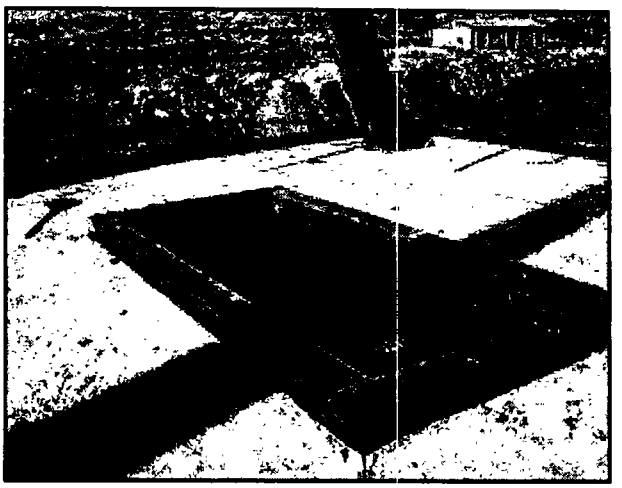

Figure 3. Bar Grate closure in Bullion-Beck Project, 1986.

(5' by 11') grate including concrete work to prepare the site and anchor the grate was $13,623.00 or an average of $2,675.37/m² ($247.69/ft²). This number is high due to the amount of concrete used to stabilize the collar before installation of the grate. The cost of the installation of the bar grating would be substantially lower if a minimum of site preparation were to be required. Equipment access was required for this installation.

Rebar Grate

Rebar grates were utilized in the Wasatch Project in 1990 and 1991 where deep shafts over 300 m (1,000') deep and a lack of available backfill material available prohibit a more permanent closure (Figures 4 and 5). A design which would require a minimum of site preparation and field fabrication was desired. The rebar grate design developed for the Wasatch Project is constructed from 16 mm (5/8") diameter rebar (#5 rebar) welded on 15 cm (6") centers. A 30 cm x 30 cm (1' x 1') concrete grade beam with two additional 16 mm (5/8") diameter rebar (#5 rebar) placed within the concrete beam was placed around the perimeter of the grate. Field fabrication and adjustments could be accomplished more easily by the contractor using rebar construction. The first rebar grate installation consisted of repair to an existing small grate by extending the rebar over the collapsing collar approximately 3 m (10'). Two of the six rebar grates were accessible only by foot and were pinned by drilling

into the bedrock collars, inserting the #5 rebar and welding the rebar at all intersections of rebar. Two rebar grates were assembled in a ski resort parking lot approximately 1.8 km (1/4 mile) below the sites. The welded grates were then transported to the sites by helicopter along with concrete and water for the perimeter grade beam. The rebar grate design used in the Tintic Project, modified to withstand higher vandalism risk, is constructed from 25 mm (1") diameter rebar (#8 rebar) welded on 20 cm (8") centers. A 30 cm x 30 cm (1' x 1') concrete grade beam reinforced by two additional 19 mm (3/4") diameter rebar was placed around the perimeter of the grate.

The grate sizes within the Wasatch Project range from 1.2 m x 1.2 m (4' x 4') to a maximum size of 6 m x 7 m (18' x 24'). The average cost of $273.22/m^2 ($25.42/ft^2) allows for variations in site access ranging from vehicle access in moderate terrain to foot only access in very steep terrain. The larger sites within the Wasatch project had an average cost of $138.88/m^2 ($12.92/ft^2) which reflects a reduced cost for volume pricing for large grates. The cost of minor site grading and revegetation of disturbed areas are not included in the above costs.

The grate sizes in the Tintic Project, utilizing the larger grate design, range from 3 m x 4 m (9' x 13') to a maximum size of 10 m x 12 m (35' x 40'). The average cost of $72.03/m^2 ($6.68/ft^2) reflects costs for sites that are equipment

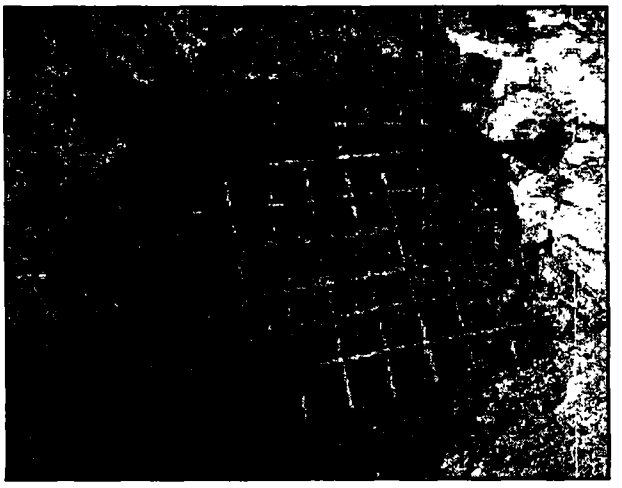

Figure 4. Pinned Rebar Grate closure in Wasatch Project, 1990.

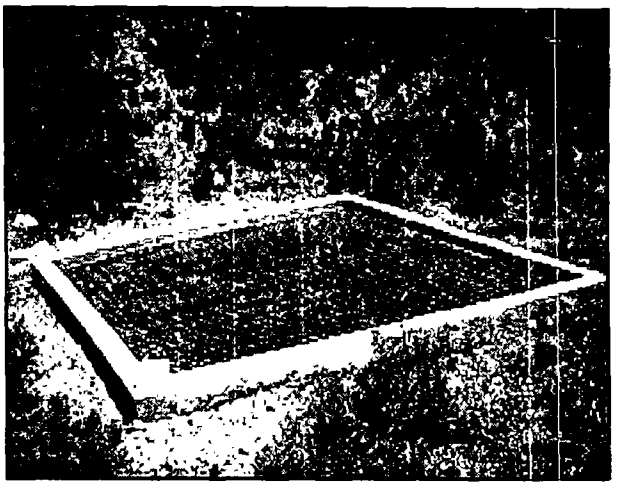

Figure 5. Rebar Grate closure in Tintic Project, 1992.

accessible. One site required transport of all materials by foot giving an average cost of $429.99/m^2 ($40.00/ft^2). The cost of minor site grading and revegetation of disturbed areas is not included in the above costs.

Cable Net

The cable net closure was utilized in the Lathrop Canyon Project in Canyonlands National Park in 1990 where the National Park Service (NPS) requested this type of closure (Figure 6). The cable nets used are fashioned after the design pioneered by the NPS in Death Valley National Monument in California. The design incorporates the use of a single length of preformed 7 x 19 construction (7 strands of 19 wires each) 64 mm (1/4 inch) diameter galvanized aircraft cable to form the grid. A perimeter cable of 79 mm (5/16 inch) of the same type construction is threaded around the grid and through rings secured to the portal wall by 46 cm (18") resin bolts. A lock box was bolted to the rib to allow access into the portals by the NPS. The contractor fabricated the nets at his facility from a jig of his own design. This design allows for small discrepancies in measurements of the portal and allows for irregularities in the ribs to be easily secured. Equipment access is preferred for installation of the nets due to the requirement of drilling the collar for placement of the resin bolts. However, no equipment access was allowed on this project by the NPS resulting in all materials and

equipment transported to the sites by foot.

The net sizes within the Lathrop Project range from 2 m x 2 m (7' x 6') to a maximum size of 6 m x 2 m (19' x 7'). The average cost of $238.88/m² ($22.17²/ft) included transporting all materials to the site by foot. The cost of revegetation of disturbed areas is not included in the above costs.

Steel Door

Steel doors have been used to a minor degree by the Utah AMRP. These doors were installed in the Wasatch Project in the Wasatch Mountains above Salt Lake City in 1990 and 1991 (Figure 7). Steel doors have only been used in non-coal applications. The doors were installed at the request of landowners for access for probable mineral development or access to water supplies. The steel door construction includes the installation of a solid concrete block wall to secure the door to the portal configuration. A 15 cm x 10 cm (6" x 4") angle iron is mounted to the block wall with a steel gate of 38 mm (1-1/2") diameter pipe frame. A 25 mm x 25 mm (1" x 1") sheet of expanded metal was welded at 15 cm (6") intervals over the pipe frame and secured to the angle iron with heavy duty hinges. The purpose of using an expanded metal cover rather than a solid metal closure was to allow for ventilation of the adit. A lock box installed on the gate allows for access to the adit.

Figure 6. Cable Net closure in Laythrop Canyon Project, 1990.

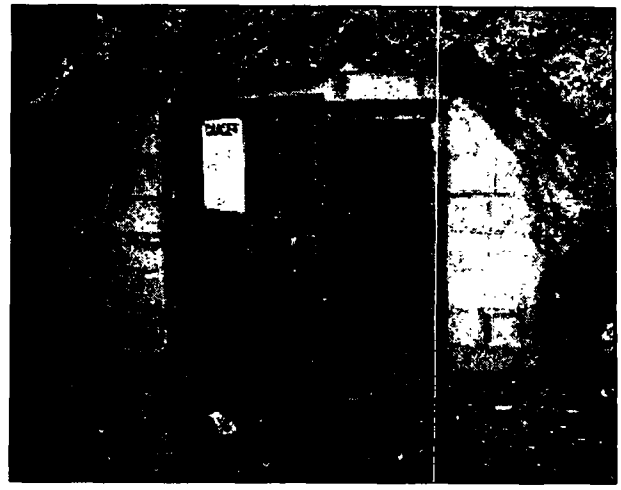

Figure 7. Steel Door closure in Wasatch Project, 1990.

The steel door sizes used in the Wasatch Project range from one small door 1.2 m 1.2 m (4' x 4') installed with foot access over an inclined shaft to larger doors 2 m x 2 m (6' x 6'). The cost of the small door with foot access was $655.07/m² ($60.94/ft²) and is high due to the single site located in steep terrain. The cost of the larger door with equipment access was lower at $289.65/m² ($26.95/ft²) due to vehicle access and a larger size. The cost of revegetation of disturbed areas is not included in the above costs.

Angle Iron Steel Bat Grate

Angle iron steel bat grates were used in the Ferron North Project in 1992, the Oyler Project in 1993, and the Wasatch Project in 1994 (Figure 8). The angle iron steel bat grate design used by the Utah AMRP is a combination of designs pioneered by Roy Powers of Mountain Empire Community College, Virginia, and is presently being used by eastern and western states. Modifications of early designs included hanging the angle iron in front of vertical supports rather than butting the angle iron into the vertical supports. The bat grates are constructed of 12 mm x 12 mm x 79 mm (4" x 4" x 5/16") angle

iron with two 38 mm x 38 mm x 64 mm (1-1/2" x 1-1/2" x 1/4") angle iron stiffeners welded inside of the horizontal bars. The horizontal bars are either welded or bolted with 12 cm (½") carriage bolts to 10 cm x 10 cm x 79 mm (4" x 4" x 5/16") square steel tubing posts. The steel posts are anchored to the roof of the adit by 16 mm (5/8") resin roof bolts placed a minimum of 20 cm (8"). The horizontal bars and base of the posts are anchored to the ribs and floor of the adit by concrete grout. The design of bolting the bars to the post simplifies field installation and allows one or two of the bars to be secured with lock boxes to allow entry into the adit. The 15 cm (6") spacing between bars allows bats to enter and exit freely while restricting public access.

The angle iron steel bat grate sizes in the Ferron North Project range from 2 m x 2 m (6' x 6') to a maximum of 4 m x 3 m (13' x 8'). The cost of $831.60/m² ($77.34/ft²) is high due to helicopter access required to deliver the materials to the sites. The sites in the Oyler Project located within Capitol Reef National Park range in size from 2 m x 2 m (6' x 8') to a maximum of 3 m x 2 m (9.5' x 8') with an average cost of $481.06/m² ($44.71/ft²). The sites in the Wasatch Project range in size from 1.5 m x 2 m (5' x 7') to a maximum of 2 m x 3 m (7' x 9') with an average cost of $844.66/m² ($78.50/ft²). The cost of revegetation of disturbed areas are not included in the above costs.

Figure 8. Angle Iron Steel Bat Grate closure in Wasatch Project, 1994.

During the late summer of 1994, one of the angle iron steel bat grate installed in the Wasatch Project was vandalized by cutting of the horizontal angle iron steel support with a hacksaw. The site located in Big Cottonwood Canyon to the east of Salt Lake City, was repaired by hard-facing all angle points exposed on the outby side of the bat grate. In the summer of 1995, the site was again vandalized by the removal of a portion of the lock-box mechanism on the grate. This vandalism was accomplished by reaching through a 64 mm (¼-inch) space between the lock box and frame with a hacksaw blade. The mild steel used in the angle iron steel bat grate is easily breached by vandals armed with hacksaws and time.

"Jail Bar" Steel Bat Grate

Jail bar steel bat grates were been used in the Summit Project in 1995 (Figure 9). The "jail bar" steel bat grate design used by the Utah AMRP is an attempt to thwart the vandalism of the mild steel angle iron bat grates utilized in previous projects. The bat grates are constructed of 25 mm (1") diameter solid manganese steel bar with two or more 12 mm x 10 cm (½" x 4") manganese steel strap vertical supports. All components are made of 12-14% manganese steel. The vertical supports have 25 mm (1") diameter holes cut on 16 cm (6-1/2") centers with the horizontal bars either electric welded or brazed to the vertical support. The vertical supports are anchored to the roof of the adit by 25 mm (1") diameter 12-14% manganese bars placed a minimum of 20 cm (8") and anchored with resin. The base of the vertical supports are anchored to the floor of the adit by concrete grout. The design and installation of a lock box to the vertical support allows one of the bars to be removed allowing entry into the adit. The manganese steel may be rough cut off-site and cut to fit onsite utilizing a cutting torch. The 14 cm (5-1/2") spacing between bars allows bats to enter and exit freely while restricting public access. Modifications of this design were warranted when a small child (6 years

old) was observed crawling between the lower bars of the closure. The modification consists of reducing the spacing of the bars to 11 cm (4-1/2") on bars located below 1.5 m (60") from the floor of the adit. The spacing of 14 cm (5-1/2") is maintained above the 1.5 m (60") threshold. This allows for one or two spacings of 14 cm (5-1/2") for the average adit. After the 1995 construction season, bats were observed in adits known to have no bat use prior to the installation of the bat grates. The design of the jail bar steel bat grate will be modified to a 10 cm (4") spacing below 1.2 m (48") threshold for all grates after the 1995 season. This change is in response to a change in the national building code standards for constructed gates in public areas.

The jail bar steel bat grate sizes in the Summit Project range from 1 m x 0.6 m (3' x 2') to a maximum of 2.4 m x 1.8 m (8' x 6') with an average cost of $1,582.37/m^2 ($147.06/ft^2). The average cost is increased to $1,884.90/m^2 ($175.18/ft^2) when including a small site (S-84) requiring helicopter access to deliver the materials. The cost of revegetation of disturbed areas is not included in the above costs.

Conclusions

Over the last twelve years the Utah AMRP has used a variety of metal closures with varying degrees of success. The AMRP closure process has been a dynamic evolution as different closure methods are tried and tested.

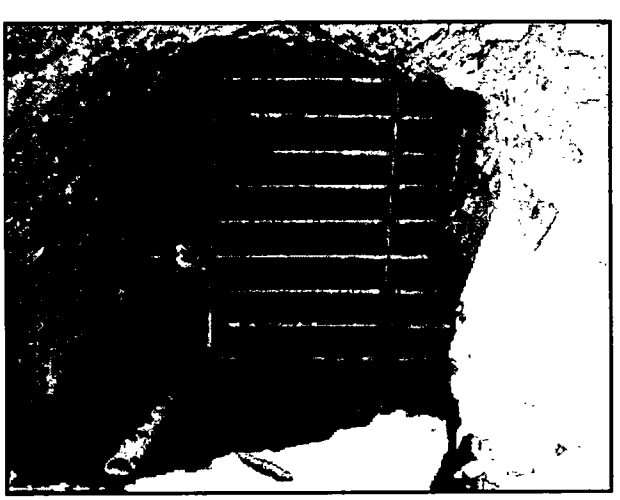

Figure 9. "Jail Bar" Steel Bat Grate closure in Summit Project, 1995.

The A-frame bird cage was the first attempt for the AMRP to find a closure method which would be both cost effective to secure the opening and withstand minor vandalism. With this design, the custom fit, offsite fabrication, difficult installation, and transport to high elevations drove the cost up. The cage is susceptible to failure under snow loads, collar failure, and vandalism.

The large cyclone fencing gives protection for minor collar failure while allowing ventilation, visibility for shaft inspection, and snow loading. The fencing does require moderate site preparation, offsite fabrication and transport to the site. A specialized jig is required to turn the bar stock for fabrication of the fencing design. To date vandalism of the fencing has not been a problem. For large sites with unstable collar configurations the cyclone fencing closure would be the best fit to provide maximum security.

The bar grate provides site ventilation, security, and high loading capacity. Equipment, offsite fabrication, large volume of concrete reinforcing of the collar, and a high amount of site preparation for the single use by the AMRP gave the bar closure an inflated cost for the project. The bar grate provides a secure closure with a high visual compatibility to the historic headframe above the shaft.

The rebar grate gives a secure closure while allowing ventilation, high loading, and visual inspection of the shaft. Minimal site preparation is required allowing the grate to be placed on slopes and uneven collar configurations. The grate is constructed of widely available rebar with onsite fabrication and modifications easily made during assembly without compromising the integrity of the structure. Equipment access to the site reduces the cost significantly due to the volume of rebar and concrete required for the installation. A relatively competent collar configuration is required to anchor the structure while the configuration and flexibility of the grate would allow for minor collar failure. For large, deep shafts with equipment access the grid is a very cost effective way to achieve security. Though configured differently than bat grates, there is anecdotal evidence suggesting bat

utilization of this design in the Tintic Project area in the spring of 1993.

The cable nets offer security while allowing ventilation of adits with a minimum of site preparation. Equipment access to the site significantly reduces costs and the cable net's light weight and capability to be rolled increases the ease of transport. The nets are less durable than the other closures requiring some maintenance to guarantee the security of the closure. The lock is still the weak link in the system with the closure only as secure as the lock. Offsite fabrication, difficult field alterations, and possible vandalism lower the effectiveness of this closure method. The National Park Service applications of this closure technique provide low visibility and minimal impact to the surrounding area. Where the closure can be monitored, this method works well.

Steel doors offer moderate security while allowing controlled access to workings. The steel doors constructed of expanded steel are susceptible to vandalism. The doors must be sized for the opening and then secured to the walls and roof of the mine with block and grout. As with the cable net closure, the lock placed on the door is the weak link in the system. Equipment access to the site significantly reduces the cost of the closure method. In Utah, the doors have only been installed at the request of landowners. This method works well and is cost effective in locations where the doors can be monitored.

The angle iron steel bat grate design offers ventilation of the working along with controlled entry. The bars are fabricated offsite but can be transported in manageable pieces and assembled onsite. Some designs require detailed cutting and welding for installation. This design has minimal effect on the ventilation and a maximum flyway for bats. The grate also gives a secure closure with a higher cost than other methods with equipment access. This method may be acceptable in locations of bat sightings where infants or small children would not be present. In Utah, footprints and other indications of visitation by small children are found in even the most remote locations.

The "jail bar" steel bat grate design also offers ventilation of the working along with controlled entry and vandalism resistance. The bars may be cut offsite with final fit onsite. This allows for the transport of the grate in manageable pieces to be assembled onsite. This design allows for welding by electric or gas with detailed cutting and welding onsite. The grate has a minimal effect on the ventilation and a maximum flyway for bats. The grate also gives a secure closure with a lower cost than angle iron steel bat grates. This method is the preferred closure in locations of bat sightings where infants or small children may be present.

Table 2 summarizes the closures by type, cost, advantages, and disadvantages

Of the eight fabricated metal closures used by the AMRP, the most successful in terms of cost, effectiveness and closure security have been the jail bar steel bat grates.

References

Fjeldsted, B.L. and Hachman, F. C. 1991. Results of the 1990-1991 Utah Skier Survey. Utah Economic and Business Review, Vol. 51, 16 P, August/ September 1991.

Table 2 - Fabricated Metal Closures by Type

TYPE OF CLOSURE	COST $/ft^2$	COST $/m^2$	ADVANTAGES	DISADVANTAGES
A-Frame "Bird Cage"	$46.63	$483.76	Ventilation, possible re-entry	Custom fit, easily vandalized, low snow loads, offsite fabrication, no protection if collar fails
Large Cyclone Fencing	$13.81	$148.01	Ventilation, secures shaft, high loads, bar stock widely available, shaft visible for inspection, collar failure not a problem	Equipment required, offsite fabrication, specialized jig required, site preparation
Rebar Grates	$ 6.68[10] $40.00[11]	$ 72.03[10] $429.99[11]	Ventilation, secures shaft, high loads, minimal site preparation, fabrication onsite, rebar available and inexpensive, field alterations easy	Equipment required, cut with hacksaw
Bar Grates	$247.69	$2,675.37	Ventilation, secures shaft, high loads	Equipment required, offsite fabrication, site preparation
Cable Nets	$22.17	$238.88	Ventilation, secures shaft/adit, minimal site preparation, light weight, transport ease	Custom fit, offsite fabrication, less durable than others, field alterations difficult
Steel Doors	$26.95[10] $60.94[11]	$289.65[10] $655.07[11]	Ventilation, controlled entry, minimal wildlife use	Offsite fabrication, vandalism
Angle Iron Steel Bat Grates	$44.71	$481.06	Ventilation, controlled entry, stock materials, allows bat and other wildlife use, onsite assembly	Offsite fabrication, equipment required, some detailed cutting and welding, cut with hacksaw

"Jail Bar" Steel Bat Grates	$147.06[12] $175.18[13]	$1,582.37[12] $1,884.90[13]	Ventilation, controlled entry, stock materials, allows bat and other wildlife use, onsite assembly	Limited availability of material, gas and/or electric welder required, some detailed cutting and welding

[10] Average site cost for equipment accessible.
[11] Average site cost for single site or foot access to site only.
[12] Average site cost for closure.
[13] Average site cost increased by helicopter access to remote site (S-84).

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APPLICATION AND ANALYSIS OF ANCHORED GEOSYNTHETIC SYSTEMS FOR STABILIZATION OF ABANDONED MINE LAND SLOPES

by

Stanley J. Vitton, M. Frank Whitman, Wendell W. Harris, and Robert Y. Liang

Abstract. An anchored geosynthetic system (AGS) was used in the remediation of a landslide associated with an abandoned coal mine located near Hindman, Kentucky. In concept, AGS is a system that provides in-situ stabilization of soil slopes by combining a surface-deployed geosynthetic with an anchoring system of driven reinforcing rods similar to soil nailing. Installation of the system involves tensioning a geosynthetic over a slope's surface by driving anchors through the geosynthetic at a given spacing and distance. By tensioning the geosynthetic over the slope's surface, a compressive load is applied to the slope. Benefits of AGS are described to include the following: (1) increase soil strength due to soil compression including increased compressive loading on potential failure surfaces, (2) soil reinforcement through soil nailing, (3) halt of soil creep, (4) erosion control, and (5) long term soil consolidation. Following installation of the AGS and one year of monitoring, it was found that the anchored geosynthetic system only provided some of the reported benefits and in general did not function as an active stabilization system. This was due in part to the inability of the system to provide and maintain loading on the geosynthetic. The geosynthetic, however, did tension when slope movement occurred and prevented the slope from failing. Thus, the system functioned more as a passive restraint system and appeared to function well over the monitoring period.

Additional Key Words: slope stability, landslide remediation, geotextiles.

Introduction

A significant problem in abandoned mine lands is the stabilization of unstable slopes. According to Iannacchione et al., (1994), the Abandoned Mine Lands program, administered by the Office of Surface Mining (OSM), has spent approximately $64 million remediating 425 landslides between 1979 and 1992, at an average cost of $152,000 per landslide. In Kentucky alone, $45 million was spent during this period on 268 landslides.

In an attempt to reduce the cost of landslide remediation as well as to provide more efficient alternatives for landslide remediation, several new systems have been considered. One of the systems considered is an anchored geosynthetic system (AGS), which was proposed by Koerner (1984, 1985, 1986a) and Koerner and Robbins (1986c) in the mid-1980s. To evaluate this system an AGS was installed and monitored on a landslide associated with an abandoned mine site near Hindman located in Eastern Kentucky (Figure 1).

The basic function of an AGS is to provide active stabilization of the slope through tensioning a geosynthetic over a slope using ground anchors as illustrated in Figure 2. As the soil beneath the

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[3]Publication in this proceedings does not prevent the authors from publishing their manuscripts, whole or in part, in other publication outlets.

geosynthetic deforms, membrane stresses develop in the tensioned geosynthetic and impart a compressive load onto the slope, which increases the stability of the slope. According to Koerner, anchorage of the geosynthetic is achieved with small diameter, ribbed steel rods (rebar) that are driven into the soil using hand held tools such as a vibropercussion hammer. The anchors are driven on a prescribed grid pattern through the geosynthetic, generally at right angles to the ground surface, to approximately 75 to 90% of their designed depth. The geosynthetic is then fastened to the anchor and the anchor is driven the remaining distance, thereby tensioning the geosynthetic and creating a curved geosynthetic-soil interface as the soil deforms the soil below the geosynthetic. This tension and curvature imparts compressive stress to the soil and an uplift loading on the anchor. According to Vitton (1991), this compressive stress, σ_{nr}, which is applied to the soil from the geosynthetic through membrane action, is directly related to the tension, N, in the geosynthetic (developed by the driving force of the anchor) and inversely related to the radius of curvature, r_c, of the geosynthetic-soil interface (from the deformation of the soil beneath the geosynthetic) as follows

$$\sigma_{nr} = \frac{N}{r_c} \qquad (1)$$

Koerner (1986a) lists the immediate and long term benefits of utilizing AGS for slope stabilization as follows:

1. Immediate slope stabilization due to bending and shearing resistance of the anchors.
2. Immediate halt to long term slope creep.
3. Immediate stabilization of the potential failure zone due to an increase in compressive stresses on the potential failure plane.
4. Immediate erosion control of the slope surface.
5. Increasing slope stability in the long term due to consolidation of cohesive soils and the densification of cohesionless soils from the compressive loading on the soil.

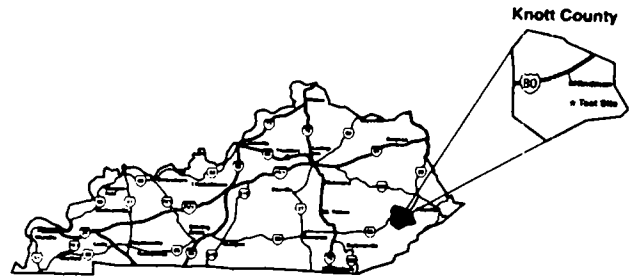

Figure 1. AGS project site location map.

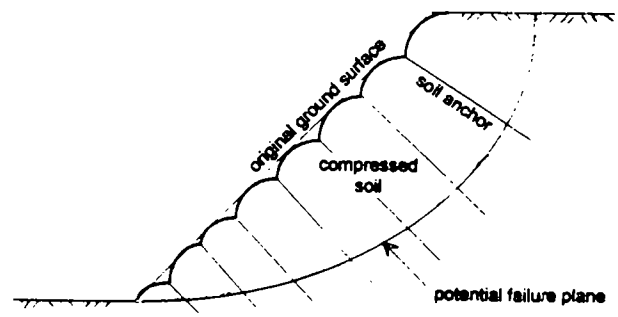

Figure 2. Cross-section of an anchored geosynthetic system.

However, as noted by Koerner, the system may require additional anchor redriving after the initial installation to maintain tension in the geosynthetic due to soil consolidation and stress relaxation in the geosynthetic. In addition to these benefits, AGS also eliminates the need for heavy construction equipment to repair the slope, resulting in substantial cost savings.

To date, only limited field research has been conducted on AGS. As previously stated, the original design concept was developed by Koerner (1984, 1985), who also performed the first theoretical AGS slope stability analysis (Koerner and Robins, 1986). Other theoretical work was done by Hryciw (1991, 1992), who calculated the optimum length and orientation of soil anchors in cohesionless soils, and by Greenwood (1985), who analyzed a soft clay slope that was remediated with a geogrid connected to duck-billed anchors.

Two field installations were completed by Koerner (1986) using a woven slit-film geotextile. This fabric was fitted with grommets on a 1.5 m (5 ft) triangular pattern and anchored to the slope using 13 mm (0.5 in) diameter steel rods. These rods were 1.2 m (4 ft) long and were coupled together to create total anchor lengths of 1.2 m (4 ft) to 7.3 m (24 ft). These lengths were varied so as to penetrate the potential failure plane by at least a meter. This design was used to stabilize a 10.7 m (35 ft) high, 60° slope of silty sand and a 4.5 m (15 ft) high, 50° slope of silty clay. Both slopes remained stable for one year, but no information has yet been published on the long-term effects of the installations.

Vitton's (1991) research, however, revealed that for cohesionless soil, the deformation of the geosynthetic-soil interface is limited to an area immediately around each anchor. This and the development of interface frictional forces on the geosynthetic limit the compressive load that can be applied to the slope. Vitton also noted that stress relaxation of the geosynthetic limits the amount of time that the system is in tension.

Research Site

The research site selected is located in Eastern Kentucky and is approximately 1.5 km south of Hindman, Kentucky in Knott County as shown in Figure 1. The landslide lies approximately 230 m (750 ft) downslope from the Fire Clay (Hazard Number Four) coal bed, which was mined in the early 1970's. A 3 ha (7 acre) landslide occurred in the mine spoil in May of 1984, sliding downslope approximately 275 meters (1000 ft) into the center of a hollow and onto a county road at the base of the hollow. The failure was remediated by the Kentucky Office of Surface Mining (OSM) as an Abandoned Mine Land (AML) project called the Madden Slide. Due to the volume of the spoil, some of it was kept at the site in an uncompacted head-of-hollow fill. In constructing the fill, part of an adjacent slope, which had not been disturbed by the original landslide, was cut back to allow for a drainage ditch to be constructed around the head-of-hollow fill. The undercut slope eventually failed in 1988 and was remediated in 1991. The slide failed again in 1993. The main cause of the instability was groundwater seepage, which was observed in at least three locations within the landslide. Due to the nature of this slide and size of the adjacent landslide, this landslide was selected for the installation and evaluation of the AGS. A cross-section of the slide is shown in Figure 3 while a plan view of the landslide area is shown in Figure 4.

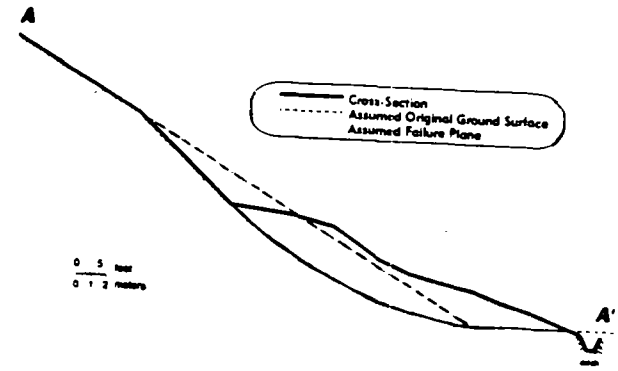

Figure 3. Cross-section of the landslide prior to remediation.

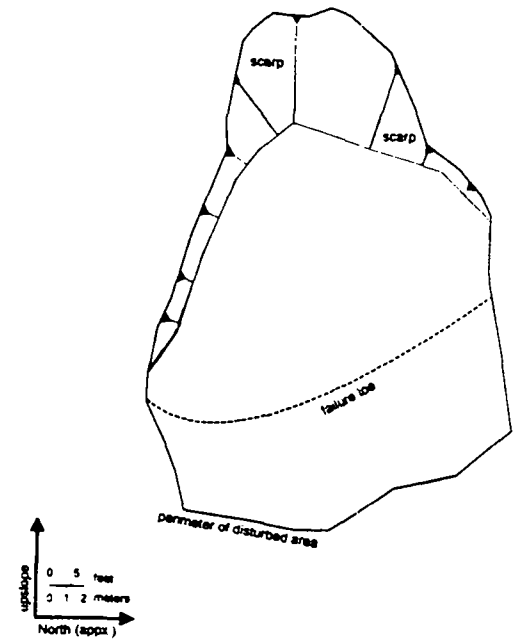

Figure 4. Plan view of landslide area.

The slope in which the slide occurred consists of colluvial soils formed from the overlying rock formations and containing significant sandstone fragments. The USCS classification for

the soil is ML, a sandy silt. However, the Atterberg limits did plot very close to the hatched zone of the plasticity chart indicating a silt-clay mixture. The liquid limit was found to vary between 25 and 35 with an average value of approximately 28. The plastic limit varied between 17 and 25 with an average value of approximately 22. The corresponding plasticity index was approximately 6, indicating a material of low plasticity. As expected the moisture content of the surface soils varied by season. Average moisture contents in the spring were typically around 30%, while in the fall the moisture content decreased to an average value of 16%. These water content values indicate a liquidity index of greater than one during the spring and less than zero during the fall. The softer response in the spring coincides with the increase of water content when a slope failure occurred.

Consolidation tests were conducted since soil consolidation will result when the compressive load from the geotextile is applied to the slope's surface. It was found that the coefficient of consolidation, c_v, averaged 15 m^2/yr while the virgin compression index, C_c, averaged 0.132 and the recompression index averaged 0.025. The preconsolidation pressure, which was difficult to determine, was estimated to be approximately 100 kPa (2100 psf) for soil samples from an undisturbed area adjacent to the slide at a depth of 0.6 m (2 ft). This indicates that the colluvial soils were overconsolidated. Although the past maximum pressure is given with some reservation due primarily to the disturbance during sampling, it does indicate that the soils are overconsolidated probably due to the wetting and drying cycles that develop the desiccated crust.

To estimate the shear strength of the colluvial soil, a series of consolidated-undrained shearing triaxial compression tests and unconsolidated-undrained compression tests were performed on shelby tube samples taken from the site. The drained strength parameters were found to be as follows; the effective angle of internal friction, ϕ', was approximately 33°, while the effective cohesion, c', was approximately 3.5 kPa. The undrained shear strength, S_u, was approximately 23 kPa (1100 psf). This would indicate a soil of medium consistency. However, the undrained shear strength would be expected to be significantly less during the wetter spring months when the natural water content was higher.

System Design and Installation

The general design of the AGS installed followed the original system proposed by Koerner (1984) with modifications as suggested by Vitton (1991). The as-constructed system, however, was based primarily on site conditions and dimensions and on anchor driving tests. Unexpected developments in the field and accumulated experience necessitated changes in the original AGS design during actual installation. Due to delays, the installation of the system was completed over a period of four months, although the total installation time was only approximately two and half weeks using an average of four people. The components of the AGS consisted of (1) geotextile, (2) anchors, and (3) a geotextile-anchor connection. The following section provides the selection and design of each these components and the installation procedure used for the AGS.

Geotextile

Since a high to moderate strength geotextile is required for an AGS, only woven geotextiles were considered for this project. Other fabric properties of concern were stress relaxation, ultraviolet (UV) radiation stability, fabric construction, and stress-strain characteristics. The fabric selected for installation was Linq's (formally Exxon) GTF-1000T, a woven polyester geotextile. This fabric has a tensile strength of 160 kN/m (925 lb/in) in the warp direction and 140 kN/m (800 lb/in) in the fill direction. The fabric strength selected was based on the pullout resistances of the ground anchors in field tests at the site in Kentucky. According to Van Zaten, (1986), if the tension in a polyester geotextile is kept below 60% of its ultimate strength the problem of creep and stress relaxation in the geotextile is minimized. Therefore, to minimize creep and stress relaxation, the strength of the geotextile selected was such that when the loading from the ground anchor approached 60% of the strength of the geotextile the anchor

pullout capacity would be exceeded and the anchors would start to pullout. In addition, this allows for a continuous load on the soil in the case of slope movement, as noted by Vitton (1991).

It was decided that one 15 m (50 ft) by 24 m (80 ft) sheet of geotextile plus two cut sections of one 3.8 m (12.6 ft) by 30 m (100 ft) roll would be sufficient to stabilize the site. The polyester geotextile was sensitive to UV radiation, so to reduce the exposure of the geotextile to the sun, a thin sheet of a nonwoven Linq's Typar was placed over the entire installation and secured to the tops of the anchors and simply acted as a sacrificial material.

Anchors

Informal driving tests at the University of Alabama demonstrated that anchors could be driven sufficiently by a Hilte TE804 electric demolition hammer powered by a 4000 watt generator. The demolition hammer was used to strike a custom-made driver head threaded over the protruding end of the anchor as shown in Figure 5. As a contingency, extra driver heads were made to fit a 13 N (60 lb) pneumatic jackhammer that was rented, along with a compressor, if necessary.

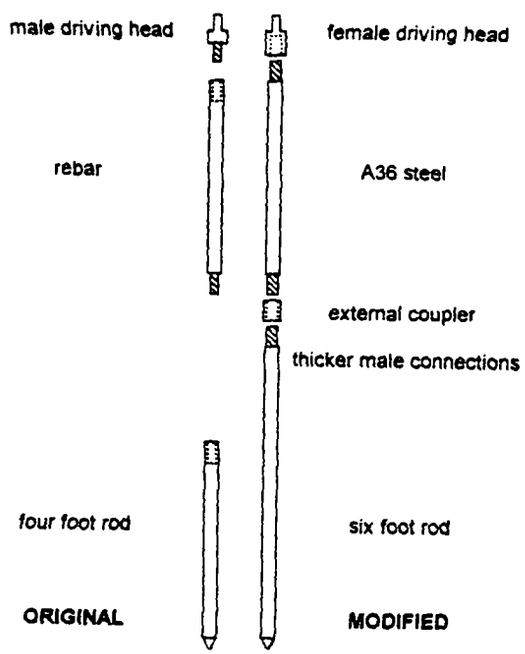

Figure 5. Anchor pattern used in the AGS.

Two anchor materials were considered: rebar and an A36 cold-rolled steel. Driving tests revealed that the rebar was too brittle for dynamic driving. The extreme vibrations caused by the impacts of the demolition hammer often caused the rods to twist apart or break. The cold-rolled steel rods, on the other hand, were more ductile due to the higher quality of steel. Therefore the rolled steel, despite being more expensive, was chosen over the rebar as the anchor material.

Since it is important to have the AGS anchors driven deep enough to penetrate the potential failure plane of a slope, a total anchor length of 3.9 m (13 ft) was chosen for the site based on an estimated maximum depth to the original failure plane of about two meters (7 ft). The total anchor assembly length consisted of four rods: (3) 1.2 m (4 ft) lengths and (1) .3 m (1 ft) length. The 0.3 m section was the last rod to be driven, and was threaded along its entire length to facilitate the assembly and tightening of the anchor-geotextile connection.

Early in the anchor testing process, the test rod lengths were manufactured with threaded male and female ends as illustrated in the left-hand portion of Figure 5. These rods could simply be twisted together without the need for any type of fastener or coupler. Driving tests showed that these unreinforced connections were vulnerable to shearing failure during driving. The new test rods were made with both ends turned down and threaded. These rods were connected by an external coupler as shown in the right-hand portion of Figure 5. This design proved sturdy, however, the jackhammer's dynamic driving action, which resulted in considerable vibrational movement of the top portion of the anchor, caused the anchor hole to enlarge at the surface. While this allowed room for the coupler, which had a somewhat larger diameter, to be driven into the soil, it also minimized the pullout resistance of the anchor since it prevented all but the first rod of the anchor assembly from having good contact with the soil. To minimize this problem, the first rod used in driving was increased from 1.2 m to 1.8 m (6 ft) in length as shown in Figure 5. The last section in the assembly was machined to a length of 0.6 m (2 ft) to maintain an

overall anchor length of 3.9 m.

Rod diameters of 13 mm (0.5 in.), 19 mm (0.75 in.), and 25 mm (1 in.) were tested at the University of Alabama to determine the relative pullout resistances of the three different sizes. These tests were conducted in a field on campus slated for construction, so good geotechnical information was available. The anchors were tested next to a Standard Penetration Test (SPT) hole that had N values ranging from 20 near the surface to 70 at resistance in a gravel layer at a depth of about 4.5 m (15 ft). The soil was a stiff clay with gravel lenses. Anchors of each size were driven to resistance with the Hilte TE804, then pulled out of the ground using a hydraulic lift or a forklift.

The 13 mm diameter rods tended to wander during driving, creating a hole that was not straight. This added to their pullout resistance, but the larger diameter rods had higher overall pullout resistance due to the increased soil-interface area. The demolition hammer was unable to effectively drive the 25 mm diameter anchor as deeply as the other anchors and were not considered further.

The 13 mm and 19 mm diameter rods were then field-tested at the research site in Kentucky using the Hilte TE804. Pullout results showed a significant increase in resistance for the 19 mm diameter anchors over the 13 mm rods and therefore 19 mm diameter rods were used in the installation.

For the remediation, the anchors would be driven 1.4 m (4.6 ft) apart in a hexagonal pattern as in Figure 6. Anchor driving was to begin in the center of the slide and proceed outward as shown in Figure 7.

Anchor-Geotextile Connections

The entire anchor geotextile connection assembly used in the installation is shown in Figure 8. The anchor geosynthetic connection cup was pressed out of one foot diameter, 14-gage steel circles to form bowls approximately 130 mm (5 in.) in maximum diameter and 50 mm (2 in.) deep. A 22 mm (0.875 in.) hole in the center of each of the connections allowed them to be placed on the .3 m, threaded section of the anchors. A flat 50 mm diameter area at the bases provided a contact surface for belleville springs. These springs were placed on the anchors between the anchor geotextile connections and tightening bolts in an attempt to maintain tension

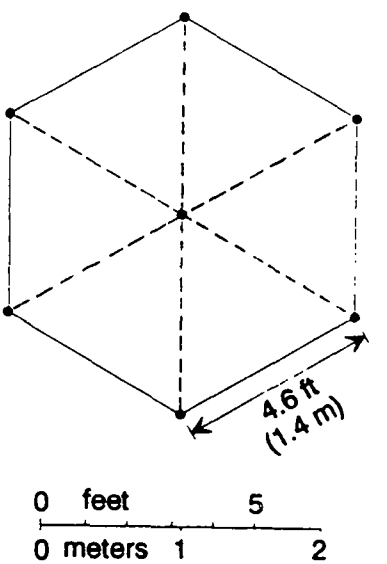

Figure 6. Hexagonal anchor arrangement.

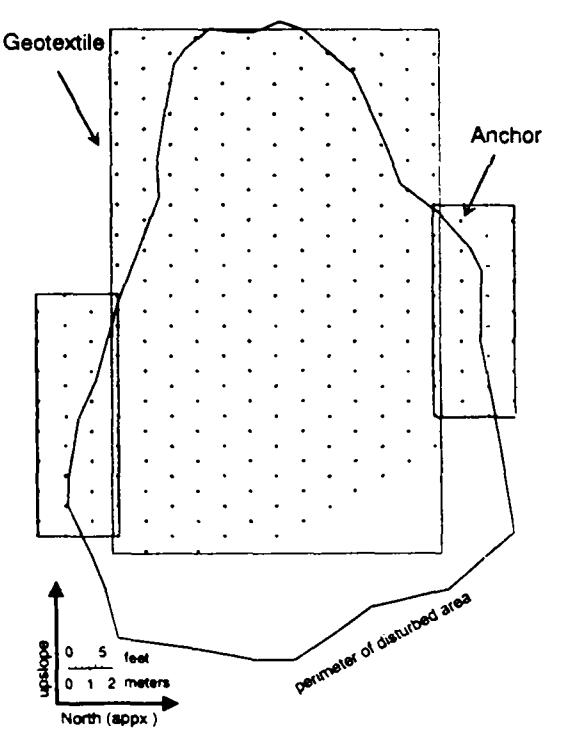

Figure 7. Anchor installation plan.

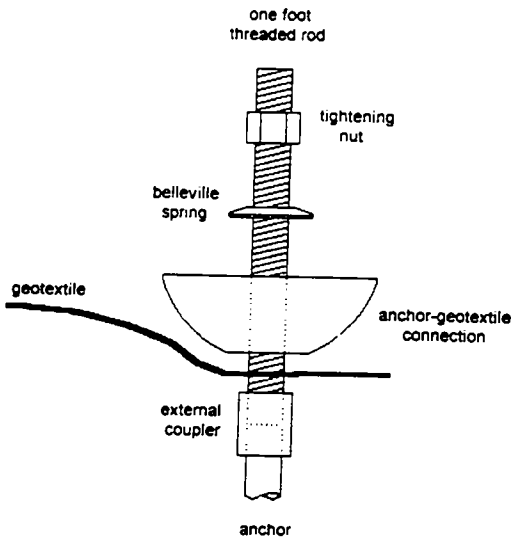

Figure 8. Anchor-geotextile connection.

on the geotextile in the event of geotextile creep or anchor movement. The tightening bolts were used to tension the AGS after anchor driving was completed and at later dates.

Installation

Installation of the AGS was started on December 1, 1993. A bulldozer was first used to placed the failed soil back up slope and to provide a more level surface for placement of the geotextile. Since some soil had been removed from the landslide in 1991, the bulldozer was unable to match the original volume and grade of the slope but did manage to adequately place the remaining material back up slope. It was very difficult in the upper portions of the slide due to very wet ground conditions, which at times were very soft. However, a relatively level surface was created in which to apply the AGS. Due to the smaller than expected area to be remediated, only the 15 m by 24 m geotextile was needed to secure the slope. Access to the landslide was limited, especially at the top of the slope, so it was necessary to manually place the fabric. Four people were able to lift the geotextile over a ditch, unroll it up the slope, and drag it into final position, though this was accomplished with difficulty.

As anchor driving began, all three available driver heads for the Hilte demolition hammer failed. At this time only seven anchors had been driven. A pneumatic jackhammer and associated driver heads were available, but considerable rain had begun to fall and the forecast for the next several days was poor. The installation was temporarily abandoned. Possible reasons for the unforeseen increase in driving resistance that caused the failure of the driving heads are the use of couplers, the unexpectedly shallow depth of soft fill over the stiffer, unfailed layer, and possibly the colder temperatures encountered in December as opposed to the temperatures in June when the field tests were conducted.

The first attempt to complete the AGS installation took place in March of 1994. This followed a very wet and snowy winter in which a portion of the slope beneath the geotextile, which had been placed in December, failed. Fortunately, however, it appeared that the seven anchors driven in December prevented this portion of the slope from completely failing downslope.

To complete the AGS installation a 13 N (60 lb) pneumatic jackhammer and diesel compressor was used in place of the electric demolition hammer and generator. The jackhammer was generally adequate for driving the ground anchors. The extensive sandstone fragments in the colluvium, however, caused difficulty in driving the anchors. Frequently, when the anchors encountered the sandstone fragments the anchor would either force the fragment out of the way or would penetrate through the fragment, which was indicated by difficult anchor driving followed by relatively easy anchor driving, i.e., the anchor would break through the fragment. In two cases, the anchors met resistance and could not be driven further. It was also observed that when an anchor hit a significant sandstone fragment, excessive vibrations would develop in the anchor causing additional enlargement to the anchor hole and reducing the anchor's pullout capacity.

The soil conditions below the geotextile ranged from firm at the base of the slide to extremely wet near the top of the slide. In some areas, even walking on the geotextile caused significant deformation of the geotextile. However, even in the very wet areas driving of the anchors resulted

in little to no general deformation of the soil between the anchors. Only very local deformation in the vicinity of the anchor occurred, which was generally within 30 cm or less of the anchor.

A total of four people completed the installation. One person worked ahead of the jackhammer laying out the anchor pattern using a fabricated triangular template which was 1.4 m (4.6 ft) on each side. By using this template the hexagonal anchor pattern could be properly positioned. This person would slide the anchor through the fabric making sure that the geotextile was not ruptured and then drive the first rod section as deeply as possible using a simple post-hole driver.

Two people were required to lift and operate the jackhammer due to the difficulties of working on a steep slope with a heavy jackhammer. A third person assisted by screwing on the driver head, guiding the jackhammer onto the head when it was lifted, and holding the driving head in place during driving since the head would often vibrate loose and damaging the rod threads if constant tightening wasn't maintained.

Because the depth of relatively soft, failed material was less than anticipated, anchor refusal during jackhammer driving occurred at depths less than the designed anchor length of 3.9 m. Most anchors in the completed system could only be driven 2.7 m (9 ft), while some anchors in the upper third of the AGS could only be driven 2.1 m (7 ft). The upper right corner of the AGS was unsecured because ground resistance in that area prevented the 1.8 m pointed rods from being driven completely, so the anchored geotextile connection could not be attached.

All but two of the anchors penetrated the shallow failure plane of the slope, though not by the 1.5 m (5 ft) suggested by Koerner (1990). Due to the difficulties in driving and the smaller area to be remediated, only 172 anchors were used to secure the slope as shown in Figure 9 then the planned 225 anchors.

Field Performance

Monitoring of the AGS included the use of anchor load cells, soil pressure gages, a rain gage, and a temperature probe. The purpose of these instruments was to record the responses of the AGS and the soil to the slope remediation process and changes in the weather. The site was also surveyed on a regular basis to detect any slope movement, and test anchors were driven at the base of the site to quantify any improvements in anchor pullout resistance over time. A Campbell Scientific CR10 datalogger was placed at the site in May of 1994 to monitor the instrumentation of the system. However, vandalism and power failures destroyed some of the data which was collected over several periods and totaled about three months throughout the monitoring period.

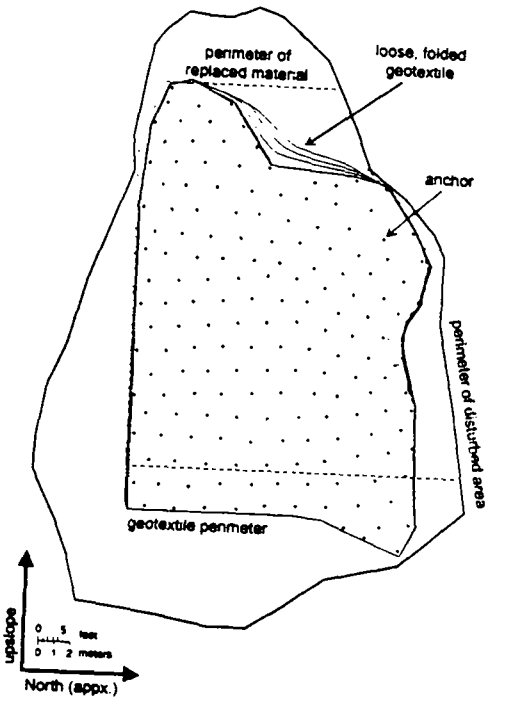

Figure 9. Completed AGS anchor layout.

A significant concern on anchor installation was that since the soils consisted of fine-grain materials, the initial pullout capacity of the anchor would be limited due to possible excess pore pressures build up during anchor driving, which would have to dissipate to achieve expected pullout capacity. Therefore, tensioning of the system was designed to be accomplished by the tightening bolt on each anchor sometime after the initial driving and not by the initial driving as suggested by Koerner (1984).

Nine load cells were installed to measure the tension in the anchors. This was done to determine the load that the anchors placed on the geosynthetic and whether or not this tensile load could be maintained over time. The load cells were fabricated from .3 m (1 ft) lengths of 19 mm diameter threaded steel as illustrated in Figure 10. After installation of the load cells, the assembly was tightened to tension the anchor and geosynthetic. The force in the anchor was recorded during and immediately after the tightening as well as for long-term changes.

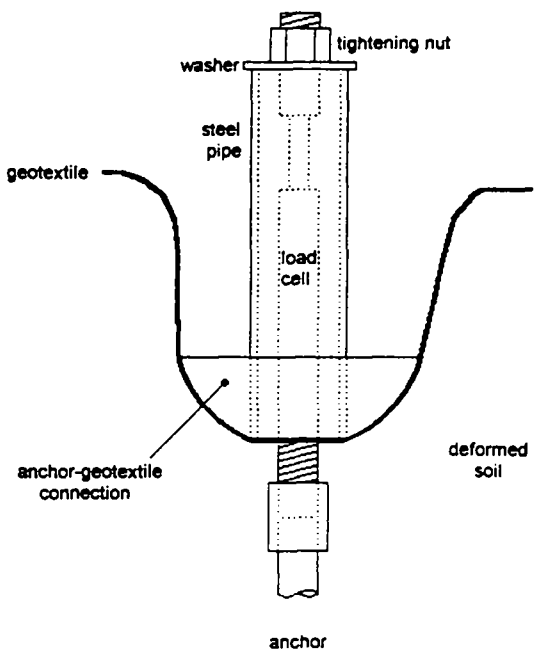

Figure 10. Anchor load cell design.

All load cells recorded an anchor load of between 110 N (500 lb) and 340 N (1500 lb) immediately after tightening. This load decayed to zero in ten to twenty days. An example of this is shown in Figure 11, where the loss in load occurs within 12 days. The load loss was believed to be due to geotextile relaxation or to local soil displacement, which resulted from soil consolidation in close proximity to the anchors, as opposed to pullout of the anchors, since anchor displacement measurements were made during tensioning that indicated little to no displacement of the anchor occurred at the time of loading. However, since the slope consisted of fine-grained soils it was also possible that soil creep along the length of the anchor could be responsible for the load loss.

After the AGS was completely installed, each anchor was tagged and numbered and surveyed using a laser theodolite. The site was first surveyed on July 11, 1994 and on an approximate two month intervals through July of 1995. The surveying showed negligible to no anchor movement for the entire monitoring period.

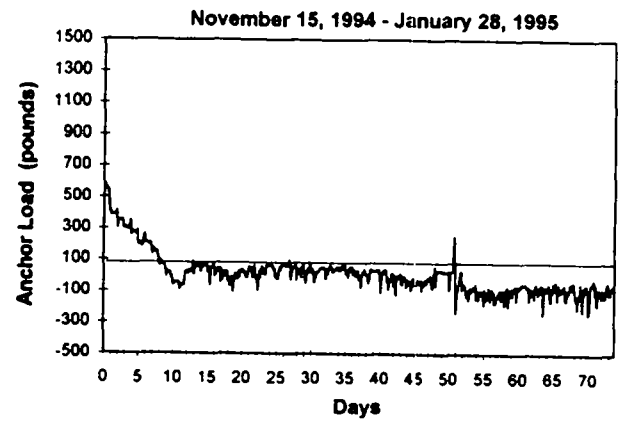

Figure 11. Anchor load versus time plot.

Anchor load changes in response to wet or dry periods were of interest as this could indicate potential slope movement in wet seasons or a shrink/swell phenomena, but no such changes were observed. However, increases of up to several hundred pounds were recorded during particularly cold weather. These spikes occurred in response to a sudden drop in air temperature to about -15° C. Closer inspection revealed trends of higher loads during colder weather and lesser loads during warmer spells. These changes are probably due to the thermal expansion and contraction of the imbedded steel anchors rather than any changes in the soil itself. The thermal changes in the steel also probably cause the daily fluctuations in load cell readings. In general, the loads reach a maximum overnight and quickly drop in the morning. This is consistent with the suspected thermal expansion and contraction of the anchors.

Before the placement of the geotextile, five soil pressure gages were installed approximately 0.6 m (2 ft) below the ground surface at various points along the slope. Three gages were

located beneath the geotextile and two gages were located below the geotextile perimeter and were not influenced by the AGS. All anchors were driven (the system was completely installed) before the datalogger was available to read the soil pressure gages. Therefore, it is not known if any pressure increases occurred during the tensioning of the geotextile over the slope. This is an important deficiency, as soil pressure increases would indicate possible soil consolidation, a major purported benefit of an AGS. However, two gages were relatively near load cell positions, where anchors were tightened on September and November of 1994. They recorded no apparent increase in soil pressure due to anchor tightening. This is consistent with the lack of deformation between the anchors, which is required to transfer the tension in the geotextile to a compressive load on the soil. While tension may exists in the geotextile, if no curvature develops through soil deformation then little to no compressive loading of the soil will result. For all of the anchors driven, only local deformation developed around the anchor with little to no deformation developing between the anchors. Therefore, there was insufficient curvature, r_c, or long term tension, N, in the geotextile to develop a compressive loading to the soil. While, all gages did show some increase in pressure due to rainfall events, the increases were small (typically less than 35 kPa (5 psi)), and were usually caused only with significant rainfall. It was concluded from the measurements made during this research, therefore, that the active stabilization of a slope by tensioning of the geotextile over the soil slope was not successful. However, based on the AGS's field performance, it is believed that other functions of the AGS did improve slope stabilization such as the effect of soil nailing from the anchors, halt of slope creep movement, and erosion control.

Conclusions and Comments

An abandoned mine land landslide located in Eastern Kentucky was remediated using an anchored geosynthetic system. Installation of the system took approximately two and half weeks utilizing four installation personnel although equipment failure and poor weather extended the installation to a period of four months. A high strength geotextile was used as the membrane and was anchored using steel ground anchors. The driving of the anchors was accomplished using a 13 N (60 lb) pneumatic jackhammer. The AGS was monitored using load cells, soil pressure cells, along with temperature and precipitation measurements over a period of about one year. Conclusions from this installation are as follows:

1. Although portions of the slope were very wet and soft, anchor driving was unable to deform the soil below the geotextile and thus minimal compressive loading was applied to the slope since limited curvature of the geotextile-soil interface developed.
2. Tensioning of the geotextile by the ground anchors was lost within ten to twenty days of application. The loss of load was believed to be a combination of soil consolidation immediately below the anchor-geotextile connection where most of the deformation occurred, and stress relaxation in the geotextile. Consequently, even if sufficient deformation developed, constant retensioning of the geotextile would be required.
3. While the AGS was not capable of functioning as an active remediation system, it did appear to function well as a passive remediation system, i.e., it prevented the slope from failing, which was in a relatively unstable state especially during wet periods, although the monitoring period was not sufficient to confirm that the system would function well into the future.
4. Driving of the anchors in the colluvial soil was difficult at times due to the presence of sandstone fragments. It was also observed that in the driving process the anchor hole would be enlarged due to the vibrations induced in the anchor. Therefore, the top portion of the anchor was not in contact with the soil. This effect was further enhanced when sandstone fragments were encountered at depth and the anchor vibrations significantly increased as the anchor attempted to break through the rock

fragment. An additional consideration on anchor driving is the build up of excess pore pressure, which reduces the pullout capacity of the anchor and makes tensioning the system difficult during anchor driving.

5. The affect of the ground anchors acting as soil nails appears to be effective but was not quantified in this research. Also, the geotextile as expected provided erosion control for the slide area.

While the results of this research are mixed, it is believed that AGS may provide an effective remediation system for certain types of slopes prone to creep. Further research, however, is needed to determine the ability of AGS to halt progressive slope failure for slopes consisting of heavily overconsolidated soils that exhibit strain-softening behavior leading to creep failure. Since colluvial soils commonly consists of these types of soil, it is believed that anchored geosynthetic systems may provide a possible remediation system for these types of slopes.

Acknowledgment

This research was sponsored by the United States Bureau of Mines under the Abandoned Mine Lands Program and administered through Contract No. 1432-J0230005 titled "The Application of Anchored Geosynthetic Systems for Abandoned Mined Land Landslide Remediation." In particular, the authors would like to thank Mr. Tony Iannacchione, the technical project officer for his support, encouragement, and timely advise throughout the project. The financial and moral support are gratefully acknowledged.

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MAXIMUM SAFE SLOPE ANGLES FOR PROPOSED SHIPLOCKS OF THE THREE GORGES DAM SITE BASED ON KINEMATIC ANALYSES PERFORMED ON MAJOR DISCONTINUITIES

by

Jeong-gi Um[2], Pinnaduwa H. S. W. Kulatilake[3], Jianping Chen[4], and Jianren Teng[5]

Abstract. The Three Gorges Project has been proposed to harness the great hydrologic power potential of the Yangtze River, which is the largest river in China. The depth of rock excavation needed to construct the permanent navigation structures (shiplocks) is high, with a maximum of 170m in some locations. The granitic rock mass which exists in the shiplock region contains a number of major discontinuities and about four sets of minor discontinuities. One hundred and thirty three major discontinuities have been mapped around the shiplock covering an area of 1740×600m. Kinematic analyses were conducted using the major discontinuities to estimate maximum safe cut slope angles with respect to plane sliding, wedge sliding and toppling failure. Kinematic analysis for plane sliding has resulted in maximum safe cut slope angles greater than 65° for most of the discontinuities. For most of the wedges, maximum safe cut slope angles greater than 45° were obtained. Maximum safe slope angles greater than 85° were obtained for most of the discontinuities in the toppling case. It seems that the shiplock faces in the proposed permanent shiplock region in fresh rock are stable up to a cut slope of about 45°. However, it is important to keep in mind that this conclusion is based on the kinematic analyses performed using only the major discontinuities. Further kinematic as well as kinetic analyses are recommended incorporating minor discontinuities, water forces, earthquake forces, etc., before making the final conclusions about maximum safe cut slope angles for the shiplock region.

Additional Key Words: Rock, Slope Stability.

Introduction

The Yangtze River is the largest river in China. From its headwaters to its estuary, the Yangtze River meanders over 6,300 km (Fig. 1) and its annual runoff into the sea amounts nearly 1,000 billion m^3 (Ha 1993). Its total drop is more than 5,800 m with a water power potential of up to 268,000 Mw. In order to harness the river and develop its water resources, extensive efforts in investigation, planning, design and scientific research have been conducted for nearly forty years. The Three Gorges Project has been proposed in order to exploit the great hydroelectric power potential of the Yangtze River. The project has tremendous multi-purpose benefits such as flood control, electric power generation, navigation, irrigation, tourism and fishery enhancement.

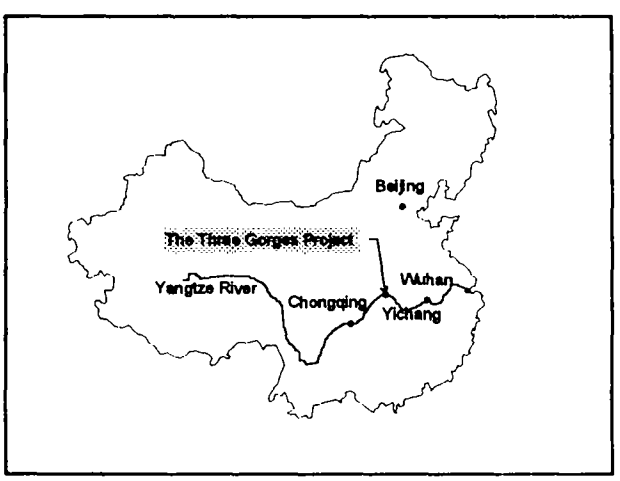

Figure 1. Geographical positions of Yangtze River and Three Gorges project in China.

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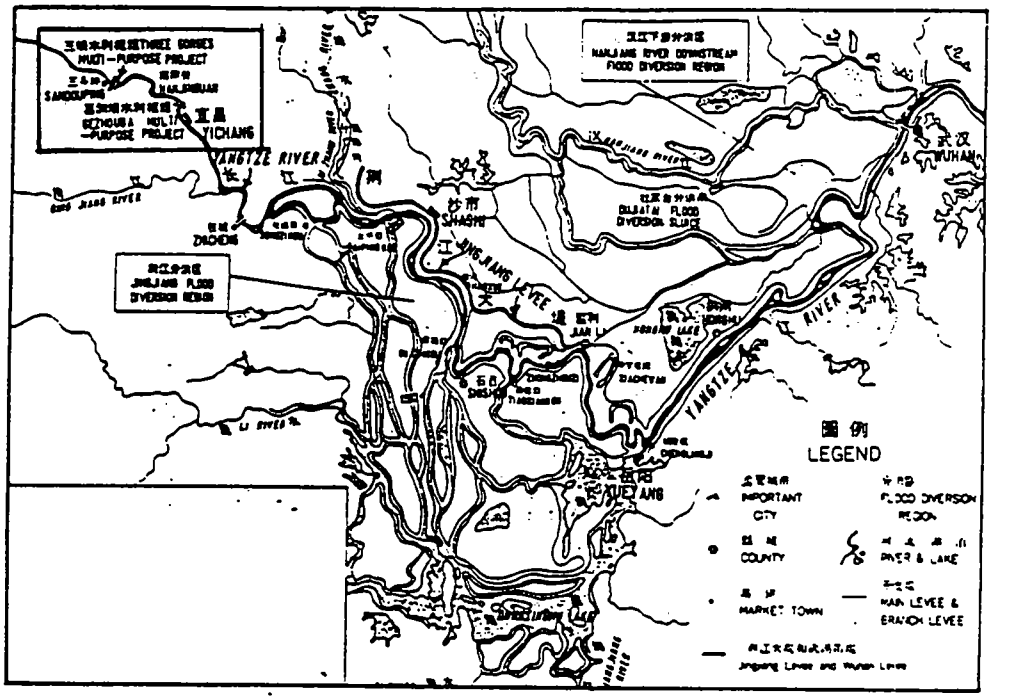


Figure 2. Locations of the planed Three Gorges Project and the completed Gezhouba project on the Yangtze River, China.

The river stretches from Fengjie to Yichang, about 200 km, and cuts through three majestic canyons, Qutang Gorge, Wuxia Gorge and Xiling Gorge - known as the Three Gorges. The Three Gorges Project is located in the middle of Xiling Gorge, about 40 km upstream of the Gezhouba water conservancy project which was completed in 1988, with its dam site at Sandouping in Yichang, Hubei Province (Fig. 2). The construction of Gezhouba dam was done not only with the aim of gaining the immediate benefits from electricity generation and navigation improvement, but also as a rehearsal for the construction of the proposed Three Gorges Project. The China Three Gorges Project Development Corporation, established in 1984, is in charge of the construction of the gigantic project. The Yangtze (Changjiang) Water Resource Commission (formally Yangtze Valley Planning Office) is in charge of the investigation, planning and design of the project (Chen 1986).

The project is composed of concrete dams, flood discharging facilities, power houses, sluice outlets, and navigation structures (Fig. 3). The concrete gravity dam will have a crest length of 2000m and a maximum height of 175m. The expected total reservoir storage capacity is 30.93 billion m^3. The normal water depth of the three Gorges reservoir is expected to be around 175m. The total generating capacity of the power stations will be about 18,200 Mw. The permanent navigation facilities will be located on the left bank, consisting of a five-stage double-lane shiplock (Fig. 4) with effective dimensions of 280 x 34 x 5 m (length x width x water depth) and a one-way shiplift with effective dimensions of 120 x 18 x 3.5 m. During the construction period, a temporary one-way shiplock is scheduled to be built (Fig. 3). The project construction began in October 1993. The total estimated construction time for the project is 18 years. For further details related to the project construction, the reader is referred to the report by Ha (1993). The dip direction (with respect to North) of the downstream axis of the permanent shiplock is 111 degrees. The depth of rock excavation needed to construct the permanent shiplocks is high with a maximum of 170 m in some locations. Operational

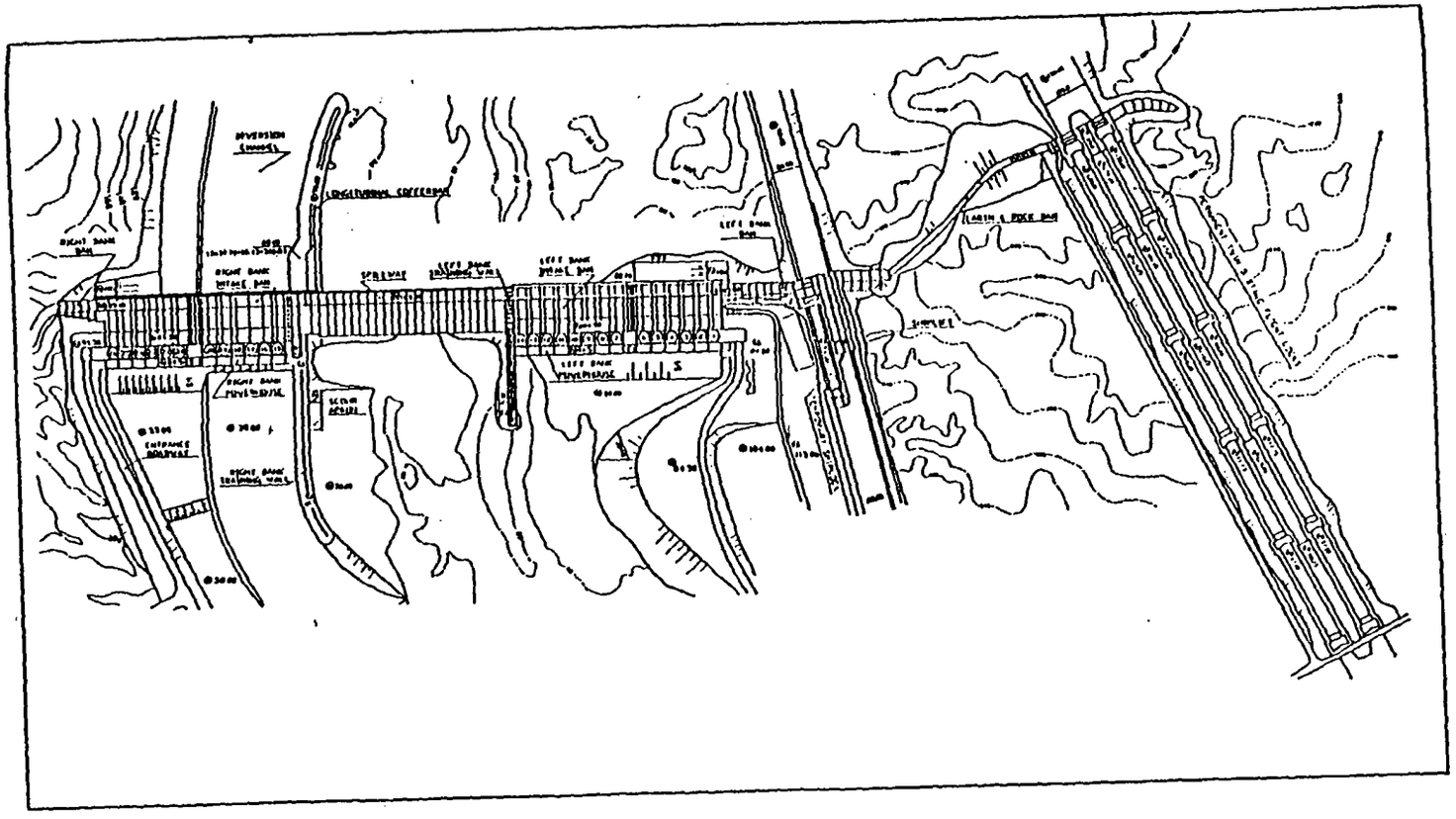


Figure 3. Layout of the Three Gorges Project [after Ha(1993)].

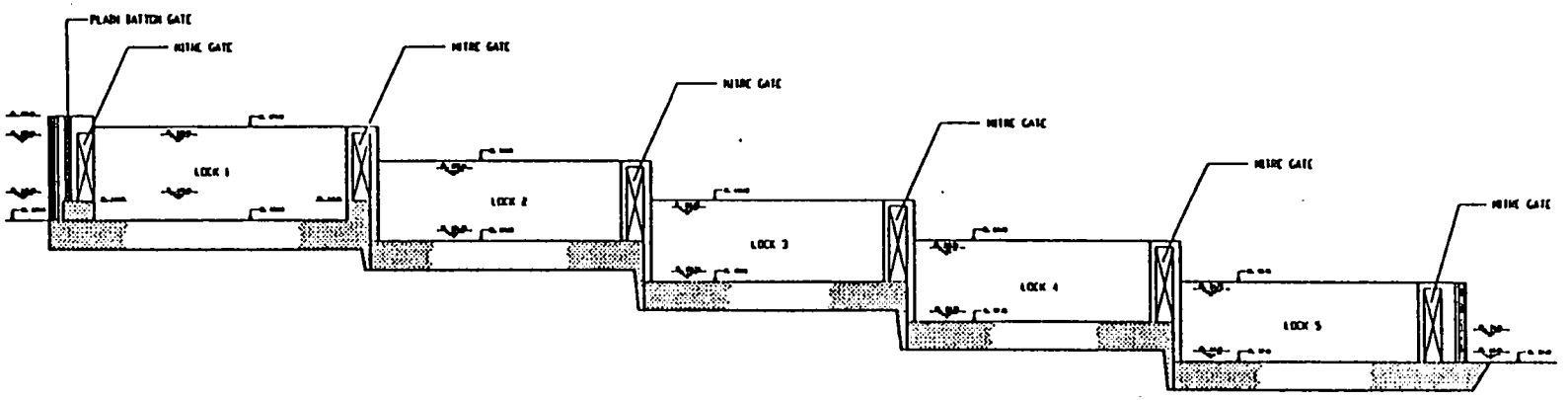


Figure 4. A vertical sectional view of the permanent shiplocks [after Ha(1993)].

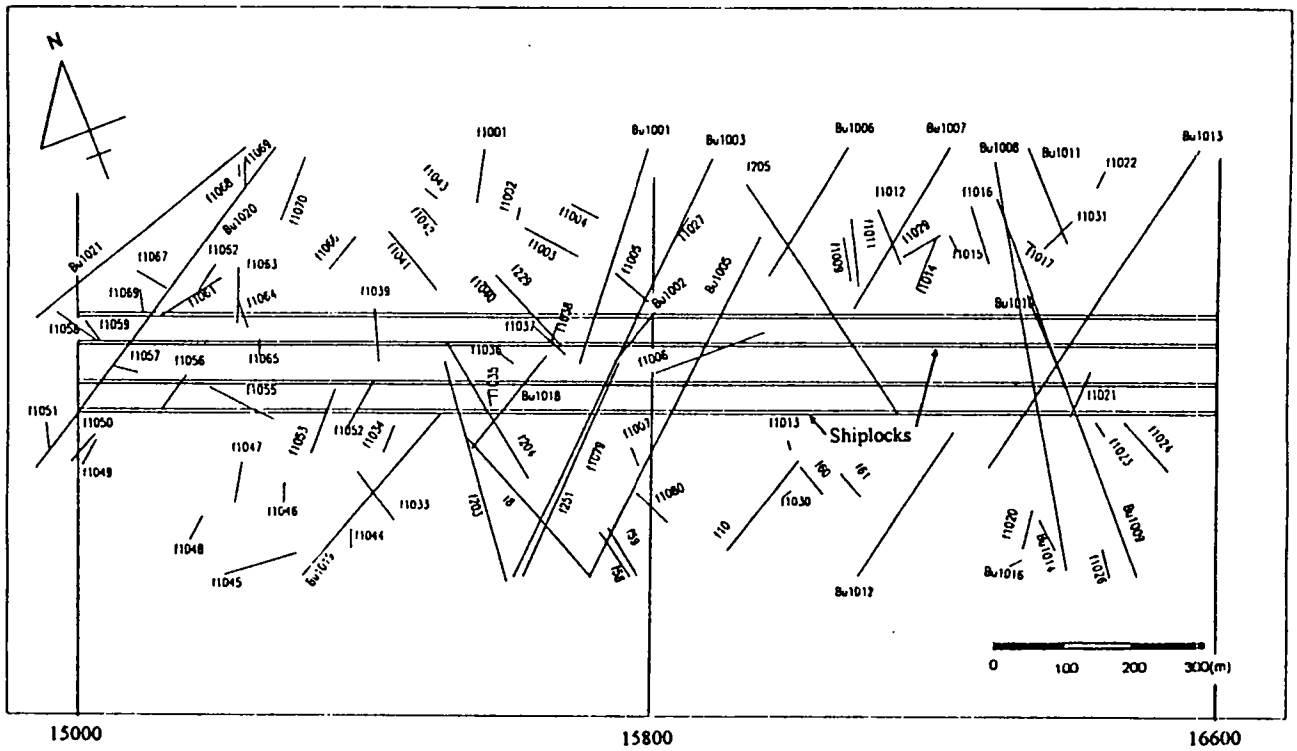


Figure 5. Major discontinuities in the proposed permanent shiplock area.

conditions of the shiplocks demand steep rock slopes on either side of the shiplocks which are about 1600 m long (Fig. 5). The granitic rock mass which exists in the shiplock region contains a number of major discontinuities, and a number of sets of minor discontinuities. Therefore, The rock engineering problems of high, steep rock slopes in the shiplock region are very complicated. Rock mass failures in the shiplock region must be fully prevented to assure safe travel of people in the ships and flood-free conditions downstream. Therefore, it is important to devise a design scheme to provide stable, steep, high rock slopes in the shiplock region. To achieve this ultimate goal, the first step was to characterize the discontinuity network in the rock mass close to the proposed permanent shiplock region (Kulatilake et al. 1996). This was done using about 2050 discontinuity trace data as mapped on the walls and the roof of a 400 m tunnel which is located close to the proposed shiplock region. This paper describes the second step performed towards reaching the ultimate goal. Note that further research is necessary to reach the ultimate goal.

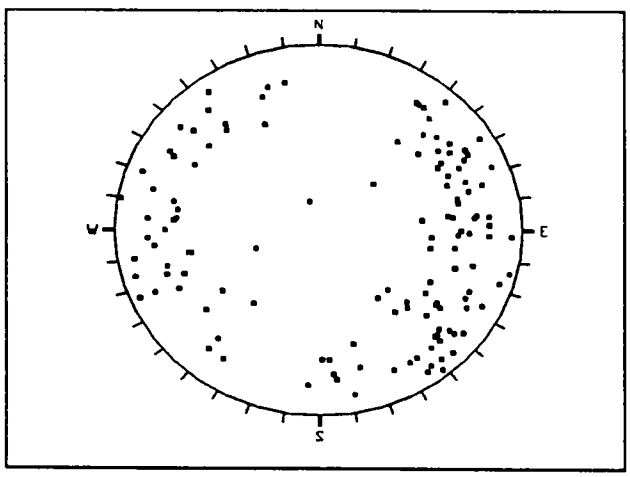


Figure 6. Orientation distribution of major discontinuity poles in the shiplock area on a lower hemisphere equal-area polar diagram.

Geology

Comprehensive geological research for the Three Gorges Region has been in progress since 1950s and is still continuing. The dam site of the Three Gorges Project is situated in the south part of the Huanglin anticline. The Huanglin anticline is 72 km long along the north-south (NS) direction and 35 km wide. It consists of crystalline rock masses of a presinian system. The major rock type is Huanglin granite (hornblend-biotite-plagioclase granite) which

is about 90% of the total area of the dam site, with the other being diorite. There are many intrusive dikes in the rock mass. The fresh granite and diorite are hard, with saturated compressive strengths around 100 MPa and 140 MPa, respectively. The ground surface has been seriously weathered. The thicknesses of weathered layers are different in ridges and ravines with an average thickness of 20-40 m. According to the degree of weathering, the rock mass is classified into 4 types (Wang 1986): (a) completely weathered, (b) strongly weathered, (c) weakly weathered, and (d) slightly weathered and fresh rock. The major rock type in the shiplock area is granite with the other being diorite. Diabase, quartzose and pegmatite can be found in the dikes or reefs. Also, lenses of Horneblende Schist can be found.

Major Discontinuities (Faults and Dikes) in the Proposed Shiplock Region

One hundred and thirty three major discontinuities have been mapped around the permanent shiplock covering an area of 1740x600m. Figure 5 shows the locations as well as the strike directions for these major discontinuities. Figure 6 shows the pole distribution of these major discontinuities on a lower hemispherical equal-area polar diagram. High orientation variability of the major discontinuities is well depicted by the polar diagram. Predominant strike directions of the major discontinuities seem to be along NEE-SWW and NW-SE directions. Most of them have steep dip angles varying between 60-70°. The lengths of most of these major discontinuities are expected to be between 50 and 100m. The friction angle (ϕ_j) values of the major discontinuities range from 35° to 45°. To obtain conservative results, the minimum value of ϕ_j of 35° was used in the performed kinematic analyses.

Kinematic Analyses

"Kinematic" refers to the motion of bodies without reference to the forces that cause them to move (Goodman 1989). Kinematic analyses are very useful to investigate possible failure of rock masses which contain discontinuities. Failure involving movement of rock blocks on discontinuities combine one or more of the three basic modes-plane sliding, wedge sliding and toppling. For the proposed permanent shiplock region, kinematic analyses were performed to estimate maximum safe slope angles with respect to the aforementioned three basic failure modes. The basic concepts related to estimation of maximum safe slope angles for the three basic modes of failure are briefly explained below.

Plane Sliding

Consider the case of plane sliding under gravity alone as shown in Fig. 7(a). Any block tending to slide on a single plane surface will translate down the slope parallel to the dip of the discontinuity. If a cut slope is inclined at an angle α to the horizontal, the conditions for a plane slide are that the dip vector of the discontinuity, D, be pointed into the free space of the excavation and plunge at an angle less than α. If δ is greater than α, the block will be stable and no sliding will take place.

This concept can be represented on the stereographic projection (Fig. 7(b)). The cut slope can be constructed as a great circle in the lower hemisphere. D_1 is the dip vector of discontinuity plane 1, and D_2 is the dip vector of discontinuity plane 2. δ_1 and δ_2 are dip angles of planes 1 and 2, respectively. In this example, δ_1 is less than α, and therefore, plane 1 would allow a plane slide. On the other hand, δ_2 is greater than α and plane 2 would not allow a plane slide. In other words, if the dip vector of the discontinuity plane lies in the shaded region, then the plane would allow a slide. The limiting situation arises when the great circle of the cut slope passes through the dip vector of the discontinuity plane. When this occurs, the dip angle of the cut slope corresponds to the maximum safe slope angle with respect to plane sliding.

Figure 7(c) shows the reverse situation. If the dip vector of a discontinuity surface is known, it is possible to determine the maximum safe slope angle corresponding to a cut of assigned strike. The maximum safe slope angle α is the dip of the great circle passing through the given strike and the known dip vector D.

It is important to note that in the case of plane sliding under self weight alone, failure can occur only if the surface of sliding dips steeper than ϕ_j. Therefore, in a lower hemispherical stereographic projection, if a dip vector D of a discontinuity lies in the shaded area in Fig. 7(d), plane sliding will not occur under any cut slope angle. That means the corresponding maximum safe cut slope angle is 90°. Thus, it is necessary to use the concepts shown in both Figs. 7(c) and (d) in estimating the maximum safe cut slope angles under the plane sliding situation.

Wedge Sliding

Sliding along a line of intersection occurs when two discontinuity planes intersect to make a wedge. Figure 8 shows how to obtain graphically the

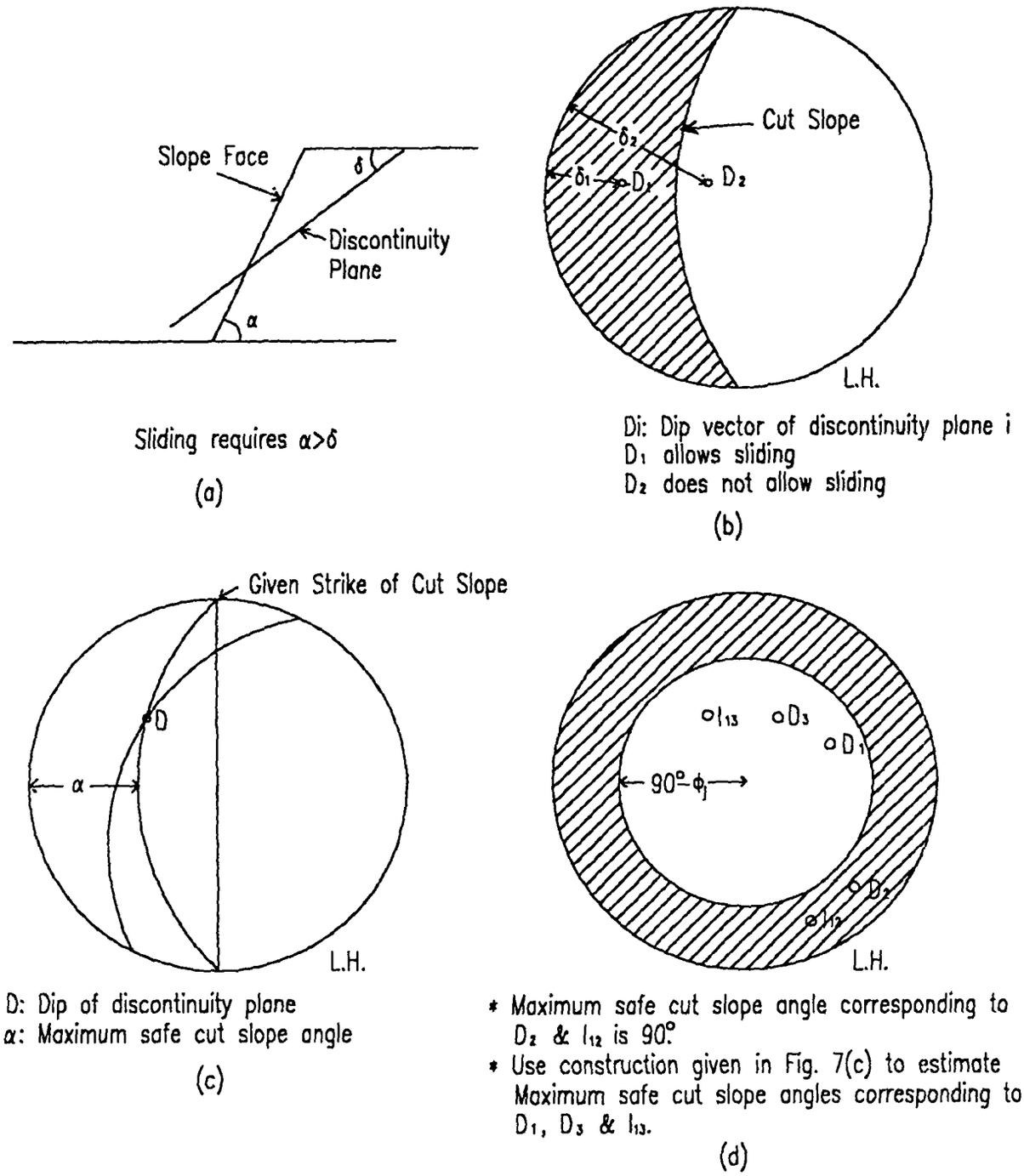
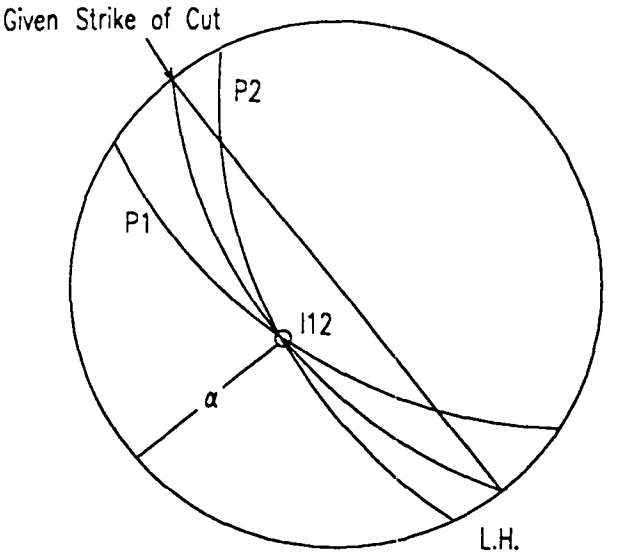


Figure 7. Concepts related to kinematic analysis for plane sliding [Goodman(1989)] (a) daylighting requirements on a pictorial diagram. (b) daylighting requirements on a stereographic plot. (c) great circle for cut slope providing the maximum safe slope angle. (d) influence of ϕ_j on maximum safe cut slope angle.

maximum safe cut slope angle for a possible sliding wedge. When discontinuity planes 1 (P1) and 2 (P2) make a wedge, the maximum safe slope angle for a cut slope of assigned strike can be found in a similar way to the plane sliding case. If the cut is made with the strike as shown in the figure, the maximum safe slope angle α is obtained by the dip of the great circle which passes through the intersection of planes 1 and 2 (I_{12}) and the points corresponding to the assigned strike.

Note that Fig. 7(d) is equally applicable for wedge sliding situation. If the line of intersection of two discontinuity planes, I_{ij}, lie in the shaded area in Fig. 7(d), then the maximum safe cut slope angle corresponding to I_{ij} is 90°. Thus, the concepts shown in Figs. 7(d) and 8 should be used in estimating maximum safe cut slope angles under wedge sliding situation.



α: Maximum safe cut slope angle

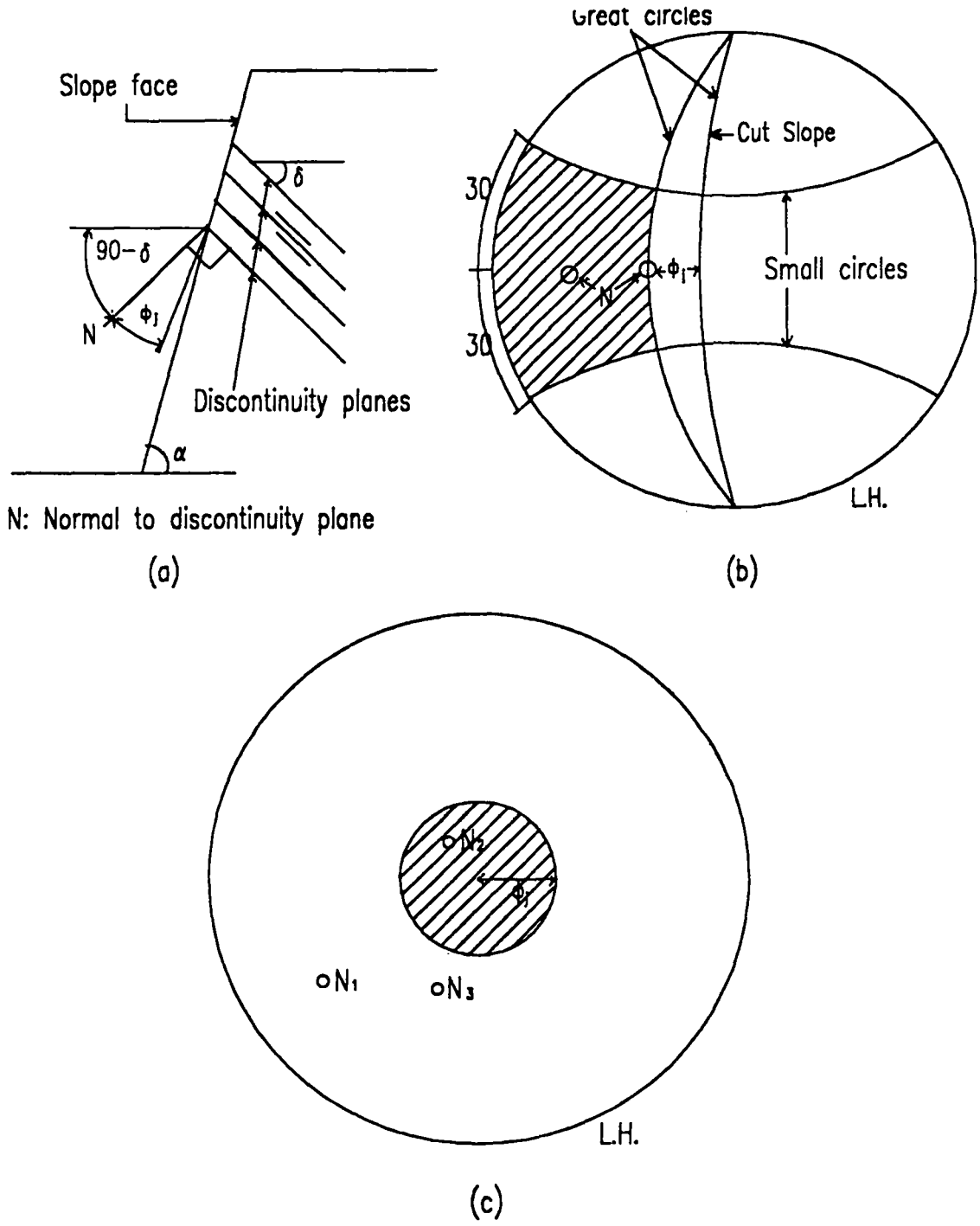
Figure 8. Stereographic construction to obtain the maximum safe slope angle for wedge sliding.

Toppling

Fig. 9 illustrates the kinematic analysis for toppling under gravity alone. Toppling can occur only if the discontinuities strike nearly parallel to the strike of the slope, say within 30°. In addition, discontinuity spacing should be low as shown in Fig. 9(a) to form thin layers of rock. For toppling failure, first it is necessary to initiate interlayer slip before large flexural deformations take place within the layers. If the layers have angle of friction ϕ_j, slip will occur only if the direction of applied compression (which is along the dip vector of slope) makes an angle greater than ϕ_j with the normal to the layers. If the cut slope is inclined α to the horizontal and the dip of discontinuity planes is δ, then the kinematic requirement for toppling is $(90-\delta) + \phi_j < \alpha$ as shown in Fig. 9(a). On a lower hemispherical stereographic projection, for toppling failure to occur, the normal vector N of the discontinuity should lie within the shaded area as shown in Fig. 9(b). The situation corresponding to the maximum safe cut slope angle occurs when N falls on the great circle which is ϕ_j degrees below the cut slope and striking parallel to it. Note that when N lies outside the two small circles shown in Fig. 9(b), the corresponding maximum safe cut slope angle is 90°. Also, it is important to note that toppling can occur only on discontinuities whose normals plunge at an angle less than $90-\phi_j$. That means with respect to the lower hemispherical stereographic plot shown in Fig. 9(c), if a normal vector of a discontinuity lies within the shaded area shown in Fig. 9(c), then the corresponding maximum safe cut slope angle is 90°. Thus, the concepts shown in both Figs. 9(b) and 9(c) should be used in estimating the maximum safe cut slope angles for toppling mode. Also, it is important to check whether the discontinuity geometry produces thin rock layers as shown in Fig. 9(a).

Results

Kinematic analyses were conducted using the mapped major discontinuities in the shiplock area to estimate maximum safe slope angles with respect to possible plane sliding, wedge sliding and toppling failure. Each major discontinuity was treated as a single feature. A computer code KINEM was developed to calculate maximum safe slope angles. Eventhough the mapped major discontinuities are shown in Fig. 5, some uncertainty exist concerning the actual extent of these discontinuities. Therefore, the kinematic analyses were conducted under two different cases as given below. For case 1, orientation of all the major discontinuities which appear on Fig. 5 were considered in the kinematic analyses. Under this case, indirectly, it was assumed that any major discontinuity appearing on Fig 5 has a possibility to cross the shiplock. Thus, results from these analyses can be considered to be on the conservative side. The major discontinuities which only crosses the shiplocks according to Fig. 5 were used in the kinematic analyses in the second case. If the major discontinuity map given in Fig. 5 is reliable, then these analyses should provide results closer to the reality.



* Maximum safe cut slope angle corresponding to N_2 is 90°
* Use construction given in Fig. 9(b) to estimate maximum safe cut slope angles corresponding to N_1 & N_3

Figure 9. Concepts related to kinematic analysis for toppling [Goodman(1989)] (a) a pictorial diagram. (b) on a stereographic plot. (c) influence of ϕ_j on maximum safe cut slope angle.

For Case 1 :

Plane Sliding. The maximum safe slope angle obtained for each major discontinuity is shown in Figs. 10(a) and (b), respectively for cut strike directions 111° (South-West shiplock face) and 291° (North-East shiplock face). The location of each bar shown in Fig. 10 corresponds to the actual location of the major discontinuity in the shiplock area. The starting and ending points of each bar represent starting and ending X coordinates of each major discontinuity. Note that X axis coincides with the strike of shiplock faces. Therefore, the location of each major discontinuity and the maximum safe slope angle corresponding to it can be identified from this diagram. Most of the bars in Figs. 10(a) and (b) indicate maximum safe slope angles between 65° and 90°. The results obtained for the whole shiplock region (disregarding the actual locations of major discontinuities) are shown through histograms in Fig. 11(a). Most of the maximum safe slope angles were found to be greater than 65°.

Wedge Sliding. For the wedge sliding mode, only the discontinuities which are located within a 50m distance were considered in calculating the maximum safe slope angles. Results for individual wedges are shown in Figs. 10(c) and (d). Similar to Figs. 10(a) and (b), each bar represents starting and ending X coordinates of each wedge. Figure 11(b) shows the histograms obtained for the maximum safe slope angle for the whole shiplock region disregarding the actual locations of the wedges in the shiplock area. For most of the wedges, maximum safe cut slope angles greater than 45° were obtained.

Toppling. It is not really necessary to consider toppling mode for the shiplock region since it is covered by a granitic rock mass. Toppling failures usually occur in layered rock such as slate, schist, and sedimentary rocks, and chances to form thin layers in a granitic rock mass are extremely low. Anyway, in this study, kinematic analysis was conducted for toppling. Due to the aforementioned reasons, these results should be considered as highly conservative estimates. Results for individual discontinuities are shown in Figs. 10(e) and (f). Figure 11(c) shows the histograms obtained for the maximum safe slope angle for the whole shiplock region disregarding the actual locations of the major discontinuities in the shiplock area. Most of the maximum safe slope angles for the toppling case were found to be greater than 85°.

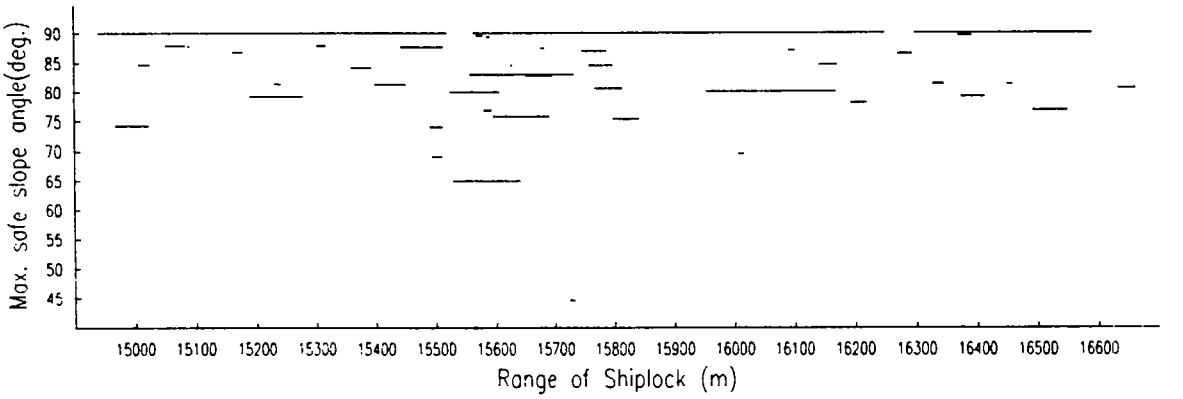
Table 1 Major discontinuity orientation data for different segments of the shiplock for case 2 kinematic analyses.

15150-15200(m)	Strike of SW Shiplock Face: 111°	
Discontinuity	Dip	Dip Direction
1. βu1020	82	316
2. F1062	60	326
3. F1061	75	351
4. F1067	80	46
15550-15600(m)	Strike of NE Shiplock Face: 291°	
Discontinuity	Dip	Dip Direction
1. F203	70	262
2. F204	82	45
3. βu1018	80	331
4. F1035	84	285
5. F8	80	247
15750-15800(m)	Strike of SW Shiplock Face: 111°	
Discontinuity	Dip	Dip Direction
1. βu1001	78	300
2. F1005	80	221
3. βu1002	80	325
4. βu1003	50	320
16080-16130(m)	Strike of SW Shiplock Face: 111°	
Discontinuity	Dip	Dip Direction
1. F205	65	269
2. F1009	80	274
3. F1011	70	101
4. βu1007	85	329
16420-16470(m)	Strike of NE Shiplock Face: 291°	
Discontinuity	Dip	Dip Direction
1. F1021	80	306
2. F1023	76	78
3. F1024	70	251
4. βu1009	78	90

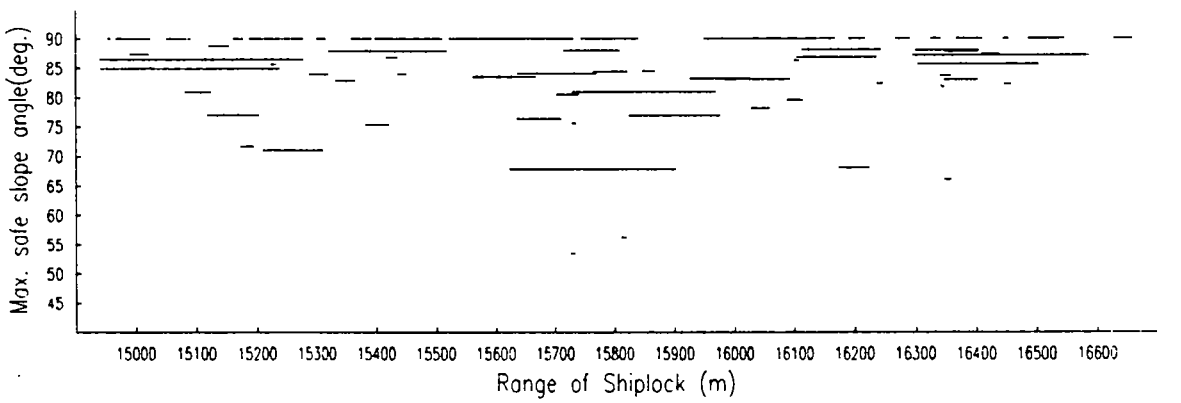
F: fault, βu: dyke
Orientations of Cut Slopes:
Strike of Cut (SW shiplock face) = 111°
Strike of Cut (NE shiplock face) = 291°
Dip Direction of SW shiplock face = 201°
Dip Direction of NE shiplock face = 21°

For Case 2 :

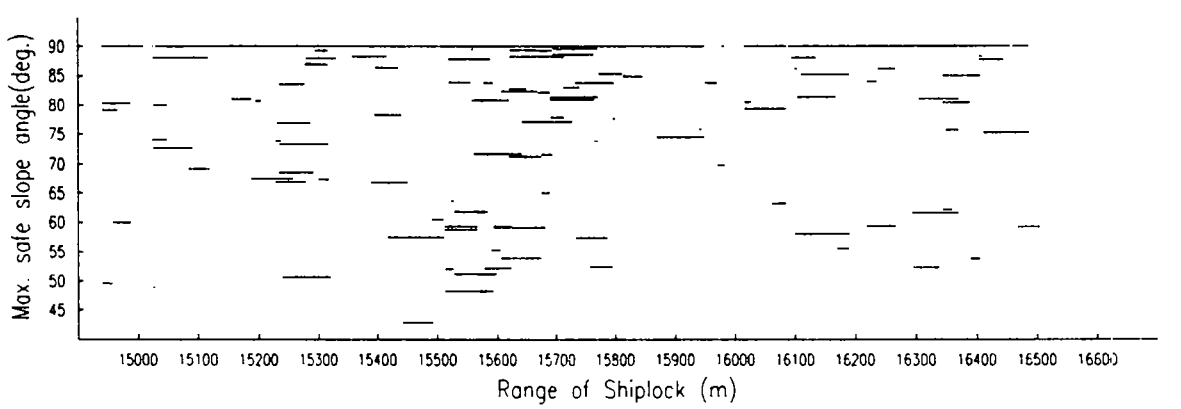
The total length of the shiplock (1750m) was divided into 50m segments. Those segments which do



(a) Plane Sliding - Cut Dir.=111°

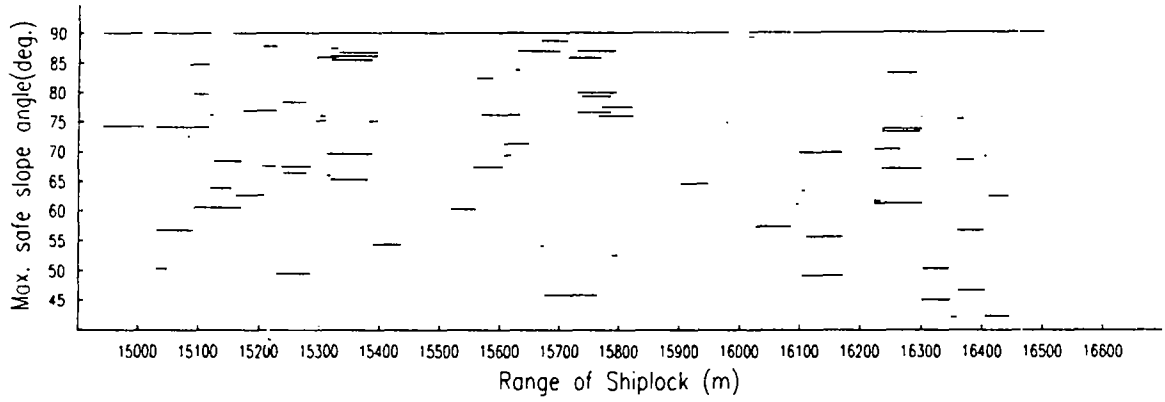


(b) Plane Sliding - Cut Dir.=291°

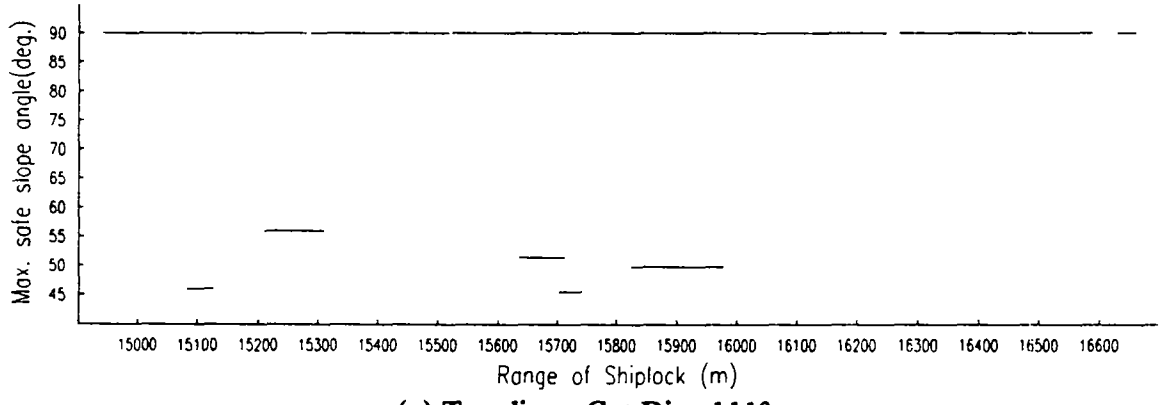


(c) Wedge Sliding - Cut Dir.=111°

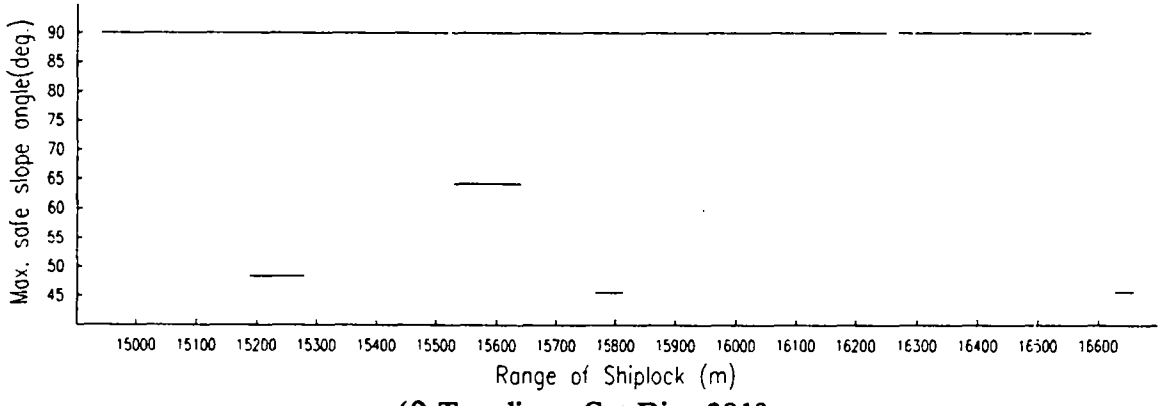
Figure 10. Maximum safe slope angles along shiplocks resulting from kinematic analyses conducted for case 1.



(d) Wedge Sliding - Cut Dir.=291°

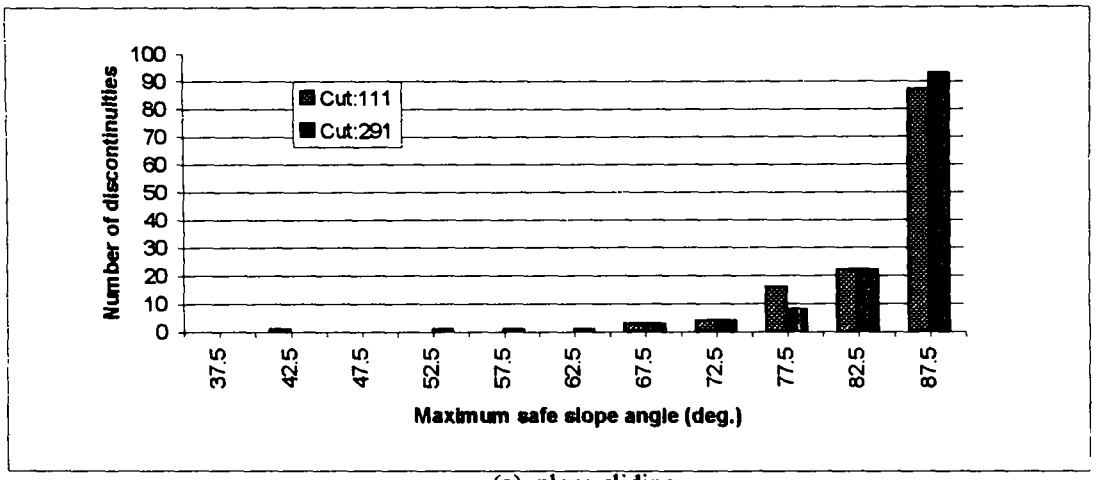


(e) Toppling - Cut Dir.=111°

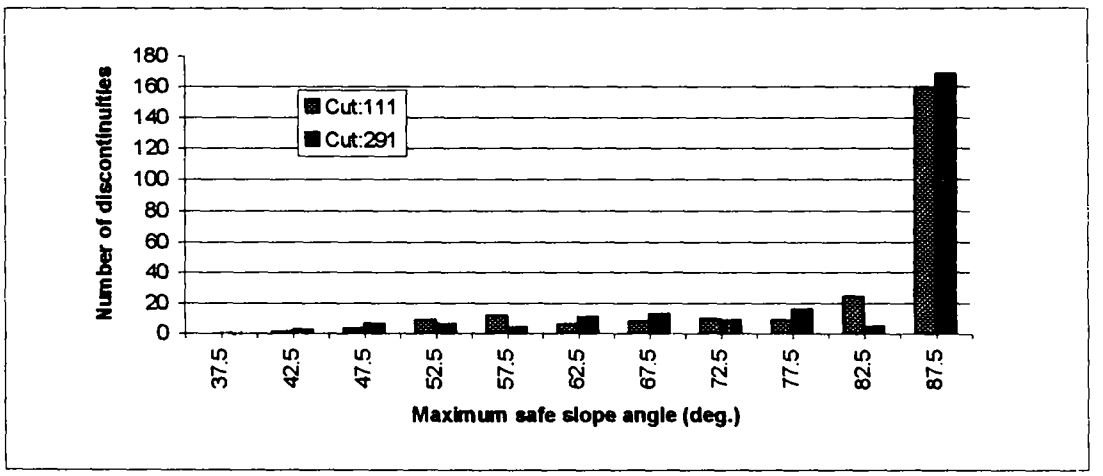


(f) Toppling - Cut Dir.=291°

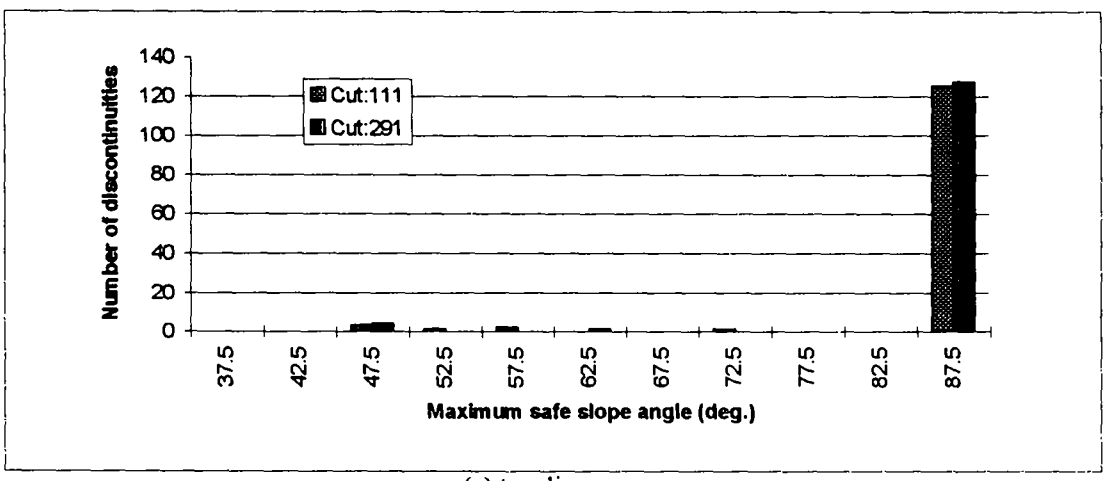
Figure 10. (Continued)



(a) plane sliding



(b) wedge sliding



(c) toppling

Figure 11. Histograms of maximum safe slope angle for different failure modes, based on kinematic analyses conducted for case 1.

not contain a potentially unstable block were not subjected to kinematic analyses. The kinematic analyses, therefore, were performed only for 5 segments where potentially unstable blocks were found. Table 1 gives the major discontinuities and their orientations for the segments which were subjected to kinematic analyses. Note that analyses were performed separately for the SW shiplock face (strike of cut 111°) and for the NE shiplock face (strike of cut 291°). The maximum safe cut slope angles obtained for plane sliding, wedge sliding and toppling modes for these different segments are given in Table 2.

Conclusions

Kinematic analysis for plane sliding has resulted in maximum safe cut slope angles greater than 65° for most of the discontinuities. Kinematic analysis has produced maximum safe cut slope angles greater than 45° for most of the wedges. For the wedge sliding mode, only the discontinuities which are located within a 50m distance were considered in obtaining the maximum safe slope angles. Maximum safe cut slope angles greater than 85° were obtained for most of the discontinuities in the toppling case.

According to the results obtained for both cases 1 and 2, it seems that the shiplock faces in the proposed permanent shiplock region in fresh rock are more or less stable up to a cut slope of about 45°. It is important to keep in mind that this conclusion is based on the kinematic analyses performed using only the major discontinuities. In the future, kinematic analyses should be performed incorporating both major and minor discontinuities. Also, it may be worthwhile to perform kinematic analysis based on the block theory and compare the results obtained through block theory against the results obtained through the traditional kinematic analyses conducted in this study.

All the analyses conducted in this study limited the loading on the rock mass to gravitational forces only. Therefore, some kinetic analyses should be performed incorporating possible water forces, earthquake forces, etc.

Table 2 Maximum safe slope angles for different failure modes in different segments of the shiplock region resulted from case 2 analyses.

Location (m)	D_1	D_2	D_3	D_4	D_5	T_1	T_2	T_3	T_4	T_5	I_{12}	I_{13}	I_{14}	I_{15}	I_{23}	I_{24}	I_{25}	I_{34}	I_{35}	I_{45}
15150-15200	90	90	90	90	-	90	90	90	46	-	90	90	90	-	80.7	90	-	90	-	-
15550-15600	90	82.7	83.5	90	90	90	90	90	90	90	65.8	90	90	67.4	79.4	78	65.2	82.5	90	90
15750-15800	90	80.6	90	90	-	90	90	90	90	-	84.1	89.7	90	-	84.9	90	-	90	-	-
16080-16130	80.1	87	90	90	-	90	90	90	90	-	90	90	71.7	-	90	86.1	-	90	-	-
16420-16470	87.4	82.3	90	85.6	-	90	90	90	90	-	62.4	90	57.8	-	90	75.3	-	90	-	-

D_1: For Plane Sliding along Fault 1
D_2: For Plane Sliding along Fault 2
D_3: For Plane Sliding along Fault 3
D_4: For Plane Sliding along Fault 4
D_5: For Plane Sliding along Fault 5
T_1: For Toppling on Fault 1 (ϕ_j=35°)
T_2: For Toppling on Fault 2 (ϕ_j=35°)
T_3: For Toppling on Fault 3 (ϕ_j=35°)
T_4: For Toppling on Fault 4 (ϕ_j=35°)
T_5: For Toppling on Fault 5 (ϕ_j=35°)

I_{12}: For Sliding on Planes 1 and 2 Along I_{12}
I_{13}: For Sliding on Planes 1 and 3 Along I_{13}
I_{14}: For Sliding on Planes 1 and 4 Along I_{14}
I_{15}: For Sliding on Planes 1 and 5 Along I_{15}
I_{23}: For Sliding on Planes 2 and 3 Along I_{23}
I_{24}: For Sliding on Planes 2 and 4 Along I_{24}
I_{25}: For Sliding on Planes 2 and 5 Along I_{25}
I_{34}: For Sliding on Planes 3 and 4 Along I_{34}
I_{35}: For Sliding on Planes 3 and 5 Along I_{35}
I_{45}: For Sliding on Planes 4 and 5 Along I_{45}

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APPLICATION OF SOIL NAILS TO THE STABILITY OF MINE WASTE SLOPES

by

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2. Eric C. Drumm and
3. Matthew Mauldon
4. Richard M. Berry

Abstract. The traditional soil nailed structure incorporates grouted or driven nails, and a wire mesh reinforced shotcrete facing to increase the stability of a slope or wall. This paper describes the construction and monitoring of a full-scale demonstration of nailing to stabilize coal mine spoil. The purpose of the investigation is to evaluate the performance of nailed slopes in mine spoil using methods proven for the stabilization of soil walls and slopes. The site in eastern Tennessee is a 12 meter high slope of dumped fill, composed of weathered shale chips, sandstone, and coal. The slope was formed by "pre-regulatory" contour surface mining operations and served as a work bench during mining. The material varies in size from silt to boulders, and has a small amount of cohesion. Portions of the mine spoil slope have experienced slope instability and erosion which have hampered subsequent reclamation activities.

Three different nail spacings and three different nail lengths were used in the design. The 12 meter high structure is instrumented to permit measurement of nail strain, and vertical inclinometer readings and survey measurements will be used for the detection of ground movement. The results of this study will aid in the development of design recommendations and construction guidelines for the application of soil nailing to stabilize mine spoil.

Key Words: Soil nailing, slope stability, contour surface mining, spoil piles

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Introduction

Throughout the Eastern Coal Province, contour surface (strip) mining has been utilized for the extraction of coal. It is not uncommon for the mine spoil and waste rock removed during mining to form unstable or marginally stable slopes. The materials forming these slopes are generally uncompacted, loose fill comprised of crushed shale and sandstone. This waste rock material forms slopes below contour surface mine high walls, or if constructed after the 1977 Surface Mining Act, may be used to reclaim high wall benches to approximate the original contour. Many of these slopes are active, with ground movement carrying trees and vegetation downslope. Throughout the Appalachian region, buildings and roadways have been constructed on or below the waste slopes. Expensive concrete retaining walls and mechanically reinforced earth structures have been constructed to stabilize the slopes and protect roads and property. In many cases the cost of remediating the slopes is not easily justified. Therefore,

practical, cost-effective, alternative construction techniques must be developed for the remediation of these slopes.

The application of earth inclusions, or soil nailing, is a proven and effective method for the stabilization of soil walls and slopes (Gässler and Gudehus, 1981; Munfakh et al., 1987; Stocker et al., 1990; Thompson and Miller, 1990; Elias & Juran, 1991). Soil nails are bars or cables, generally steel, used to reinforce a vertical or sloping face, and are usually driven or grouted into place. The construction sequence is depicted in Figure 1 (Clouterre, 1991). The nails are not tensioned and although shotcrete is often applied to

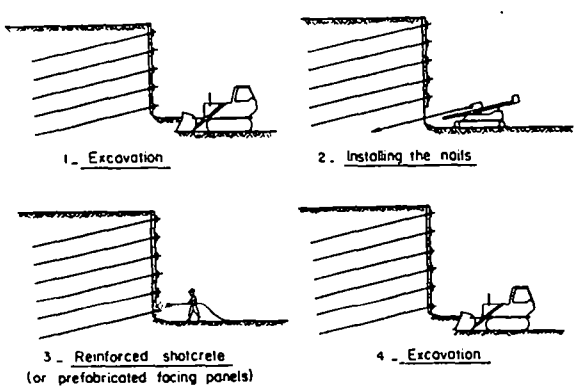

Figure 1 Schematic of soil nailed wall construction process (Clouterre, 1991)

the face, the face treatment plays a minor role in the overall stability. Construction usually occurs from the top down, with the nails gradually being loaded as the excavation progresses and the soil deforms. There are many analytical techniques used, and the construction technique is often governed by site conditions, making the technique somewhat site dependent. However, this is a major advantage of soil nailing, since the design (nail inclination, spacing, and length of nail) can easily be changed during construction based on the observed performance (Mitchell and Villet, 1987).

The purpose of this research program is to evaluate the performance and stability of nailed slopes in mine spoil using methods proven for the stabilization of soil walls and slopes. The application of these techniques to mine spoil materials has not been previously demonstrated or investigated.

Site Description

The site chosen for stabilization is in the Windrock area north of Oak Ridge, Tennessee, and is designated Buffalo Mountain on the Windrock Tennessee topographic map. The surrounding area is a combination of reclaimed sites, partially reclaimed surface mine bond forfeiture sites, and pre-1977 sites. The specific slope identified for the research is a bond forfeiture site that has experienced one or more slope failures since the mining operations were completed. The slope is comprised of loose uncompacted fill material which varies in size. It currently supports little vegetation, thus, construction and subsequent monitoring will be simplified.

The site on Buffalo Mountain is located just above 914 m (3000 ft), and is surrounded by a terrain covered with locust trees and various hardwoods. The appearance of the area changes dramatically from season to season. The site remains wet throughout the year, and from October to April, maintains a semi-frozen state. Figure 2 (Walker, 1995) shows the failure slope and remnants of the original slope, which has come to rest at the bottom of the scarp.

Figure 2 Aerial photograph of Buffalo Mountain site

Numerous contour surface mine operations were conducted in the

region. In an investigation of mine spoil stability in the eastern U.S. coal province, Swanson et al. (1983) included two nearby sites that are geologically similar to the Buffalo Mountain site. The sites (in Scott and Anderson counties) are in the Wartburg basin which is "bounded on the southeast by steeply dipping beds which form the boundary between the Cumberland Plateau and the Valley & Ridge Province"(Swanson et al., 1983). The study includes published material properties such as grain size distributions, shear strength, and compaction properties for the mine spoil. A cross section of the mined coal seams and resulting high walls in the Buffalo Mountain area is shown in Figure 3.

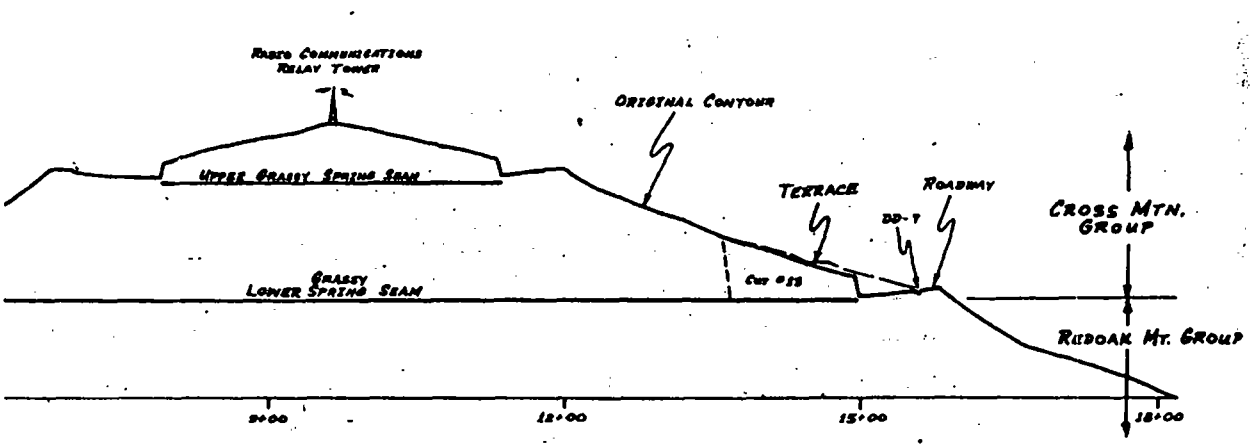

Figure 3 Geologic profile of Buffalo Mountain with strip mine highwalls

Topographic maps from pre-mining and post mining operations

An investigation of the original site terrain was conducted to determine the effects of the mining on the site topography. To investigate these effects, topographic maps from 1952 (before mining) and 1975 (after mining) were obtained. Using the coordinates supplied by the mine operators, the location of the road passing over the slope was found on the 1975 topographic map. Then the same point was located on the 1952 topographic map. Portions of the two topographic maps are shown in Figure 4 and Figure 5. A line (designated A-A) was drawn up and down slope of the road on both topographic maps. From the elevations and distances between contours along this line, a profile of the pre-mining and post mining slope was created. A comparison of the two profiles revealed that a spoil pile about 12 m (40 ft) deep was placed in the project area as shown in Figure 6.

Material Properties of Spoil

Two vertical auger borings were drilled from the road crossing the crest of the slope. From the borings, bulk samples of the spoil materials were obtained, and soil nail pullout tests were conducted. Both borings were terminated at a predetermined depth of 9 m (30 ft), which was within the mine spoil. The borings did not encounter residual soil or the mountain face. This was consistent with the profile obtained from the topographic map survey which suggested there was about 12 m (40 ft) of overburden in the slope. No ground water was encountered in either boring.

Two nail pullout tests were conducted to measure the maximum shear stress between the grout and the mine spoil. Test nails were installed in the bore holes, and after the grout cured were loaded to failure. From the pullout tests, an ultimate friction value was obtained which is an important input parameter in the design of the nailing system. The grouted portion of the nails were 200 mm (8 in) diameter and 2.4 m(8 ft) in length. Both tests yielded identical results. Table 1 lists the results of the pullout test.

Cuttings were collected from the

Figure 4 Topographic Map 1952

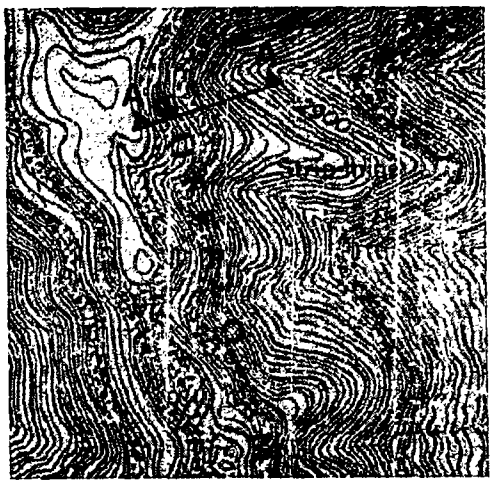

Figure 5 Topographic Map 1975

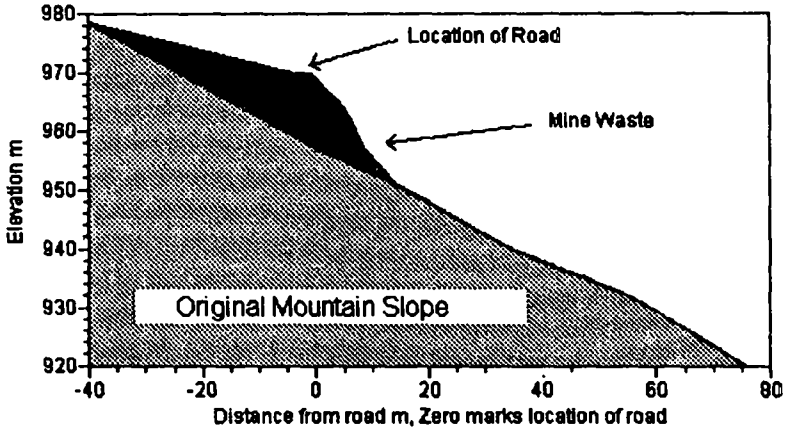

Figure 6 Profile of Buffalo Mountain in 1975 as estimated from topographic maps.

Table 1 Nail Pullout Test Results

Results of Pullout Tests Conducted 3/13/95			
Applied Load at Failure	Surface Area of Soil Nail	Displacement at Failure	Ultimate Stress in Nail
52.6 kN (11.8 kips)	1.56 m^2 (16.8 ft^2)	0.152 mm (0.006 in)	33.7 kN/m^2 (704 lb/ft^2)

boreholes and washed through a #200 sieve. A sieve analysis indicated at least 95% finer than #200. During the #200 wash, it was noticed that some of the spoil material was breaking down which suggests the material is subject to slaking. Atterberg limits were determined on a sample of mine waste from the southwest portion of the slope. The results are shown in Table 2. These results suggest that the mine spoil has a relatively narrow plasticity index, and is therefore unlikely to undergo significant volume change due to water content variation. The material would be classified as a low plasticity silt (ML) by the USCS

Table 2 Atterberg Limits of Windrock Spoil

Sample Origin	LL Liquid Limit	PL Plastic Limit	PI Plasticity Index
Windrock Spoil	31	26	5

classification system (Howard, 1984). The low PI also suggests that the nailed slope will not be subject to significant creep deformation and is well below the maximum value of 20 recommended for permanent soil nailed walls (Elias and Juran, 1991).

In-Place Testing

Due to the granular nature of the material and wide range of particle sizes, undisturbed tube samples for strength testing could not be obtained. Therefore, two in-place borehole shear tests (Lutenegger, 1987) were conducted. The tests were conducted with various consolidation times to determine effective strength parameters. The two tests yielded nearly identical results as shown in Figure 7. A best

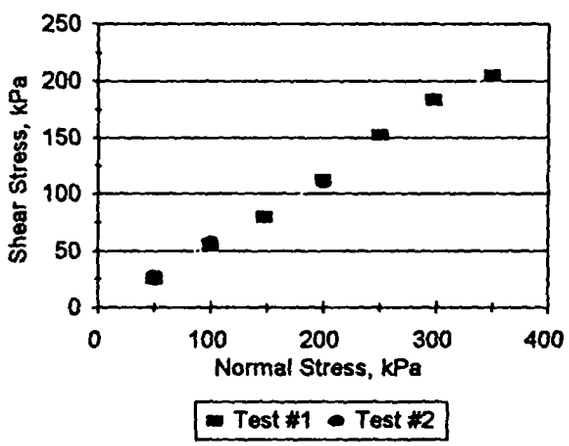

Figure 7 Results of borehole shear tets on Buffalo Mountain mine spoil

fit line through the data yields a failure envelope with a friction angle (ϕ) of about 30°. This was consistent with data reported by Swanson et al., (1983) on remolded samples of mine spoil from nearby mining operations.

Slope Stability Analysis of 1975 Slope Profile

Inspection of the slope suggests that the material possesses a small amount of cohesion that would not be measured in disturbed samples from lab and field tests. To estimate the cohesion, a profile of the slope prior to the failure (Figure 8) was estimated from the 1975 topographic map. This profile was evaluated using the STABL5 (USDOT, 1985; Carpenter, 1986) slope stability analysis program. Using the material properties in Table 3, the STABL5 program was used to back-calculate a value for the cohesion corresponding to a factor of safety, FS=1. The cohesion value was determined to be 8.6 kPa (180 psf).

Stability Analysis and Soil Nail Slope Design

Slope Stability of 1995 Slope Profile

With an estimate of the cohesion it was possible to analyze the existing slope on Buffalo Mountain. A recent slope profile obtained from an optical survey (June 1995) was used in the analysis. The water table was assumed to flow along the original topography of the mountain beneath the spoil and through the toe of the slope. This assumption was based on the absence of water at a depth of 9.0 m (30 ft) in the auger holes in the road and the observation of water at 0.3 m (1 ft) below the slope surface in a hand augered borehole at the toe of the slope. From the STABL5 analysis the factor of safety was found to be about 1.18. A factor of safety greater than one was not surprising since the slope had failed previously, as indicated by the soil mass and trees in Figure 2. Figure 9 depicts the existing slope.

Design and Analysis of Cut Slope

Since the 1995 slope profile was judged to be stable, it was decided to modify the slope to reduce the stability. By creating a steep

Table 3 Properties used in STABL5 analysis of 1975 slope profile.

Dry Unit Weight γ_d	Saturated Unit Weight γ_{sat}	Friction Angle ϕ
16.6 kN/m³ (106 lb/ft³)	18.2 kN/m³ (116 lb/ft³)	30°

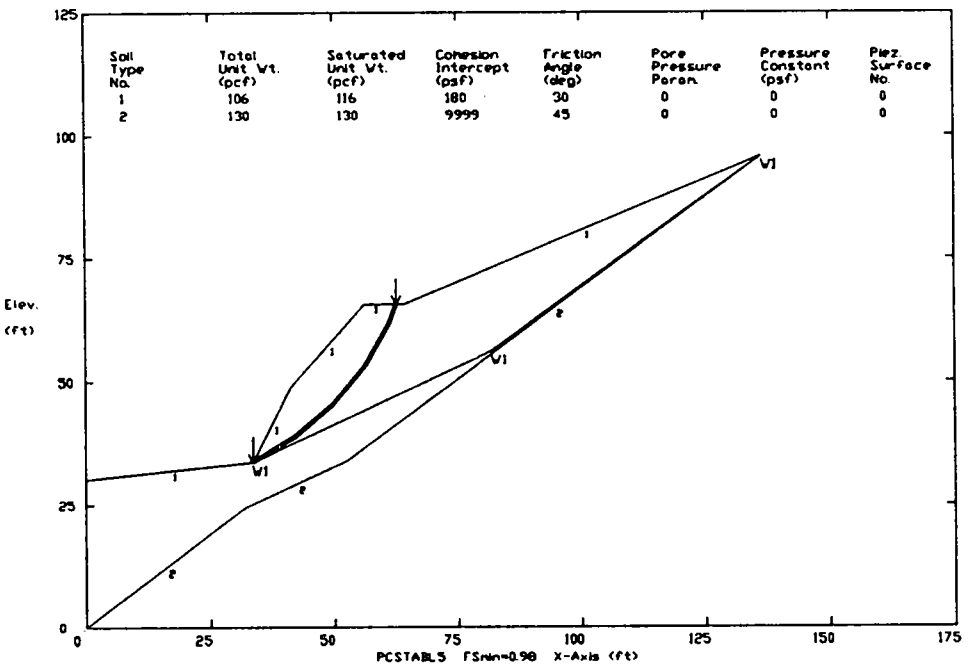

Figure 8 Stability analysis of 1975 slope profile

cut in the toe of the slope the stability was reduced. The highest unreinforced cut in the toe that can be made without instability is around 3.7 m (12 ft). Thus, it was determined that the lower 6.1 m (20 ft) of the slope would be cut as shown in Figure 10. The new slope profile consists of an upper portion 6.1 m (20 ft) high at the natural slope, and a lower portion about 6.1 m (20 ft) high cut to 15° from vertical.

To evaluate different nailing schemes, the new slope was divided into four sections. Three sections have different nailing schemes, and one control section is without nails. The upper natural slope is reinforced with four rows of nails 3.0 m (10 ft) long, spaced 1.5 m (5 ft) vertically, with horizontal spacing consistent with that of the nails below. The various nailing schemes for the lower slope are shown below in Table 4, and an elevation view of the slope is provided in Figure 11. As indicated in Table 4, the nail length as well as the horizontal spacing, S_h and vertical spacing S_v is different in all three sections.

Each section of slope was analyzed using five different methods SNAIL (Caltrans, 1992), EDINA, (Law Environmental Inc., 1991), NAIL (Geoconsult Inc., 1991), the Kinematic method (Elias and Juran, 1991) and STABL5 (USDOT, 1985; Carpenter, 1986). The material properties and resulting factors of safety for each method are listed in Table 5 and Table 6, respectively.

Different factors of safety resulted from different soil nailing analysis methods due to variations in the idealized slope. SNAIL, EDINA, FHWA, and STABL5 were all able to accommodate the natural slope above the cut slope portion in its analysis. For the NAIL program, only

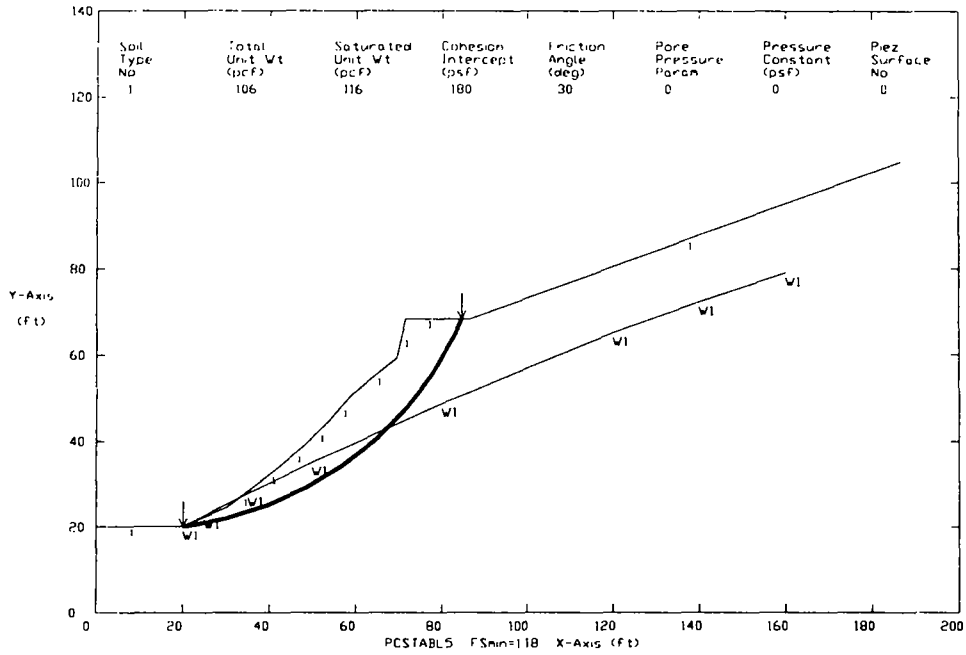

Figure 9 Stability analysis of 1995 slope

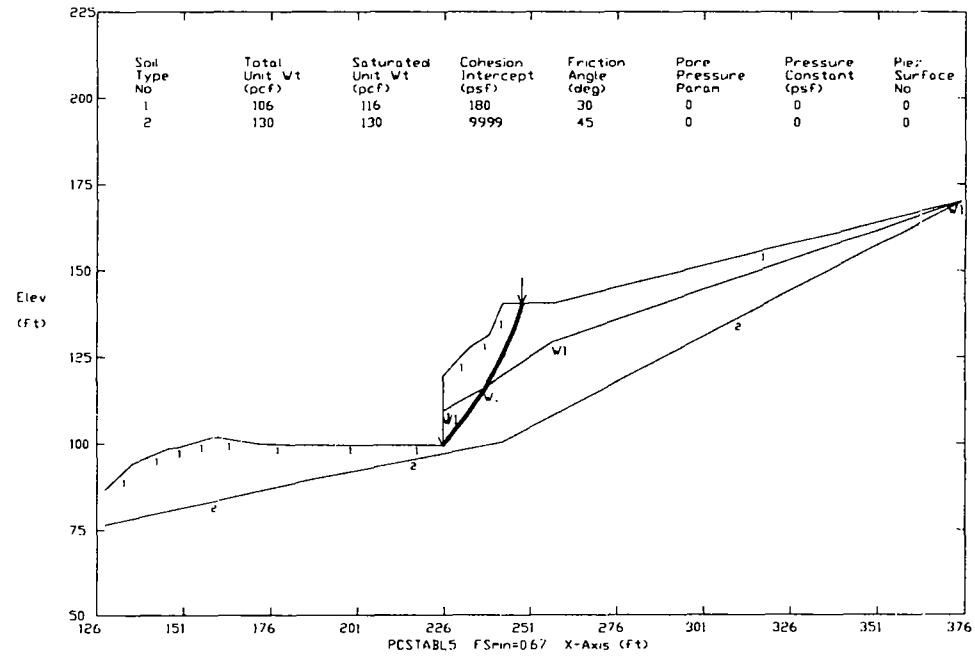

Figure 10 Profile of cut slope used in soil nailing design

Table 4 Summary of Nailing Schemes for Cut Slope

Section Row	Section 1		Section 2		Section 3	
	Sh= 1.5 m (5 ft)	Sv= 1.5 m (5 ft)	Sh= 1.8 m (6 ft)	Sv= 1.5 m (5 ft)	Sh= 1.8 m (6 ft)	Sv= 1.8 m (6 ft)
Row 1 Nail Length	6.1 m (20 ft)		4.6 m (15 ft)		4.6 m (15 ft)	
Row 2 Nail Length	6.1 m (20 ft)		4.6 m (15 ft)		3.0 m (10 ft)	
Row 3 Nail Length	6.1 m (20 ft)		4.6 m (15 ft)		3.0 m (10 ft)	
Row 4 Nail Length	6.1 m (20 ft)		3.0 m (10 ft)		3.0 m (10 ft)	

Table 5 Summary of Material Properties used for Analysis of Nailed Slope

Property	Dry Unit Weight	Saturated Unit	c	Φ
Value	16.6 kN/m³ (106 lb/ft3)	18.2 kN/m³ (116 lb/ft³)	8.6 kPa (180 lb/ft²)	30°

Table 6 Summary of Factors of Safety from Stability Analysis

Analysis Method	Program	Section 1	Section 2	Section 3	Section 4 Unreinforced
Stocker and Riedinger (1990)	SNAIL	1.31	1.12	1.06	0.9
Gässler and Gudehus (1981)	EDINA	2.39	1.03	0.83	N/A
Juran et al. (1990)	FHWA Method	1.8 6.1 m (20 ft) Nail	1.25 4.6 m (15 ft) Nail	0.5 3.0 m (10 ft) Nail	N/A
Shen et al. (1981)	NAIL	1.64	1.06	0.91	N/A
Bishop (1955)	STABL5	N/A	N/A	N/A	0.75

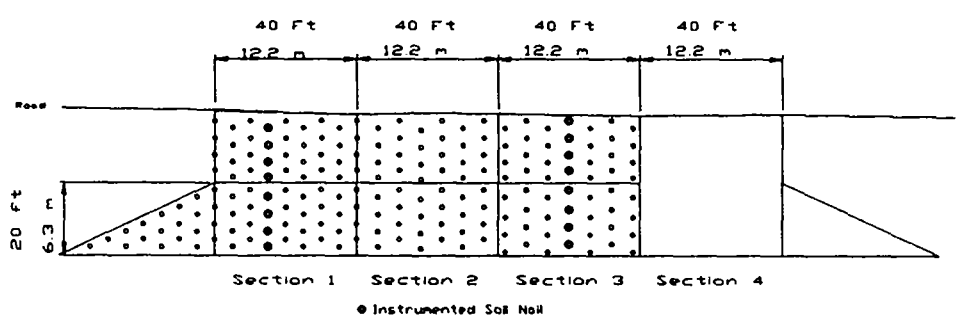

Figure 11 Elevation view of nailing and instrumentation scheme

the cut portion of the slope was analyzed, resulting in slightly elevated factors of safety. In the FHWA analysis only uniform nail length could be analyzed, therefore each section was analyzed using the uniform nail lengths included in Table 6. The results from Table 6 are compared graphically in Figure 12. The results reflect the decrease in the factor of

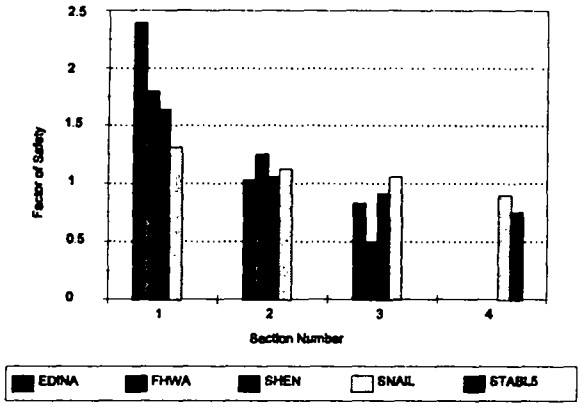

Figure 12 Comparison of factors of safety fo various analysis programs

safety from Section 1 to Section 4. The differences between the results of the analysis methods can be attributed to the difference in the idealized slopes and the manner in which the method is implemented in the computer code.

The purpose of this study is to investigate the effectiveness of soil nailing in mine spoil, therefore the four sections were designed with low factors of safety which would allow deformations of the slope and in the facing that would be observed over the life of the project.

The above analyses were conducted with an assumed water table exiting at the toe of the cut slope. This water table location assumes that the construction will draw the water table down to this elevation. To reflect the worst case water table condition, the analysis was repeated for an elevated, or short-term water table. These conditions reflect the location of the water table as it might be encountered during construction. The results of these analyses are shown in Table 7.

Like many abandoned mine land sites, Buffalo Mountain is very remote and not well suited to the application of shotcrete. Therefore, a reasonable facing which can easily be delivered and installed was selected. The facing of the slope is not the conventional shotcrete and wire mesh facing. Instead, a geosynthetic reinforced erosion blanket was installed, and held in place by steel plates attached to the nails. This should encourage the development and growth of vegetation on the slope to reduce the effects of erosion.

Instrumentation and Monitoring Program

Monitoring of the nailed slope is an important aspect of the research project. The slope deformations are monitored using vertical slope inclinometers, and surveying benchmarks. The slope inclinometers were placed in each of the four test sections and were installed to monitor the slope before, during, and after construction. To assure that the bottom of the inclinometer casing remained stationary, it was installed to a depth of 15 m (50 ft) which is 3 m (10 ft) deeper than the toe of the slope.

To record stresses in the nails, embedment strain gages, model EGP-5-350 manufactured by the Micro Measurements Group (Figure 13) were

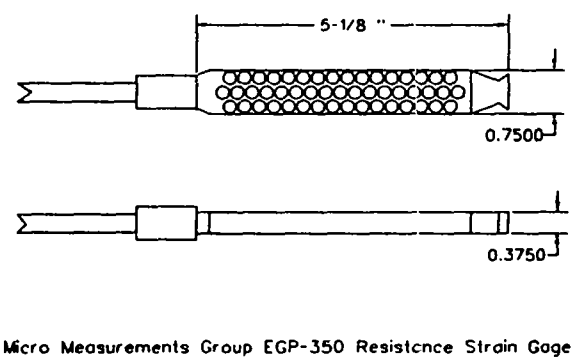

Figure 13 Embedment strain gage

utilized. In the upper natural slope, two instrumented columns of nails (Section 1 and Section 3) were instrumented as shown in Figure 11.

Table 7 Summary of Factors of Safety from Stability Analysis (Short Term Analysis)

Reference	Program	Section 1	Section 2	Section 3	Section 4 Unreinforced
Stocker and Riedinger (1990)	SNAIL	1.21	1.02	0.87	0.77
Bishop (1955)	STABL5	N/A	N/A	N/A	0.75

Both sections will each have four nails instrumented with gages, with three gages along the length of each nail. The gages are placed at 100 cm (3.2 ft), 180 cm (5.9 ft), and 230 cm (7.6 ft) along the nail relative to the facing end. For the lower portion of the slope, one column in each section will be instrumented as shown in Figure 11. The instrumentation scheme for various nail lengths is listed in Table 8. A minimum of four resistance gages will be used for a single nail.

A permanent benchmark was established to monitor deflections in the face of the slope. Reflectors will be attached along the facing, and a total station system used to monitor movements in the nails and the facing.

Construction And Monitoring

The installation of the nails was underway at the time this paper was prepared. The instruments will be monitored during construction and afterward to obtain measurements of slope deformation, nail deformation, and stress distribution in the nails. The results will be compared with those in the literature for nails in soil. Recommendations will then be made regarding the application of soil nailing to mine waste.

Acknowledgments

This project was supported by the by U.S. Bureau of Mines, Abandoned Mine Lands Program, Denver Research Center, through Contract No. 1432-J0240002. Erik Westman served as the technical contract officer. The input from Danny D. Rahnema, Douglas Siddell, and Craig Walker of the Office of Surface Mining is also appreciated.

Table 8 Summary of Strain Gage Placement for Various Nail Lengths

Nail	Gage 1*	Gage 2	Gage 3	Gage 4	Gage 5	Gage 6	Gage 7
6.1 m (20 ft)	45 cm (1.5 ft)	120 cm (4.0 ft)	195 cm (6.4 ft)	270 cm (8.9 ft)	345 cm (11.3 ft)	420 cm (13.8 ft)	495 cm (16.2 ft)
4.6 m (15 ft)	45 cm (1.5 ft)	120 cm (4.0 ft)	195 cm (6.4 ft)	270 cm (8.9 ft	345 cm (11.3 ft)	Not Used	Not Used
3.0 m (10 ft)	45 cm (1.5 ft)	120 cm (4.0 ft)	195 cm (6.4 ft)	270 cm (8.9 ft)	Not Used	Not Used	Not Used

* Location of all gages are relative to the facing end of the nail and is installed at the neutral axis to eliminate effects due to bending.

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HYDROLOGIC INVESTIGATION AND REMEDIATION OF A POST-REMINING ACIDIC SEEP[1]

by

William W. Aljoe[2]

Nile A. Linberg[3]

Abstract. Surface remining of coal pillars in abandoned underground workings in the Pittsburgh seam in southwestern Pennsylvania has often resulted in post-remining discharges whose water quality is the same or better than the pre-existing discharges. However, at one such operation in Washington County, PA, an increase in contaminant loading occurred at an outcrop seep after remining. This problem was believed to be at least partly related to a small unstripped area of the old deep mine workings immediately upgradient from the seep. A hydrologic investigation that included a chemical tracer test, slug tests in the remined spoil, and water quality monitoring indicated that the mine pool in the old workings discharged through the seep. However, the water in the mine pool and much of the remined spoil was consistently alkaline; this suggested that the acidic water may have originated in other areas of the spoil and old workings, and passed rapidly to the seep through a highly transmissive portion of the spoil. Acting on this assumption, the mine operator successfully implemented a remediation scheme in which the spoil was excavated to intercept the acidic spoil water. The excavation was then re-emplaced with an anoxic limestone drain at its base. The drain now serves both to add alkalinity to the water and to divert the seep to an area where metals can be removed easily via precipitation in wetlands.

Additional Key Words: mine hydrology, mine spoil

Introduction

A great deal of uncertainty and confusion often exists with respect to the origin of contaminated "off-site" seeps which emanate from unmined areas near reclaimed surface mine spoils. The hydrology of the spoil material and surrounding strata is often only poorly understood, and the potential sources of contamination within and/or around the spoil are usually not well-defined. The situation is complicated even further in remining situations or when other surface mines exist on adjacent properties. In these cases, the local hydrology may have already been altered by the previous mining. The consequences of this uncertainty become critical in the context of SMCRA enforcement, where a regulatory agency must make a determination as to whether a mining permit holder is responsible for the remediation of the contaminated off-site discharge.

The degree of uncertainty associated with the origin of an off-site discharge can sometimes be reduced by conducting a preliminary hydrologic investigation at the mine site. While the investigation may still not reveal

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the exact source of the contamination, the information may be sufficient to narrow the differences of opinion which had existed between the permit holder and the regulatory agency. This paper describes how a preliminary hydrologic investigation was used to help a mine operator identfy the potential source of a contaminated off-site seep, implement a cost-effective remediation scheme, and avoid a protracted litigation process.

Mining History

The study site is located in Washington County, Pennsylvania, about 15 miles southwest of Pittsburgh. As shown in Figure 1, the entire Pittsburgh coal seam in the general area had been mined by underground methods prior to 1940. Portions of the deep mine barrier pillar at the Pittsburgh coal outcrop adjacent to the eastern side of the mine permit area had also been strip mined previously during the early 1950's. However, enough coal still remained in the deep mine pillars to allow an economically viable surface mine operation within the permit area shown in Figure 1.

Figure 2 shows the topographic contours and a cross section of the mine permit area prior to the most recent period of surface mining. This mining took place between 1985 and 1990, beginning at the coal outcrop area shown in Figure 2 and progressing to the southwest, southeast, and northwest until the permit boundary or an uneconomic overburden depth was reached. The overburden consisted mostly of shale, with some sandstone. A persistent limestone bed of 5-10 ft thickness was located approximately 40 ft above the Pittsburgh coal. This limestone was believed to be responsible for the alkaline discharges that were produced from remining operations in the vicinity of the study site, despite the fact that the deep mines had discharged acidic water prior to remining. Because of this, the mine operator sought a permit under SMCRA regulations rather than under the Pennsylvania remining program. Therefore, while the subject mine was a remining site in a physical sense, it

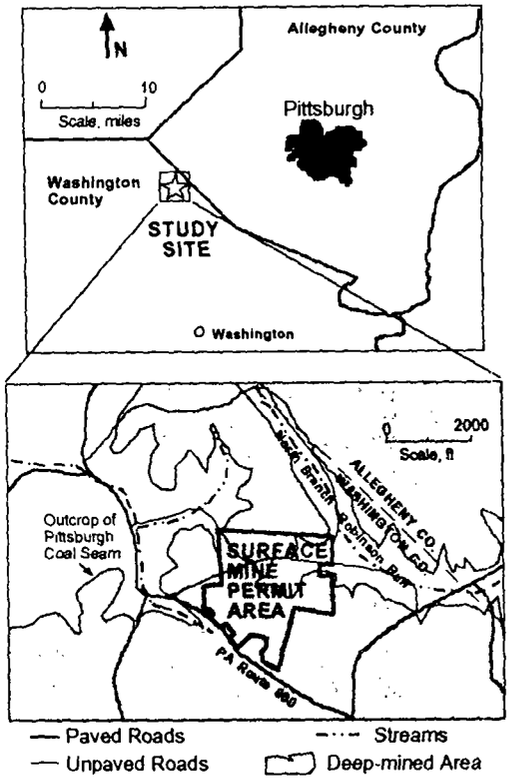

Figure 1. Location Map of Study Site

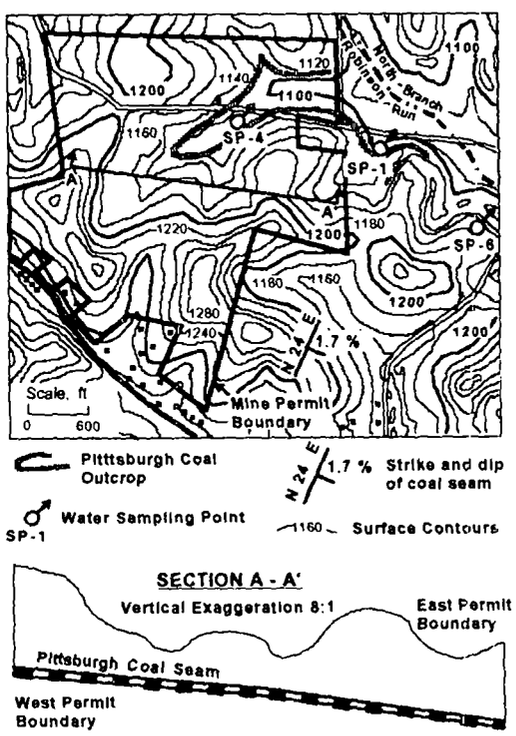

Figure 2. Pre-remining Topography and Sampling Points

was not defined as such in the regulatory arena.

Water Quality Changes

Although no ground water elevation data were available for the 1985-90 period, it can be assumed that the local ground water flow direction was from northwest to southeast due to geologic structure, and was dominated by the presence of the active surface mine and the mined-out Pittsburgh coal seam (see cross-section in Figure 2). Figure 2 also shows the locations of the three surface water sampling points which were closest to the downgradient edge of the mine permit boundary. Sample point SP-4 was intermittent and consisted mostly of surface runoff; it did not show evidence of mining-related contamination before or after the period of mining. Sample point SP-6, located about 1000 ft southeast of the permit boundary, was moderately acidic before, during and after mining. This discharge occurred at the side of a collapsed mine portal structure, and it was obviously connected to a deep mine pool; however, its relationship to the surface mining operation could not be determined. The sampling point of greatest interest was SP-1; prior to mining, this discharge originated as seepage from a swampy area in a natural hollow near the east permit boundary. However, for compliance purposes, sampling was performed at the outlet of a concrete culvert that carried the discharge under the road. Samples were collected on approximately a quarterly basis from 1984 through 1994. Table 1 summarizes the changes in pH, flow rate, and the concentrations of acidity, iron, manganese, and sulfate that occurred at sample point SP-1 before, during, and after the 1985-90 period of surface mining. Note that the discharge was slightly acidic with elevated contaminant levels before mining occurred, and became progressively worse during and after mining. It was unclear whether the source of contamination was the abandoned deep mine workings, the 1950's surface mine spoils just to the east of SP-1, or the new surface mine spoils. Although the discharge did not emanate directly from the surface mine spoil, it was close enough to suggest that a hydrologic connection existed; therefore, the state regulatory agency required the mine operator to initiate treatment of SP-1.

However, the mine operator pointed out that another factor besides the surface mine may have been controlling the degradation of SP-1. At some time shortly after the cessation of surface mining, the landowner to the east of the permitted area made several excavations that began to discharge acidic water at a rate that was greater than that of the original SP-1. Flow from the original SP-1 decreased concurrently. As shown in Figure 3, these post-remining discharges (Seeps A and B) eventually passed through the SP-1 compliance monitoring point. The mine operator contended that the excavations were tapping a portion of the deep mine pool that had not been impacted by surface mining, and that the mining company should not be held responsible for treatment of the discharge.

Initial remediation efforts

The mining company conducted temporary chemical treatment of SP-1 by

Table 1. Water Quality Changes at Sample Point SP-1

Period	# of Samples	Statis-tic	Flow gpm	Lab pH	Net Acid mg/L CaCO3	Total Fe mg/L	Mn mg/L	Sulfate mg/L
Pre-remining	6	Median	4.5	5.00	11	0.44	2.85	520
1/84 - 1/85		Mean	5.4	4.78	43	0.76	3.17	527
During remining	13	Median	1.5	4.90	46	0.90	4.90	790
1/85 - 1/90		Mean	2.2	4.04	132	2.79	4.24	872
After remining	16	Median	18.0	3.27	111	17.49	10.75	1623
1/90 - 1/94		Mean	20.1	3.24	145	21.49	11.22	1355

placing sodium carbonate briquettes in the contaminated stream; however, this was not viewed as a viable long-term solution by either the company or the regulatory agency. To supplement this treatment, the mining company injected a large quantity of anhydrous ammonia into the abandoned mine voids which were believed to be the source of the discharge. The injection wells, labeled A-1 through A-4 in Figure 3, were located between the surface mine highwall and the mine permit boundary, and penetrated the mine voids within 300 ft of the post-remining seeps. It was expected that the alkalinity and aeration introduced via the ammonia injection would eliminate the acidity of the seeps and cause precipitation of metals to occur within the mine voids. However, the water quality of the discharges showed no appreciable change after ammonia injection, thus causing the mining company to review its approach toward remediation.

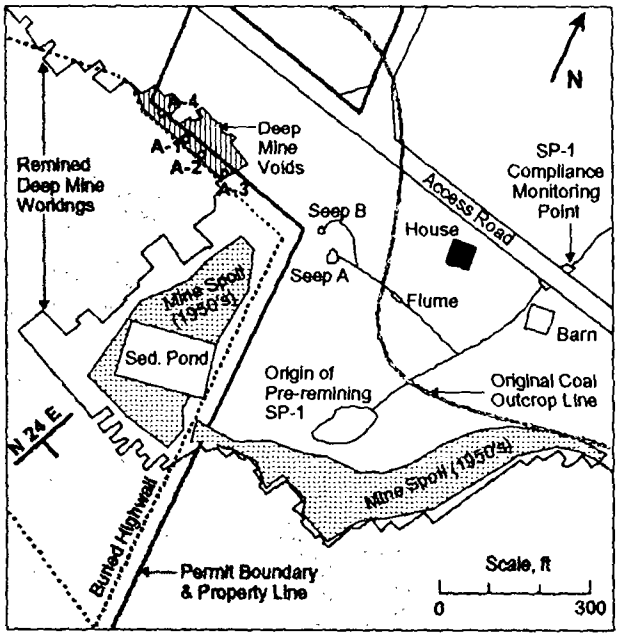

Figure 3.Post-remining Discharge Area

Hydrologic Investigation

During January 1994, the mining company agreed to cooperate in a preliminary hydrologic investigation at the site, with the intent of determining why the initial ammonia injection had failed, and whether the desired remediation could be achieved by injecting cement kiln dust into the abandoned deep mine voids. The following sections describe the data collected and analyzed by the U. S. Bureau of Mines in support of this effort.

Preliminary Data Collection

Water elevations in the mine void wells were measured weekly from March through June of 1994; the combined flow rate of Seeps A and B was also measured weekly with an H-type flume (Figure 3). Water quality samples were collected once every two weeks from the void wells and the seeps during the same period. Site conditions precluded the direct measurement of the flow rates of these seeps, but estimates were obtained via ion mass-balance methods described previously by Aljoe (1995). Concurrently with these efforts, a chemical tracer test was conducted by injecting a concentrated sodium bromide solution into two of the four ammonia injection wells. The following sections describe the results of these investigations.

Water Elevations and Flow Rates. Water elevations in wells A-1, A-2, and A-3 were almost identical (Figure 4), suggesting that these wells penetrated the same mine pool. The mine voids were only partially flooded, with about 2 ft. of water in a void height of about 7 ft. By June 14, the mine void at Well A-1 became dry. It was inferred that a small mine pool existed in the voids penetrated by these three wells, and that the pool was drained by a consistent discharge source.

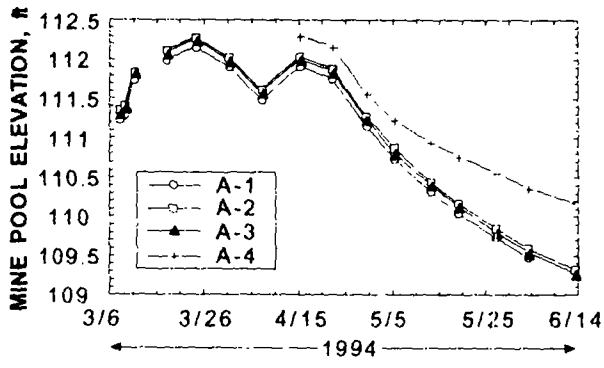

Figure 4. Water Elevations in Mine Void Wells (arbitrary reference datum).

The water elevation in Well A-4 was consistently higher than in the other three mine void wells by 0.5 to 1.0 ft; this difference was considerably greater than the error that could be attributed to surveying and level measurement. Also, the decline in the water elevation at Well A-4 was less steep than in the other three wells during May and June 1994. From these data, it was inferred that another small mine pool existed in the portion of the deep mine penetrated by well A-4. The degree of hydrologic connection between the two pools could not be confirmed; however, the behavior of Well A-4 during May and June suggested that this pool was not directly connected to the same discharge source as the pool penetrated by the other three wells.

The combined flow rate from seeps A and B was directly related to the elevation of the mine pool at wells A-1, A-2, and A-3 (Figure 5). This relationship was not surprising, given that both the pool elevation and flow rate would be similarly influenced by precipitation. Figure 5 shows that the mathematical relationship at this site was exponential (linear plot on log scale); however, mine pool elevation-discharge plots at other sites had exhibited a direct linear relationship (Aljoe, 1994).

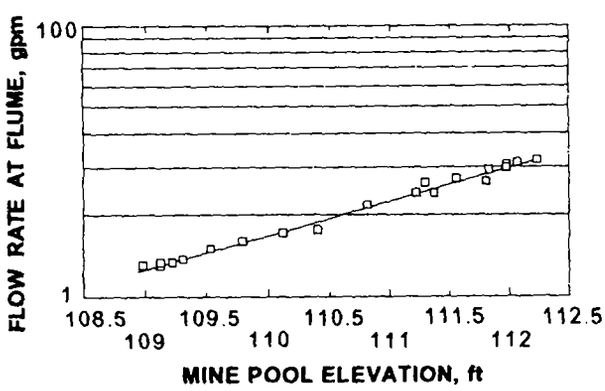

Figure 5. Mine Pool Elevation vs. Discharge Rate.

Another deviation from past experience was that the total discharge rate, about 30 gal/min for much of the study period, was much larger than would be expected from such a small postulated recharge area (about 2/3 acre of surface area above the mine voids). The lack of a visible physical connection between the seeps and the mine voids also suggested that more evidence had to be gathered to verify the hydrologic connection between the mine pool and the contaminated seeps.

Tracer Test. One of the most direct ways to verify the hydrologic connection was to inject a chemical tracer into the mine voids and monitor the seeps for its arrival. The bromide ion was chosen as the tracer because of its conservative behavior in mine waters, the lack of detectable bromide concentrations in the mine voids and the seeps, and the ease of application in the form of a concentrated bromide salt solution. On March 8, 1994, a total of 6.0 kg of NaBr salt (4658 g of Br^-) was added to the mine voids; half of this total was added to Well A-1, and the other half to well A-3. These wells were chosen because Well A-2 still possessed large quantities of ammonia and Well A-4 had not been found at the time the tracer test was begun. If the extent of the pool were limited to the area occupied by the mine voids (approx. 433,000 gal), this amount of bromide would have been sufficient to raise the bromide concentration in the mine pool to a level that was about 15 times the laboratory detection limit of 0.1 mg/L, even if the solute became uniformly distributed within the pool. This ensured that if the mine pool and seeps were indeed hydrologically connected, bromide would be detected at the seeps even if a substantial degree of dilution were occurring.

In general, the results of the tracer test showed that there was a distinct hydrologic connection between the mine pool and the seeps, but that the connection was not as direct or exclusive as first anticipated. Figure 6 shows the bromide concentrations (a) and loads (b) at the three discharge locations. If the mine pool were indeed the only source of water to the seeps, the high discharge rate would result in a pool turnover time (complete passage of the tracer cloud

through the seeps) of about 10 days. However, the peak bromide cncentration did not occur until the twelfth day, and the total time required for the bulk of the tracer to pass through the seeps was about 50 days. This suggested that one or both of the initial assumptions about the flow system was incorrect, i.e., the mine pool was much larger than the area of open mine voids, and/or the monitored portion of the pool was not the only source of water to the seeps.

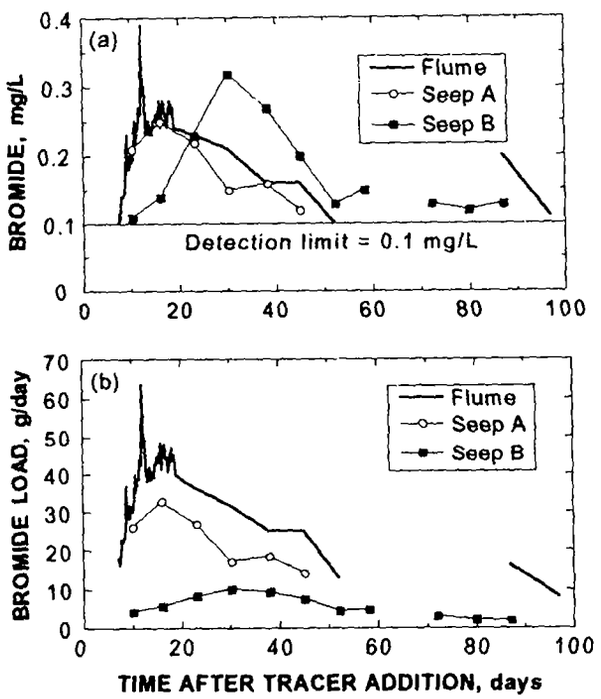

Figure 6. Results of Bromide Tracer Test (a) concentrations; (b) loads.

From the peak bromide concentration and load shown in Figure 6, the average flow velocity from the bromide injection wells to the seeps was estimated to be 12 to 27 ft/day. This velocity is more than an order of magnitude lower than found by Aldous and Smart (1988) for underground mine workings with pure conduit-type flow. Instead, the velocity is closer to those for sealed underground mine workings (Aljoe and Hawkins, 1993), mine spoil (Hawkins and Aljoe, 1992), or glacial sediments (Grisak, et al., 1977). These results suggested that the tracer cloud moved through relatively resistive material (mine spoil and/or unmined, fractured strata) between the injection wells and the seeps.

Analysis of tracer data gathered weekly from the injection wells and seeps showed that the mine pool was hydrologically connected to both seeps, but the degree of connection was different for each seep. Seep A, which consistently accounted for 75% to 80% of the flow at the flume, also appeared to be the most direct outlet for the mine pool. As shown in figure 6, the bromide profiles of Seep A generally paralleled those of the flume, and the bromide load at Seep A was generally greater than at Seep B. Higher bromide concentrations were often found at Seep B, but the path from the mine voids to Seep B appeared to be less direct. The peak concentration occurred two weeks later than at Seep A; Seep B also had a less-pronounced loading peak, and concentrations at Seep B remained detectable until almost 90 days after the test was begun. This suggested a slower, more evenly distributed release of tracer through Seep B.

Figure 7 shows the bromide concentrations in the injection wells over the sampling period, along with the flume concentration curve. The consistently lower concentrations in Well A-3 were attributed to the greater amount of flushing water that had been used in this well. The rates of concentration decline in the two wells were very similar, but the higher initial concentration in Well A-1 allowed bromide to remain detectable until the mine void became dry.

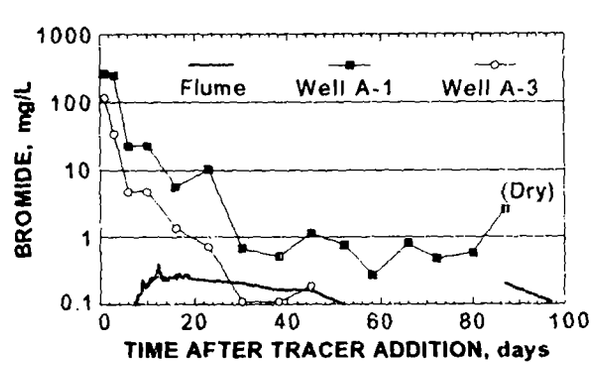

Figure 7. Tracer concentrations in Mine Pool and Discharge.

The fact that bromide concentrations in part of the mine pool were on the order of 1 mg/L while seep concentrations were undetectable indicated that dilution was occurring between the mine voids and the seeps.

Water Quality Data. Initial water quality data collected at the site showed that the most critical difference between the water quality in the monitoring wells and the seeps was the net acidity over the study period, as depicted in Figure 8.

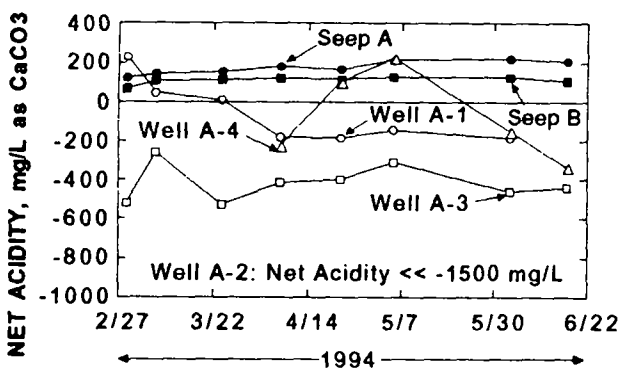

Figure 8. Net Acidity in Mine Pool and Seeps.

Net acidity at the seeps was very stable at 100 - 200 mg/L over the study period; conversely, the mine voids generally possessed strongly alkaline water (negative net acidity = net alkalinity). Although wells A-1 and A-4 were acidic on several occasions, they showed temporal fluctuations that contrasted markedly with the stable pattern of the seeps. Furthermore, Well A-3, which was closest to the seeps and was shown to be hydologically connected to the seeps via the tracer test, was consistently net alkaline by more than 400 mg/L. In order for the seeps to be net acidic, while still receiving recharge from the alkaline mine pool (as verified in the tracer test), one or both of the following must have been occurring: (1) significant geochemical changes were occurring along the flow path; or (2) another source of recharge that was more acidic and probably larger in volume than the mine pool was present.

Table 2 lists the average values of the other important mine water quality parameters over the study period, along with the average cation/anion ratios calculated from the laboratory analyses. The substantial differences in water quality within the mine voids and the lack of tracer detection in wells A-2 and A-4 suggested that there was very little geochemical mass transfer within the mine pool. It would therefore be inappropriate to combine the parameter values across different wells to obtain "average" values for the pool as a whole. In fact, it may be justifiable to exclude Well A-2 completely from the analysis; its anomalous characteristics were attributed to the continued presence of ammonia from the previous injection. Water samples were not routinely analyzed for ammonia; however, the presence of ammonia (as the NH_4^+ cation) was suspected when the very low cation/anion ratio of Well A-2 (see Table 2) was first noted. Several subsequent samples from Well A-2 were found to contain ammonia concentrations of greater than 1000 mg/L. The cation/anion ratio of Well A-3 was also significantly lower than unity, so it is possible (though not verified by laboratory analysis) that ammonia was

Table 2. Average Water Quality Parameters, March - June 1994

Location	pH	Fe, mg/L	Ca, mg/L	Mg, mg/L	Al, mg/L	Na, mg/L	Mn, mg/L	SO4, mg/L	Cations /Anions	Notes
SEEP A	4.37	43.0	324	133	6.0	19.2	13.2	1483	1.00	
SEEP B	3.62	20.2	370	170	20.8	12.6	16.4	1903	0.93	
Well A-1	6.68	73.0	528	227	5.7	17.6	14.4	2203	1.03	Tracer Inj.
Well A-2	8.40	0.8	98	150	0.1	22.7	1.3	2999	0.13	High NH3
Well A-3	6.67	7.7	459	191	0.9	21.2	13.4	2188	0.75	Tracer Inj.
Well A-4	6.40	198.0	532	267	4.4	14.6	21.4	2654	0.97	

also partially responsible for its alkaline character. However, wells A-1 and A-4 had cation/anion ratios near unity even when they were net alkaline, indicating that ammonia concentrations were very low. Therefore, except for Well A-2, it was concluded that the ammonia injection alone was not responsible for the predominantly alkaline character of the mine pool.

Some of the water quality differences shown in Table 2 could have resulted from geochemical reactions along the flow path from the wells to the seeps; however, these reactions alone cannot explain all the differences. For example, the pH and iron concentrations were generally higher in the wells than at the seeps, while the aluminum concentrations were lower. There were also significant decreases in net acidity (Figure 8), calcium, magnesium, and sulfate. Magnesium in mine drainage has been found to be particularly conservative in almost all situations (Hedin, et al., 1994); calcium and sulfate would also be expected to be conservative under these site conditions (Bencala, et al., 1987). The most reasonable explanation for these decreases would be dilution by other recharge sources whose concentrations of these ions were lower.

Additional Data from Spoil Wells

Given the results described above, it was clear that the mine voids and seeps were hydrologically connected, but that the flow rate and water quality of the seeps were heavily influenced by one or more other sources. The most likely other source was the mine spoil; however, in the initial analysis, no hydrologic data were available for the spoil. In order to investigate this possibility, the mining company installed four monitoring wells in the spoil (Wells B-1 through B-4) as shown in Figure 9. Water level and water quality data were collected from the spoil wells for a short period after their installation (from June 17 through August 9, 1994). Slug injection tests were performed to determine the relative hydraulic conductivity of the spoil at the well locations. Longer-term data from these wells were not collected. However, the initial data generally supported the conclusion that ground water in the spoil was contributing to the seep discharges, and formed the basis of the coal company's subsequent remediation effort.

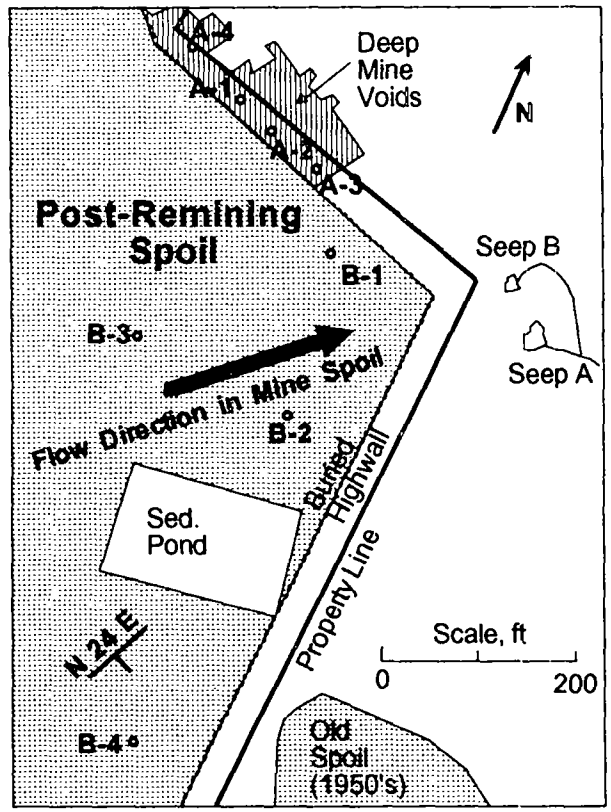

Figure 9. Mine Spoil Well Locations.

The water level in the monitored mine pool was about the same as that of Well B-3, and about 0.5 to 1.0 ft above the levels in wells B-2 and B-1. The water elevation in Well B-4 was 3 to 4 ft below those of the other wells; however, because of its physical separation from the other wells and the seep areas, and the tendency of mine spoil to have multiple flow paths, Well B-4 was considered to be relatively unimportant to the flow system in the seep area. Figure 9 shows the local ground water flow direction as determined from the water levels in wells B-1, B-2, and B-3. Note that the flow direction did not parallel the dip of the pit floor, but was directed toward the seeps.

Table 3. Average Water Quality Parameters, June - August 1994

Location	pH	Net Acid, mg/L CaCO3	Fe, mg/L	Ca, mg/L	Mg, mg/L	Al, mg/L	Na, mg/L	Mn, mg/L	SO4, mg/L	Cations /Anions
SEEP A	4.82	95	52.5	403	170	2.95	23.86	13.25	1788	1.01
SEEP B	3.55	186	41.5	446	216	15.44	14.58	16.10	2207	0.97
Mine Void Wells										
Well A-2	7.29	-2685	0.2	162	167	<0.1	18.44	1.65	2116	0.24
Well A-3	6.56	-422	3.6	496	208	<0.1	19.71	13.63	2141	0.82
Well A-4	6.65	-419	158.1	479	239	<0.1	14.78	18.77	2497	0.85
Spoil Wells										
Well B-1	5.86	-89	84.9	555	213	0.56	26.33	18.41	2091	1.12
Well B-2	**4.28**	**52**	**66.9**	**382**	**153**	**0.72**	**30.18**	**12.57**	**1617**	**1.09**
Well B-3	6.64	-152	89.6	647	211	0.11	31.73	17.34	2111	1.18
Well B-4	7.19	-284	1.2	346	75	1.07	34.85	2.36	878	1.05

Table 3 summarizes the water quality data (average of 2 to 4 samples per well) collected during the study period after the spoil wells were installed. Like the mine voids, the spoil contained primarily alkaline water at the well locations. However, the well closest to the seep area (Well B-2) contained water that was consistently net acidic; although the concentration of acidity was not as great as that of the seeps, it confirmed that acidic water was indeed present in the spoil. Furthermore, the concentrations of the calcium, magnesium, and sulfate in Well B-2 were lower than those of the seeps, as would be expected if the mixing scenario postulated above were taking place.

Table 4 summarizes the hydraulic conductivity data collected from slug injection tests in the spoil wells. As would be expected for mine spoil, the hydraulic conductivity values had a broad range; the hydraulic conductivities of the wells closest to the seep area (B-1 and B-2) were more than an order of magnitude greater than those of the other two wells. Although the range of influence of a slug test is not large (on the order of a few feet), the fact that the hydraulic conductivity of the acidic well (B-2) was the greatest suggested that it could lie along or near a preferred flow path for acidic water originating in other parts of the spoil and/or the undisturbed deep mine workings upgradient of the remined area.

Table 4. Hydraulic Conductivity of Mine Spoil Wells

Well	Hydraulic Conductivity, ft/sec
B-1	3.91E-04
B-2	6.12E-04
B-3	3.28E-05
B-4	9.27E-08

Conceptual Description of Flow System

While the data collected in the hydrologic investigation were far from conclusive, a preliminary conceptual description of the flow system at the site was developed from the available data. Given the high carbonate content of the mine overburden, the most likely source of the acidic water at the seeps was the large undisturbed portion of the abandoned deep mine workings outside the remined area (see Figures 1 and 3). Recharge from the mine voids to the spoil could occur either from unflooded workings northwest (upgradient) of the spoil, or lateral (along strike) inflow from a large mine pool to the southwest of the mine permit area. Existence of such a mine pool could be inferred from the large, consistent, acidic discharge at SP-6 (see figure 2). Under "normal" conditions, the resistance to flow of the mine spoil and the undisturbed material at the eastern edge of the remining permit boundary would have

been much less than that of the areas where the mine voids were very close to the surface (such as at SP-6). As such, very little flow would be expected at the east permit boundary. However, the excavations made by the landowner could have decreased the overall resistance to flow in that area. Under these conditions, acidic water that entered the spoil from the deep mine could have passed relatively quickly, and without much chemical alteration, through large conduits that commonly exist in spoil to emerge at Seeps A and B.

The predominantly alkaline character and associated ion concentrations of the water in the mine spoil and void wells (except for the influence of ammonia in well A-2, as discussed earlier) were indicative of mine drainage that had been neutralized by calcium carbonate. Under the flow scenario described above, much of the recharging water would follow a comparatively direct path to the seeps, with relatively little modification of water quality resulting from its contact with the alkaline mine spoil. The water quality of Well B-2 may be indicative of a portion of the spoil that is relatively close to one of these conduits. Another percentage of the inflowing water would follow a more circuitous path to the seeps; the pH and alkalinity of the water would increase in proportion to the contact time with the alkaline spoil. Calcium and perhaps magnesium concentrations would increase, while other indicators of mine drainage contamination (sulfate and manganese) would remain high. The behavior of iron would be much more complex; for example, ferrous iron in the mine water would be relatively unaffected by passage through the alkaline spoil under anoxic conditions. However, ferrous iron that oxidized to the ferric state would precipitate immediately due to the high pH conditions. Water that entered the small pocket of mine voids at wells A-1 through A-4 could "pool" in this area prior to exiting through the seeps via the unmined strata and/or the spoil in the vicinity of Well B-1.

Remediation Efforts

The results of the hydrologic investigation indicated that although the evidence was not clear-cut (i.e., a source of water of extremely acidic character was not positively identified) the the acidic water at the seeps was almost certainly passing through the mine spoil. Conditions of the mining permit dictated that since the seeps were hydrologically connected to the mine spoil, the discharges had to be treated to SMCRA effluent standards. Since the acidic quality of the seeps probably resulted from the lack of sufficient contact time between the mine water and the alkaline spoil material, treatment of the seeps via an anoxic limestone drain (ALD) was chosen. In January 1995, the mining company excavated a trench within the spoil (Figure 10) to intercept the inferred area of acidic water, and installed an ALD along the base of the old pit floor.

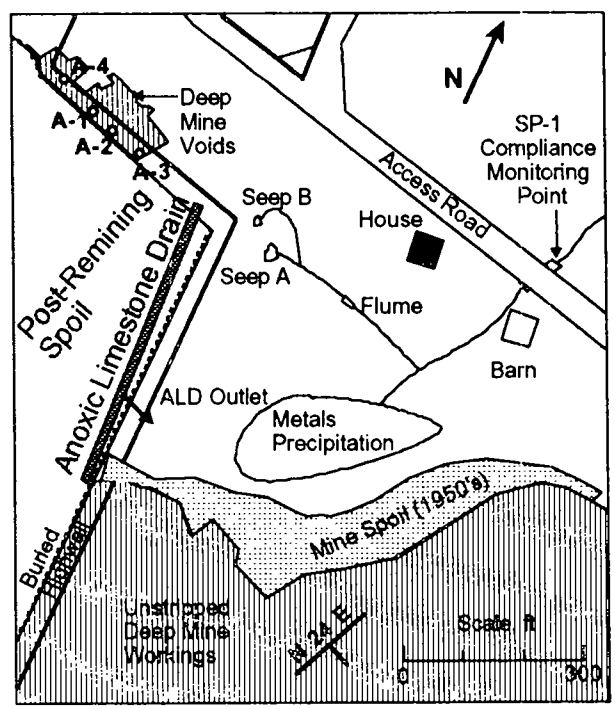

Figure 10. Location of Anoxic Limestone Drain in Remined Spoil.

This approach was considered to be technically feasible because the iron at the flume location was almost entirely in the unoxidized (ferrous)

state, and the aluminum concentrations were not excessive. Furthermore, permission of the adjacent landowner was not required for this activity because it took place within the original permit boundary.

As the excavation of the trench progressed northward, flow rates from the trench increased suddenly to several hundred gallons per minute. This suggested that water had pooled in the spoil, and was released as the primary storage conduit(s) were intercepted by the trench. Flow rates remained very high for almost a week before decreasing to the rate that existed before the excavation began (about 15-20 gpm). The bottom of the 500 ft long, 12 ft wide trench was filled with 500 tons of limestone (bed depth approx. 2 ft). Piping was installed above the limestone to allow it to remain inundated while diverting the discharge away from seeps A and B. The drain pipe discharged into the swampy area of the original SP-1, where sufficient space for precipitation of metals was available. After backfilling of the trench was completed, discharge from the ALD outlet pipe continued while flow from seeps A and B ceased. Flow rates from the ALD were not measured; however, visual estimates by both the state inspector and the mine operator indicated that the flow was about the same as that of the seeps prior to construction of the ALD.

Table 5 summarizes the water quality data at the ALD outlet and SP-1 during the 5-month period after the drain was completed. As expected, alkalinity concentrations at the ALD outlet were consistently greater than 100 mg/L. Iron concentrations were lower in the ALD outlet than in the spoil wells during the previous year (Table 3), suggesting that iron may have been precipitating in the ALD. There were no concurrent spoil well data to compare to those in Table 5, however, so this could not be concluded definitively. It was clear that almost all of the iron leaving the ALD was being precipitated prior to reaching SP-1. It also appeared that significant amounts of manganese were being precipitated; this is consistent with the results of Hedin, et al. (1994) which indicate that manganese can be removed in wetland systems if ferrous iron concentrations are very low.

Discussion

The remediation effort described above, which appears to be successful thus far, may not have been attempted without the additional knowledge gained through the hydrologic investigation. When the contaminated off-site seeps first occurred, there was a substantial difference of opinion between the mine operator and the regulatory agency with respect to the origin of the seeps and the responsibility for their abatement. At first, the investigation yielded results that were somewhat contradictory. Although a chemical tracer test confirmed the hydrologic connection between the seeps and a portion of abandoned deep mine workings that were not disturbed by the mineoperator's remining efforts, other hydrologic and water quality data

Table 5. Water Quality Parameters after ALD Installation, March - August 1995

Location	# of Samples	Statis-tic	pH	Total Alk. mg/L CaCO3	Net Acid mg/L CaCO3	Fe mg/L	Al mg/L	Mn mg/L	SO4 mg/L
Anoxic drain outlet	16	Mean	6.28	172	4	20.85	1.48	10.33	1506
		Median	6.25	174	0	17.45	0.93	10.55	1511
Culvert outlet SP-1	11	Mean	7.49	208	0	0.99	0.64	4.48	1336
		Median	7.50	146	0	0.42	0.50	5.67	1331

Alkalinity values in this table are total alkalinities; they are not directly comparable to the "negative net acidities" shown in Table 3 and Figure 8 because of differences in laboratory measurement procedures

suggested that the seeps were influenced more heavily by other sources that had not yet been identified. This explained the failure of the mine operator's initial remediation effort and discouraged further remedial efforts involving the abandoned mine voids. Further investigation revealed that the ground water in the mine spoil was somewhat acidic; this was in direct contrast to the mine operator's experience at several other remining sites in the same area. The degree of acidity found in the spoil water was not as severe as that of the seeps, but the hydrologic and water quality data were sufficient to convince the mine operator that remedial measures would have to be taken. The mine operator used the available data to implemement a remediation scheme which involved re-excavating the spoil to intercept the acidic water and routing the water through a passive treatment system (ALD and wetlands). Through these efforts, a situation which could have resulted in lengthy litigation and a legally enforced, inefficient temporary treatment scheme was resolved by a relatively cost-effective, long-term treatment system.

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WATER QUALITY IMPROVEMENTS RESULTING FROM FBC ASH GROUTING OF BURIED PILES OF PYRITIC MATERIALS ON A SURFACE COAL MINE

by

Joseph Schueck, Mike DiMatteo, Barry Scheetz, and Mike Silsbee

Abstract. A 37 acre surface coal mine in Clinton County, Pennsylvania, was mined and reclaimed between 1974 and 1977. Buried pyrite-rich pit cleanings and tipple refuse were found to be producing severe acid mine drainage (AMD). The pyritic material is located in discrete piles or pods in the backfill. The pods and the resulting contaminant plumes were initially defined using geophysical techniques and confirmed by drilling. Isolating the pyritic material from water and oxygen will prevent AMD production. A grout, composed of fluidized bed combustion (FBC) ash and water, was used in two different approaches that attempted pyrite isolation. Pressure injecting grout directly into the buried pods to fill the void spaces within the pods and coat the pyritic materials with a cementitious layer was the first approach. In the second approach, pods that would not accept grout because of a clay matrix were capped with the grout to isolate the pyrite from percolating water. The grout was also used in certain areas to blanket or pave the pit floor to prevent dissolution of clays, which are a suspected primary source of high aluminum (Al) concentrations at this site. Monitoring wells have been sampled since 1990 to monitor changes in the water quality resulting from grouting efforts. Grouting occurred during the summers of 1992 and 1993. Statistically significant water quality improvements have been noted as a result of the grouting, although results are varied. Any water quality improvements resulting from the grouting are expected to be permanent because of the nature of the cementitious grout.

Additional Key Words: Magnetometry, Electromagnetic Terrain Conductivity, VLF, AMD, Acid Mine Drainage, Abatement, Fluidized Bed Combustion Ash, Grouting, Trace Metals

Introduction

Effective in-situ abatement technology requires that the source(s) of the AMD production first be located. AMD source location can be done with little difficulty on many surface coal mines using geophysical mapping techniques (Schueck, 1988, 1990, 1994). Pyritic material such as tipple refuse or pit cleanings are often placed in discrete piles and buried. When placed in discrete piles or pods and not properly isolated from infiltrating precipitation or groundwater, severe localized AMD production may result. Under these conditions geophysics can detect the pods and the resultant AMD plume. Geophysical detection of the AMD sources would be less successful on sites where the pyritic material was disseminated throughout the backfill.

Abatement technology can be applied once the precise locations of the acid producing materials buried within the backfill are determined. The contact of pyrite with oxygen and water is needed to produce AMD. In-situ AMD abatement will result through permanent removal or isolation of any one of these three (pyrite, water, or oxygen). Fluidized bed combustion (FBC) ash grout was used to isolate pyrite from oxygen and water in this research effort.

This paper provides further information on an AMD mitigation effort that was described in the 1994 ASSMR Proceedings (Pittsburgh, Pa.). This earlier paper should be referenced, for detailed discussions of the geophysical methods and grouting techniques used. The primary purpose of the current paper is to discuss water chemistry data.

Site Description

The project site is located in north-central Pennsylvania in the Sproul State Forest, Clinton County. This site was permitted and mined by the mountain top removal method between 1974 and 1977. The Lower

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Kittanning coal seam was present in two splits separated by 10 to 20 feet of clay. Only the upper split was mined, leaving a thick underclay as pavement. The coal was overlain by black shale capped by a sandstone unit. The black shale is thought to be pyritic and acid producing. Infiltrating precipitation is the only source of groundwater. Acidic discharges developed soon after reclamation and were first noted after a fish kill in 1978. The discharges (surface and underground), estimated to average 35 gpm, destroyed five miles of native trout streams. The operator was unable to maintain treatment facilities and forfeited $9,400 in bonds.

Technical Approach

The primary objective of this AMD abatement approach was to isolate pyritic material from water and oxygen. Geophysical investigations were used to locate pods of acid-producing materials, evaluate local hydrology, and monitor grout propagation. Monitoring wells were used to evaluate water quality across and adjacent to the site. Water quality was also monitored at a toe-of-spoil seep and in receiving streams.

Fluidized bed combustion (FBC) ash was the material selected to be used in this isolation approach. FBC ash was chosen primarily because of its pozzolanic (or cementitious) properties as well as for economic reasons. A low viscosity grout was pressure injected into the geophysically identified zones (pods) as a means of encapsulating the pyritic materials with a cementitious coating. Diversion grouting was also used. This approach included capping of some of the pods and pit floor paving. The grouting should generate zones of low permeability and redirect groundwater flow, significantly reducing water contact with the pyrite. Although the spoil across the entire site was thought to be acid producing, the grouting effort was directed only toward reducing the AMD being produced by the buried piles of refuse and pit cleanings. The ultimate goal was to restore five miles of lost stream by improving the quality of the discharges. Final evaluation of the receiving streams has not yet been completed.

Pre-grouting Investigations

Geophysical Investigation

Several geophysical mapping techniques were used for site characterization. These techniques included electromagnetic terrain conductivity (EM), magnetometry, and very low frequency (VLF). EM was used to map the location of the AMD plume throughout and off the site. The piles of buried refuse and pit cleanings were located with magnetometry. VLF was used to map bedrock fracture zones beneath and adjacent to the site (Schueck, 1994).

EM mapping and drilling data indicate the general groundwater flow path through the site to be from the northwest to the southeast (Figure 1). This is the down-dip direction. EM mapping indicates pooling of pit water in the down-dip portions of the site. These down-dip portions are referred to as the east and south lobes. The EM mapping further indicates groundwater pollution plumes from the site at three locations where toe-of-spoil (surface) discharges were not present. VLF mapping indicates the presence of fractures in the pit floor which coincide with the pollution plumes.. These fractures apparently channel AMD from the mine spoil to a major joint system beneath the site. The joint system then conveys the AMD discharges to Rock Run as base flow, some 250 feet lower in elevation and east of the site.

A toe-of-spoil discharge is present beyond the south lobe. The discharge varies from 2 to 20 gpm and appears as a diffuse seep. It is at an elevation equal to the lower, unmined split of coal. This discharge flows into Camp Run, a tributary of Cooks Run.

Magnetometer mapping was used to determine the locations and configurations of concentrated pods or piles of pyritic materials such as tipple refuse. Magnetic anomalies indicate the aerial extent of the pods of pyritic material buried beneath the surface. In Figure 1, polygons are used to indicate the location and extent of the pods of pyritic material and to show the extent of grouting.

When the magnetic anomaly map was overlain with the EM map, it was observed that conductivity values were highest at or adjacent to the magnetic anomalies. The conductivity values gradually decreased in the direction of groundwater flow. This observation is consistent with severe AMD production within the buried pods of pyritic material, followed by dilution as the AMD migrates further away from the pods.

Water Sampling

Forty two monitoring wells were drilled on and adjacent to the site. The wells were located using the results of the combined geophysical mapping. Wells located on the site were drilled through spoil to the pit floor, with depths ranging from 10 to 40 ft. Monitoring wells located adjacent to the site were drilled into the unmined lower split of the Lower Kittanning coal seam. This initial drilling effort also confirmed the locations of the pods of refuse and pit cleanings identified with

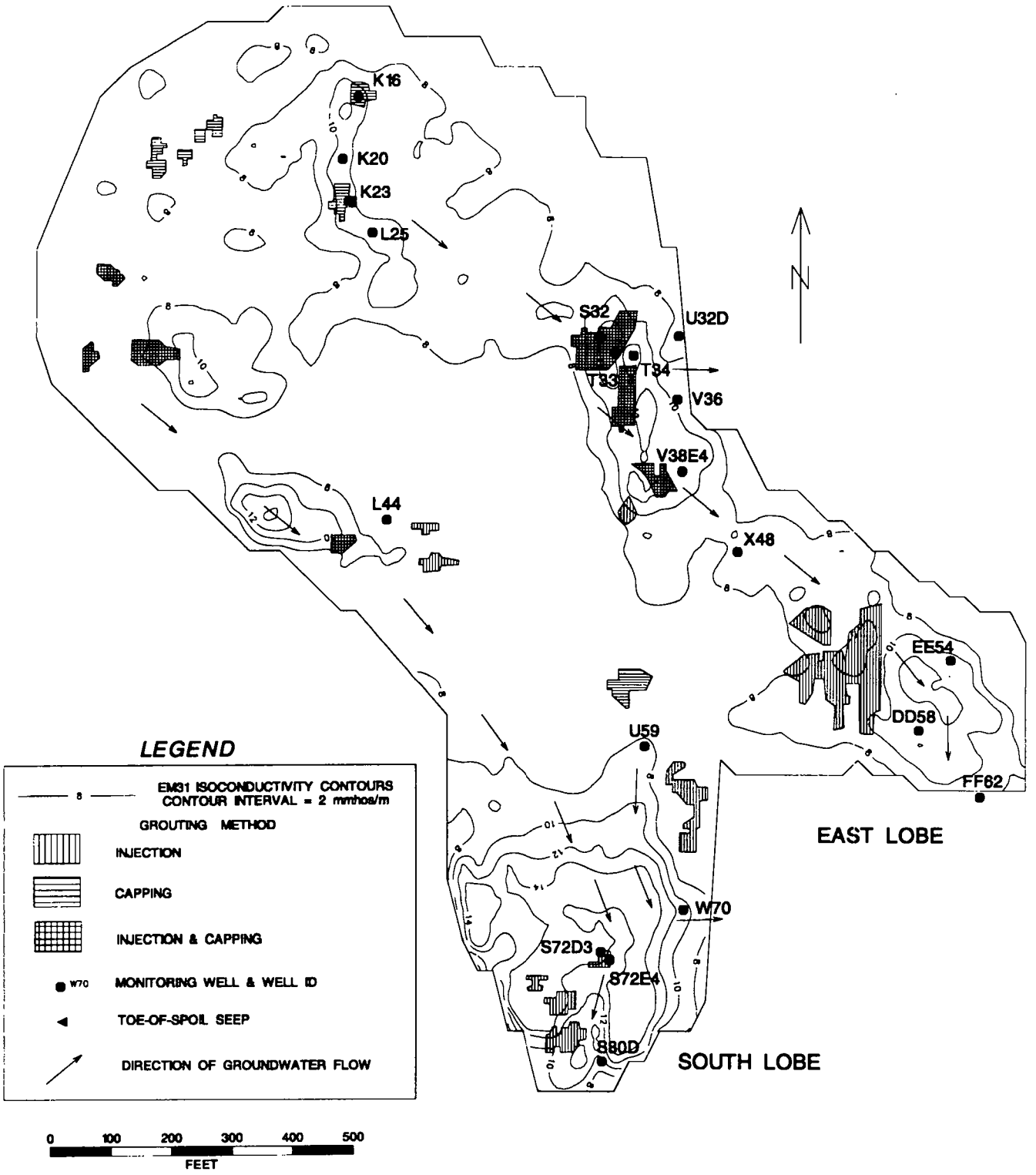
K16
K20
K23
L25
S32
U32D
T33
T34
V36
V38E4
L44
X48
EE54
DD58
FF62
U59
W70
S72D3
S72E4
S80D
EAST LOBE
SOUTH LOBE
N
LEGEND
EM31 ISOCONDUCTIVITY CONTOURS
CONTOUR INTERVAL = 2 mmhos/m
GROUTING METHOD
INJECTION
CAPPING
INJECTION & CAPPING
W70 MONITORING WELL & WELL ID
TOE-OF-SPOIL SEEP
DIRECTION OF GROUNDWATER FLOW
0 100 200 300 400 500
FEET

FIGURE 1

D3

magnetometry. Water quality monitoring was initiated in 1990 and continued through 1995. The grouting occurred during the summers of 1992 and 1993. Sampling of the monitoring wells was done on approximately a monthly schedule from April through November each year. The site is generally not accessible during the winter months. Monitoring wells discussed in this report are shown on Figure 1. Water sampling was also performed at the only surface discharge, D3, which is a toe-of-spoil seep located 200 feet south of the site.

Water Quality

Table 1 lists pre-grouting water quality for the toe-of-spoil seep and selected monitoring wells which are considered to be representative of the site (Figure 1). High, low, and mean pH values and metals concentrations are presented. These data are based on 13 to 21 sampling events.

The toe-of-spoil seep, labeled D3, and well FF62 represent the pre-grouting water quality which exits the site from the south and east lobes, respectively. Well FF62, located off-site and adjacent to the east lobe, intercepts a subsurface discharge plume (as identified by EM survey) suspected to enter Rock Run as base flow. Also, well FF62 samples water in the lower, unmined split of coal. The poor water quality demonstrates fracture communication between the pit floor and lower coal seam. D3 also appears to be coming from the lower coal seam. Comparison of the mean concentrations indicates that the water discharging from the east lobe (D3) is of poorer quality than that discharging from the south lobe (FF62). Drainage from the majority of the pods of pyritic material flows towards the east lobe.

Monitoring Point D3--Toe-of-spoil seep area N=16

	Lab	TDS	SO_4	Acid	Fe Tot	Fe^{+3}	Al	Mn	Cd	Cu	Cr	Zn	Temp
	pH	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	ug/L	ug/L	ug/L	mg/L	deg C
High	2.7	8828	3749	3520	543	385	348	62.3	66	852	269	4.2	n/d
Low	2.4	4540	1245	2220	183	149	205	28.4	1.9	399	125	2.5	n/d
Mean	2.49	6475	2571	2995	321	254	268	48.3	25.5	612	200	3.4	n/d

Monitoring Well FF62 - East Lobe - Groundwater Discharge Point N=15

	Lab	TDS	SO_4	Acid	Fe Tot	Fe^{+3}	Al	Mn	Cd	Cu	Cr	Zn	Temp
	pH	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	ug/L	ug/L	ug/L	mg/L	deg C
High	2.6	12706	5773	6440	1500	1294	562	67.3	167	1110	301	7.5	13.1
Low	2.2	3404	1272	1940	386	246	114	14.8	29	611	142	2.1	11.6
Mean	2.32	7970	3477	4088	876	737	256	39.2	83	806	221	4.3	12.0

Monitoring Well L44 - Lower Kittanning Spoil N=21

	Lab	TDS	SO_4	Acid	Fe Tot	Fe^{+3}	Al	Mn	Cd	Cu	Cr	Zn	Temp
	pH	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	ug/L	ug/L	ug/L	mg/L	deg C
High	3.1	16376	5376	7020	1390	1346	493	66.6	190	1300	229	12.0	15.7
Low	2.2	1555	828	1120	221	14	120	13.7	19	286	70	2.5	11.2
Mean	2.47	7920	2958	3828	747	598	236	48.1	111	751	163	6.8	12.9

Monitoring Well K23 - Severe AMD Production in Pods N=13

	Lab	TDS	SO_4	Acid	Fe Tot	Fe^{+3}	Al	Mn	Cd	Cu	Cr	Zn	Temp
	pH	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	ug/L	ug/L	ug/L	mg/L	deg C
High	2.2	128205	25110	23900	5690	2320	2240	79.0	1190	7410	1816	39.0	14.9
Low	2.0	33706	9868	18280	3320	120	281	30.3	128	3360	593	10.8	10.2
Mean	2.1	46352	15639	21315	5437	1200	1515	60.5	610	5950	1108	27.6	13.3

Monitoring Well X48 - Combined Spoil and Pyritic Pods N=14

	Lab	TDS	SO_4	Acid	Fe Tot	Fe^{+3}	Al	Mn	Cd	Cu	Cr	Zn	Temp
	pH	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	ug/L	ug/L	ug/L	mg/L	deg C
High	2.8	25372	9950	9760	2210	1816	690	125	368	3370	687	10.5	13.8
Low	2.2	947	4763	3780	342	0	299	7.8	11	1680	492	6.2	11.2
Mean	2.37	14085	6991	7470	1707	439	492	72.8	227	2543	559	8.7	12.7

Table 1 Pre-grouting Water Quality

Monitoring well L44 represents the pre-grouting water quality resulting from the Lower Kittanning spoil. L44 is not located downgradient of buried refuse or pit cleanings. Monitoring well L44 is considered a suitable control well to monitor normal water quality variations on the site because it is not influenced by the grouting activities. Flow from L44 would be toward the south lobe and eventually to the toe-of-spoil seep.

Monitoring well K23 demonstrates that the pods of refuse or pit cleanings can be sources of severe, localized AMD production. Concentrations of the mine drainage parameters in the water in and adjacent to these piles are often several times greater than the concentration of the same parameters in the discharge or elsewhere on the site. K23 is located within a pile of buried tipple refuse. It is located on the up-dip portion of the mine site and is also at the upper end of a pollutional plume as defined by EM. The mean concentrations of the AMD parameters at K23 are four to six times greater than the concentrations of the same parameters in L44, which represents AMD generated by the spoil alone. This implies that enhanced AMD production from the pyritic material at K23 and similar locations significantly contribute to the degradation of the final discharge quality. The isolation of these subsurface pods of pyritic material from water and/or oxygen should improve the final discharge quality. This was the premise for this research effort.

Monitoring well X48 is located several hundred feet downgradient of piles of pyritic material, but is within the flow path of mine drainage as it migrates through the site towards the east lobe. The water quality sampled from this well would be influenced by both the Lower Kittanning spoil as well as AMD formed in the buried pods of pyritic materials. Note that the mean concentrations of the mine drainage parameters in well X48 are about double the mean concentrations of L44. X48 indicates dilution of the severe AMD as it migrates and mixes with less severe mine drainage.

Certain trace metals also contribute to the pollution from this site. Elevated concentrations of zinc, copper, chromium, cadmium, and arsenic were common in the drainage from this site. The concentrations of other trace metals, such as lead, nickel, and selenium were generally below detection limits.

FBC Ash Characterization

The quality of the coal combustion ash is extremely important, particularly when it is being placed into an acidic environment. A poor quality ash which leaches undesirable trace metals could make an existing environmental problem worse. The characterization of the FBC ash used at the site was completed at the Penn State Materials Research Lab (Zhao,1993). Chemical analysis of the ash is as follows: Al_2O_3 - 12.51%, CaO - 38.03%, SiO_2 - 23.91%, SO_2 - 16.02%. The ash was tested using the EPA's Toxicity Characteristic Leaching Procedure (TCLP). All elemental contents of the ash fell within the established guidelines of the TCLP.

Mixed with only water, the FBC ash forms a low strength cement. After 20 days, at a water to solids ratio of 0.5, an unconfined compressive strength of 1920 psi is developed. The compressive strength continues to increase slowly to slightly over 2000 psi in 90 days (Zhao, 1993).

Field Operations

Grout Applications

Only those pods of pyritic material identified with magnetometry were targeted for grouting. The grout injection wells were installed on 10 foot centers using 2 1/2 inch, perforated, schedule 40 PVC casing. The injection wells were installed in August 1992. Grouting operations began September 1, 1992 and continued through the end of October 1992. The grouting operation resumed in June 1993 and was completed in August 1993. Grout injection wells were located within the polygons indicated in Figure 1. The amount of grout accepted by the wells ranged from less than 0.3 to 83 yd^3. Approximately 4500 yd^3 of grout were used on this project.

Originally only pressure injection directly into the pods was planned for the isolation of the pyritic materials. However, soon after the grouting operations began, it became evident that this method would not work for all the pods within the site. Several of the piles refused to accept grout. Excavation within these pods showed the pyritic material was within a clayey matrix. These piles were capped with grout to divert infiltrating precipitation. The spoil above the piles was excavated. The excavated area was then pooled with fly ash grout. After the grout hardened, the excavated area was backfilled and regraded. However, the capping would not be effective in preventing lateral flow along the pit floor from coming into contact with the pyritic materials. Several of the pods were both capped and grouted. This combined application occurred where several of the wells within the pod accepted little or no grout.

Aluminum concentrations in excess of 1000 mg/l at various locations within the spoil suggested that the clay pit floor is a primary source of Al. Therefore, paving areas of the pit floor with a grout slurry was another water diversion technique applied at this site. The purpose was to isolate the pit floor clays from contact with the AMD within the spoil. Success in this approach depends upon a high permeability of the spoil materials covering the pit floor. Well L25 shows the results of the paving activity and is discussed later in this report.

Post Grouting Operations

Water Quality Changes

Grouting occurred during the summers of 1992 and 1993. Water quality was monitored on approximately a monthly basis, April through November, from 1990 through 1995. In order to test the effectiveness of fly ash injection on water quality improvement, a series of one-tailed t-tests were computed for the water quality variables sampled from the monitoring wells. A t-test represents a test of significance that compares the mean and standard deviation of one group of samples to that of another to test if both groups came from the same population. The one-tailed t-test further compares not only if there is a significant difference between the two groups but if one group mean is significantly greater than the other. The $p <= 0.05$ level of significance was used as the rejection level of the null hypothesis for all tests. Readers are referred to Davis (1986) and Krumbein and Graybill (1965) for additional information regarding the t-test.

For the tests, the water quality data for the respective monitoring wells were divided into two groups: the data sampled before fly ash injection occurred and the data sampled after fly ash injection was completed. It was obvious in some of the wells that the grout supernatant was impacting the water quality during and shortly after the grouting effort. These water samples were excluded from the data analysis.

During the summer of 1995, drought conditions existed at the site. The water quality of the control well, L44, changed dramatically. Mean SO_4 values increased four-fold during the summer of 1995 and concentrations of TDS, acidity, Al, and Fe more than doubled. Moderate increases in concentrations of the mine drainage parameters were noted in most of the monitoring wells across the site during 1995. However, the increases are slight when compared to L44. The conclusions within this paper are based on mean values of water chemistry parameters including the data from the summer of 1995. No effort has been made to correct this data for the drought conditions. Water monitoring at the site is planned for several more years.

Table 2 lists the percentage reduction in mean concentrations for several of the mine drainage parameters for the three different applications. The applications included 1) injection only, 2) capping only and 3) a combination of injection and capping Data are included for both wells located within the pod (IP) and for wells located downgradient (DG) of the pod. The percentage change in concentrations for L44, the control well, are included for comparison.

Data for the first four wells in Table 2 reflect water quality changes resulting from injection grouting only. There is within-pod water quality data for only one pod that was subjected to grout injection. There are two wells located within this pod, S72D3 and S72E4 (Figure 1). Both wells exhibited modest reductions in the mean concentrations of the common mine drainage parameters ranging from 23 to 52%. These reductions are statistically significant with the exception of SO_4 in well S72E4. Significant reductions in trace metal concentrations from 42 to 88% were observed as well. This suggests that AMD production was reduced within the pod as a result of the grout injection. Wells EE54 and DD58 are located downgradient of a pod that was only pressure injected. These wells exhibited 16 to 37% statistically significant reductions in mean concentrations of the common AMD parameters. The exception is SO_4 which remained virtually unchanged. Significant trace metal reductions were also noted, although percent reductions were less than observed in the within-pod wells. Wells EE54 and DD58 are situated where the flow of AMD from the site can influence them, however.

The second situation are those pods that would not accept the pressure injected grout and were therefore capped. Wells K16 and K23 are located within pods that were only capped. The changes that occurred in this situation are quite mixed, ranging from a 44% non-significant increase in mean SO_4 concentration in well K23 to a 37% decrease in mean Al concentration in well K16. Few of the chemical changes in these two wells were statistically significant Decreased infiltration because of capping may have abated some of the AMD production occurring in the upper portion of the pod. However, lateral flow of water along the pit floor can still react with the pyritic material at the base of the pods

Application: Injection Grouting Only													
Well	Loc*	TDS	SO_4	Acid	Fe Tot	Fe^{+3}	Al	Mn	Cd	Cu	Cr	Ca	Temp
S72D3	IP	***39%***	***30%***	***40%***	***36%***	***52%***	***41%***	***50%***	***88%***	***54%***	***47%***	***-83%***	***5%***
S72E4	IP	***35%***	23%	***35%***	***39%***	***52%***	***27%***	***42%***	***85%***	***42%***	***44%***	***16%***	***7%***
EE54	DG	20%	3%	***21%***	***29%***	21%	***21%***	***30%***	***82%***	***22%***	***32%***	7%	***12%***
DD58	DG	***16%***	-4%	***20%***	***25%***	***26%***	***23%***	***37%***	***76%***	***22%***	***26%***	***-52%***	***6%***
Application: Capped Only													
Well	Loc *	TDS	SO_4	Acid	Fe Tot	Fe^{+3}	Al	Mn	Cd	Cu	Cr	Ca	Temp
K16	IP	0.2%	1%	5%	-0.4%	-15%	***37%***	-40%	***88%***	32%	26%	***-122%***	4%
K23	IP	21%	-44%	***13%***	11%	-35%	19%	1%	***75%***	11%	***26%***	***-62%***	2%
K20	DG	***51%***	***60%***	***57%***	***53%***	***50%***	***47%***	33%	***90%***	***56%***	***50%***	6%	7%
U59	DG	***29%***	12%	***34%***	***30%***	***33%***	***32%***	***33%***	***87%***	***38%***	***44%***	14%	***10 %***
Application: Grouted and Capped													
Well	Loc *	TDS	SO_4	Acid	Fe Tot	Fe^{+3}	Al	Mn	Cd	Cu	Cr	Ca	Temp
T33	IP	***42%***	30%	***46%***	***59%***	***64%***	***52%***	***44%***	***88%***	***62%***	***66%***	***-236%***	***7%***
T34	DG	***93%***	***94%***	***95%***	***92%***	***91%***	***97%***	36%	***97%***	***97%***	***91%***	***50%***	9%
V38E4	DG	19%	4%	***34%***	***40%***	***-169%***	***43%***	***38%***	***87%***	***53%***	***52%***	***-63%***	6%
Application: Control Well — No Grouting Influence													
Well		TDS	SO_4	Acid	Fe Tot	Fe^{+3}	Al	Mn	Cd	Cu	Cr	Ca	Temp
L44		***-106%***	***-248%***	***-122%***	***-165%***	***-208%***	***-103%***	-7%	17%	***-67%***	-34%	-8%	-2%

Table 2. Percent reductions in mean concentrations as a result of the three grouting applications. Negative values indicate a percent increase in mean concentration. Statistically significant changes where P <=0.05 are indicated in bold italics. Control well L44 is included for comparison. *Location IP=In-pod DG=Downgradient

Monitoring well K20 is located between the K16 and K23 pods and may receive drainage from both, based upon the EM mapping. Significant reductions in mean concentrations of 47 to 60% are observed in K20 for the common AMD variables, with the greatest reduction being that of SO_4. It is thought that a decreased groundwater contribution as a result of capping pods K16 and K23 is responsible for the observed improvement in K20 rather than AMD abatement within the pods, i.e. a reduction in load.

U59 is downgradient of a capped pod. Significant decreases in mean concentrations of 29 to 34% are observed in U59. Unlike the pods at K16 and K23, the pod upgradient of U59 is believed to be located high in the spoil and free of influence from lateral groundwater movement. Improvements in water quality in U59 are likely a result of AMD abatement in the upgradient pod because of capping. The reductions in well U59 are similar to those observed for wells DD58 and EE54. Trace metal reductions are also quite significant.

The final case is where pods were both injection grouted and capped. Well T33 is located within a pod that accepted grout quite well, but was capped as an assurance measure. Significant decreases in mean concentrations range from 42 to 64% within the pod, which suggests a reduction in mine drainage production. Well T34 is immediately downgradient of this pod. Reduction in concentrations of 43 to 97% are noted for this well. The data suggest a reduction in AMD production within the pod and minimal migration of mine drainage from the pod toward the discharge points.

Well V38E4 is situated downgradient of another pod which was both grouted and capped. Improvements noted here are less dramatic than in well T34. In well V38E4 significant reductions in concentrations of 34 to 53% were noted with a 4% non-significant reduction in mean SO_4 concentration. This suggests that grouting of this pod was somewhat less successful than the grouting of the pod at the T33 well location. However, the results are similar to the downgradient well observations for the grouted only situation, wells DD58 and EE54. It should be noted that prior to grouting, virtually all of the iron present in this well was in the form of Fe^{+2} and a small increase in Fe^{+3} concentrations post-grouting is causing the 169% increase.

The observed changes in SO_4 concentrations are not statistically significant for most of the wells monitored. In several of the wells, the SO4 concentrations exhibited little change or even increased. However, most of the wells show a decrease in the concentrations of the other mine drainage parameters. The lack of reduction in SO_4 concentrations is not consistent with the premise that AMD production has been reduced as evidenced by the reductions of the other mine drainage parameters.

Dissolution of sulfate salts is believed to be the reason for the lack of change in sulfate concentrations. Precipitates on the black shales are abundant on the site. Other studies indicate these are likely sulfate salts, most commonly Pickeringite, $MgAl_2(SO_4)_4 \cdot 22H_20$, and Halotrichite, $Fe^{+2}Al_2(SO_4)_4 \cdot 22H_20$ (Brady, 1996). Re-dissolving of these salts would explain the observations. Continued, long term monitoring will test this hypothesis.

There has been concern over the use of FBC ash because of the possibility of trace metals leaching from the ash once it is in place. Trace metals concentrations are not normally determined in routine mine drainage analysis. However, they were determined on this project and their concentrations are presented in Tables 1 and 3 for several of the wells. It should be noted that the concentrations of these metals produced on the site prior to FBC ash introduction are often orders of magnitude higher than the concentrations of the same metals which leached from the ash in the TCLP. Of perhaps greater significance is the reduction in concentrations that were observed as a result of using the ash. These data are presented in the tables. Although not included in the data, arsenic was also present in the mine drainage in concentrations greater than 2 mg/L. Reductions in arsenic concentrations similar to those of copper were common. In this case, the benefits of using the FBC ash far outweigh the concerns with respect to metals leaching from the ash.

It has be questioned whether post-grouting reductions in concentrations of the AMD parameters are a result AMD abatement or a continued neutralization by the grout supernatant. During the grouting operation it was common to note spikes in calcium concentrations along with significant rises in pH and dramatic reductions in concentrations of other mine drainage parameters due to neutralization reactions. If the supernatant was still present and was responsible for the changes, then we would expect to see statistically significant increases in calcium concentrations everywhere there was a significant decrease in the concentrations of the other parameters. This is not the case. The data in Table 2 show significant reductions in concentrations of the mine drainage parameters regardless of whether the calcium increased or decreased. Also, the noted spikes in pH usually disappeared within a few months after grouting indicating a flushing of the supernatant. Subsequent increases in calcium concentrations are thought to result from leaching of the grout.

The concentrations of ferric iron generated from the pods of pyritic material must also be considered. Garrels and Thompson (1960) have shown that pyrite is rapidly oxidized by ferric iron in the absence of oxygen and at low pH values:

$$FeS_2(s)+14Fe^{3+}+ 8H_2O \rightarrow 15Fe^{2+} + 2SO_4^{2-} + 16H^+ \quad \text{(eq. 1)}$$

According to the monitoring well data, Fe^{+3} concentrations from these pods commonly exceed 1000 mg/L with pH values close to 2.0. Once the water exits from the pod, such as the K23 location, it must migrate through 1500 feet of spoil before discharging from the site. The Fe^{+3} is available to rapidly oxidize pyrite located along its flow path to the discharge point. Reduction in Fe^{+3} formation should thus result in reduced pyrite oxidation. It can be seen from the data in Table 2 that reductions in Fe^{+3} concentrations closely paralleled those of total Fe.

Considering that the overburden on the site is acid producing, as demonstrated by the control wells such as L44, evaluation of the various grouting techniques used is difficult, since many of the grouted pods were within the flow path of the AMD as it migrated downdip towards the discharge points. The dramatic increase in concentrations in the control well during the summer of 1995 clouded the evaluation as well.

It is necessary to examine the changes in the water quality which discharges from the site to evaluate the overall effectiveness of the grouting effort.. When reviewing the data it is important to keep in mind that only 2 of the 37 acres (5% of the site) was directly affected by the grouting effort. Table 3 presents the pre- and post-grouting mean concentrations for several of the parameters tested, along with the percent reductions. A negative value indicates the concentrations increased rather than decreased.

There are five monitoring wells which intercept AMD discharge plumes, as identified by the EM mapping, Figure 1. All of these wells are located along down-dip portions of the mine site and reflect the quality of water leaving the site from the lower split of the Kittanning seam. However, all five wells indicated a wide range in water quality variations. During the spring the water quality is historically much better than during the summer and fall months. The lower split of the Lower Kittanning seam extends beyond the limits of the mining on the north and west sides. The winter snow pack is normally up to five feet thick. It appears that as the snow pack melts water infiltrates to this lower split of coal and migrates down dip, influencing the quality of the monitoring wells. It is believed that many of the water quality changes noted in Table 3 are not statistically significant because of the wide scatter in the data even though the percent change exceeds 30%.

Monitoring Well FF62 Mean Concentrations

Condition	Lab pH	TDS mg/L	SO_4 mg/L	Acid mg/L	Fe Tot mg/L	Fe^{+3} mg/L	Al mg/L	Mn mg/L	Cd ug/L	Cu ug/L	Cr ug/L	Ca mg/L	Temp deg C
Pre-Grout	2.3	7970	3477	4088	876	737	256	39.2	83.6	806	221	58.1	12.0
Post-Grout	2.5	5780	3110	2879	527	373	173	24.7	29.3	813	168	61.4	11.7
% Reduction	***-7%***	***27%***	11%	***30%***	***40%***	***49%***	***32%***	***37%***	***65%***	-1%	24%	-6%	3%

Monitoring Well S80D Mean Concentrations

Condition	Lab pH	TDS mg/L	SO_4 mg/L	Acid mg/L	Fe Tot mg/L	Fe^{+3} mg/L	Al mg/L	Mn mg/L	Cd ug/L	Cu ug/L	Cr ug/L	Ca mg/L	Temp deg C
Pre-Grout	2.4	9951	3500	5096	937	749	394	45.5	108.5	1542	394	66.4	12.7
Post-Grout	2.7	7222	3483	3230	530	254	282	32.4	24.9	771	232	58.7	12.7
% Reduction	***-9%***	***27%***	1%	***37%***	***43%***	***66%***	***28 %***	***28%***	***77%***	***50%***	***41%***	12%	0%

Monitoring Well W70 Mean Concentrations

Condition	Lab pH	TDS mg/L	SO_4 mg/L	Acid mg/L	Fe Tot mg/L	Fe^{+3} mg/L	Al mg/L	Mn mg/L	Cd ug/L	Cu ug/L	Cr ug/L	Ca mg/L	Temp deg C
Pre-Grout	2.6	9689	3695	4611	735	606	397	49.4	60.2	985	221	81.9	11.3
Post-Grout	3.0	4795	3327	2348	268	185	180	21.8	17.3	635	156	56.3	10.9
% Reduction	***-17%***	***51%***	10%	***49%***	***63%***	***69%***	***55 %***	***56%***	***71%***	34%	30%	***31%***	4%

Monitoring Well V36 Mean Concentrations

Condition	Lab pH	TDS mg/L	SO_4 mg/L	Acid mg/L	Fe Tot mg/L	Fe^{+3} mg/L	Al mg/L	Mn mg/L	Cd ug/L	Cu ug/L	Cr ug/L	Ca mg/L	Temp deg C
Pre-Grout	2.7	6777	3568	3624	570	396	316	63.7	57.3	526	104	83.1	11.4
Post-Grout	3.1	5330	3111	2351	380	263	212	33.8	21.6	689	149	71.9	10.9
% Reduction	-13%	21%	13%	35%	33%	34%	33 %	***46%***	***62%***	-30%	-44%	13%	4%

Monitoring Well U32D Mean Concentrations

Condition	Lab pH	TDS mg/L	SO_4 mg/L	Acid mg/L	Fe Tot mg/L	Fe^{+3} mg/L	Al mg/L	Mn mg/L	Cd ug/L	Cu ug/L	Cr ug/L	Ca mg/L	Temp deg C
Pre-Grout	2.5	5163	1868	2840	649	602	222	24.3	59.4	1185	245	51.2	10.0
Post-Grout	2.7	4838	2852	2235	356	317	130	13.7	16.1	638	142	185.6	10.8
% Reduction	-6%	6%	-53%	21%	45%	47%	41 %	43%	***73%***	***46%***	42%	-262%	-8 %

Table 3. Pre- and post-grouting mean concentrations of mine drainage constituents and percent reduction in mean concentration for wells located in discharge plumes. Statistically significant % reductions are indicated in bold italics. A negative value indicates a % increase.

Well S80D is located between the limit of mining on the south lobe and the toe-of-spoil discharge, D3. To date, the quality at D3 has remained fairly constant. However, S80D shows marked improvement which should eventually show up at the toe-of-spoil seep, D3. It should also be noted that, for all the wells in Table 3, the percent reductions in concentrations are similar to those observed within the spoil. In addition, changes in SO4 concentrations were generally minor and not statistically significant. This condition was also noted in the on-site wells

Chemical Interactions Between AMD and FBC Ash Grout

Monitoring well L25 provides a fairly clear picture of the chemical changes which occurred as a result of the ash injection into an acidic environment. L25 is located in the AMD plume downgradient of the pod of pyritic material at the K23 location. Pre-grouting water quality indicated the presence of severe AMD. Three injection wells were drilled within 50 feet of well L25 to accommodate a pit floor paving experiment. Only the bottom of the casing was open so that all of the injected grout would be directed to the pit floor. The two adjacent wells accepted a total of 90 yd^3 of grout and a third located 50 feet downgradient accepted 20 yd^3.

The following graphs illustrate the chemical changes which occurred when the grout interacted with the AMD. The first graph, Figure 2, depicts the changes which occurred to pH. Prior to the grouting, the mean pH was 2.3. Grouting occurred on day 1520. The pH increased immediately to 8.9 because of the highly alkaline supernatant. Over the next several months, the pH value gradually dropped. In 1994 the mean pH was 2.8 and in 1995 the mean pH was 2.4, close to pre-grouting levels.

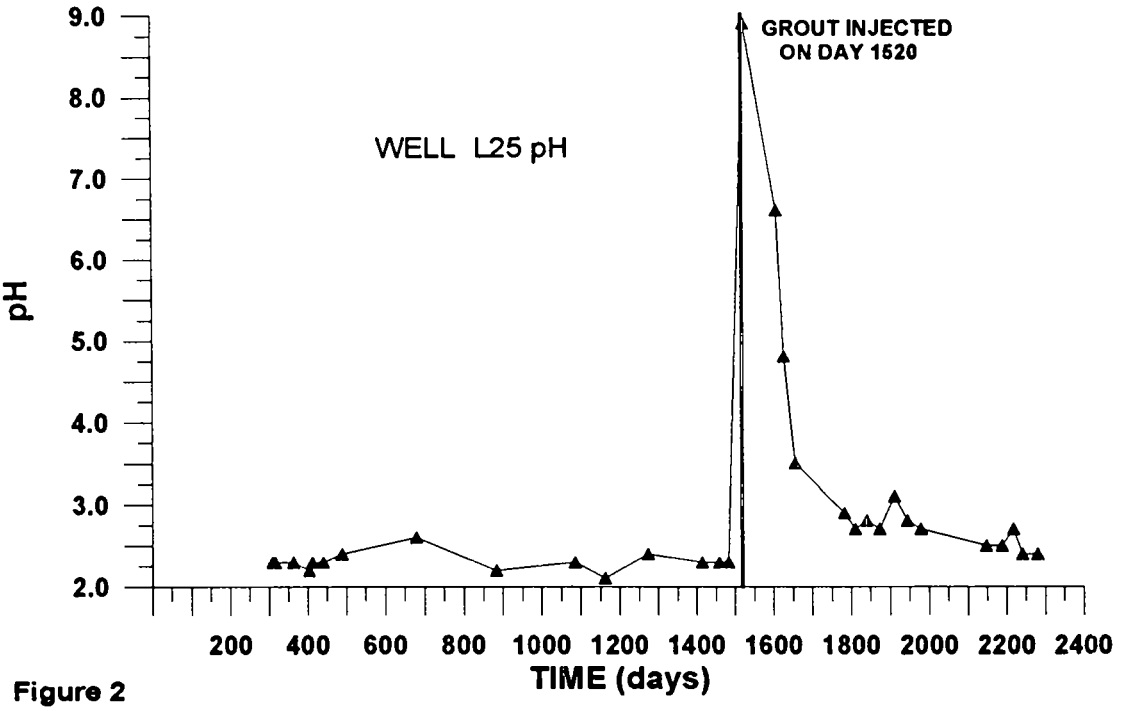

Figure 2

Figure 3 is a graph of SO_4 and acidity concentrations over time. These two variables likewise show a response to the grout supernatant. Sulfate concentrations averaged 6496 mg/L prior to grouting. After grouting, the concentrations were close to 1500 mg/L for the next four months. Since that time, the SO_4 concentrations have gradually risen to pre-grouting levels. Acidity also dropped dramatically as a result of the supernatant. In fact, the alkalinity determination 90 days after the grouting was 766 mg/L. During 1994 and 1995 the acidity concentrations have risen but remain at below pre-grouting levels.

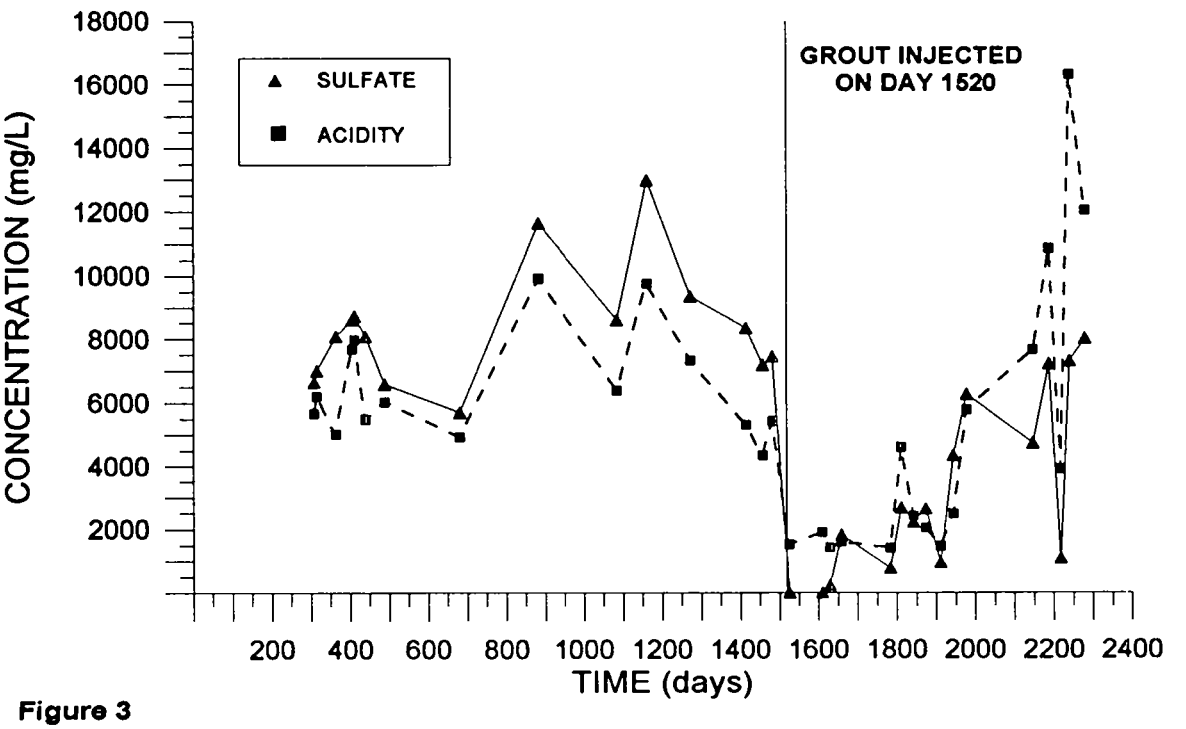

Figure 3

The third graph, Figure 4, illustrates the behavior of the trace metals: arsenic, cadmium, and barium. Throughout the history of sampling on this site, arsenic and cadmium behaved in similar fashion. Following the injection of the grout on day 1520, the concentrations of these two metals decreased dramatically and remain well below pre-grouting levels. Barium, on the other hand, increased in concentration from a background of less than 10 ug/L to 106 ug/L. It was common to note increases in barium shortly after grouting on other areas of the site. However, barium has since returned to pre-grouting levels.

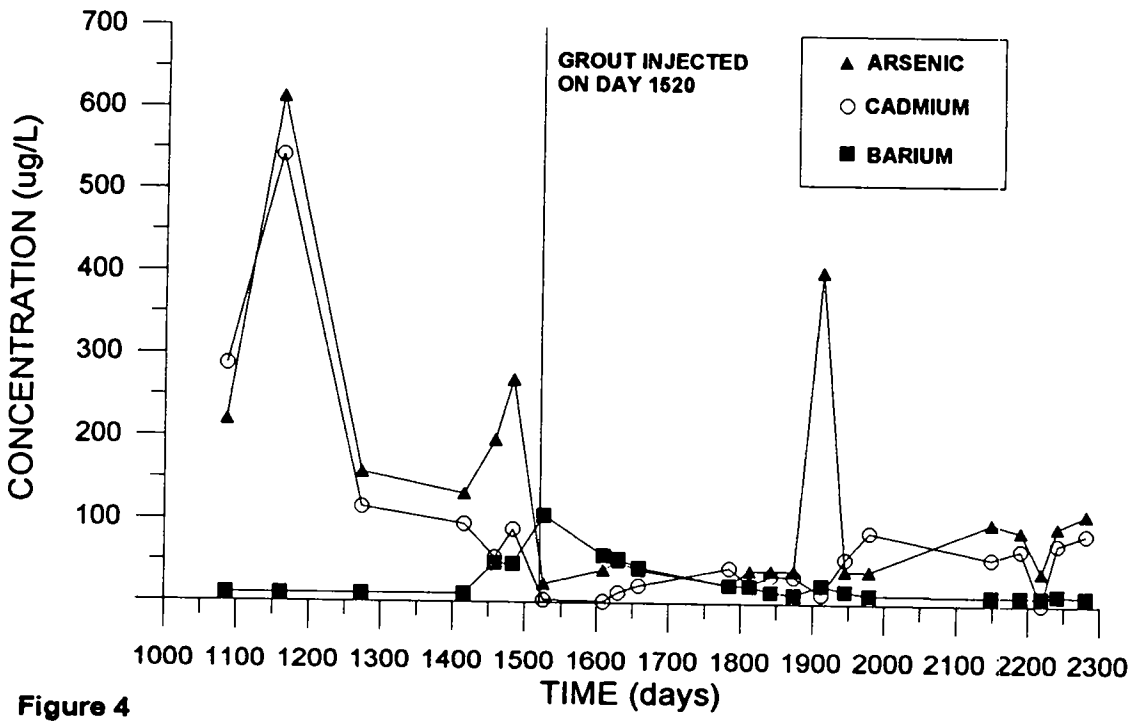

Figure 4

Calcium and aluminum concentrations are included on the final graph, Figure 5. Shortly after grouting, Al concentrations dropped to a low of 8 mg/L but have since returned to near pre-grouting levels. Calcium shows a rather different behavior. The mean Ca concentration prior to grouting was 72 mg/l for well L25. After injection, the concentration rose sharply to over 700 mg/L as a result of the supernatant. Since then, the mean concentration has averaged 350 mg/L, well above the pre-grouting value.

Based upon testing performed in the laboratory by the Penn State Materials Testing and Research Lab, there is an explanation for this. Within the grout, the calcium is normally present in three chemical phases . These include, in the order of decreasing solubilities, calcium hydroxide (logK=-5.05), calcium aluminum silicate hydrate (log K=-8.16 to -22.54), and calcium aluminum sulfate or ettringite (log K=-44.5). The high calcium concentrations are most likely a result of the dissolution of the calcium hydroxide. The calcium aluminum sulfate and the calcium aluminum silicate hydrate are likely to leach over time, but at a much slower rate than the calcium hydroxide. These reactions are similar to what would be expected if Portland cement had been injected instead of the fly ash (Silsbee, 1995).

A second possible source of the high calcium content might be the ash cap placed over and beyond the limits of the pile at K23. The ash which remained on the surface near the mixing bin was incorporated into this cap. Much of this was placed in the dry state and would be subject to leaching by infiltrating precipitation. The edge of this cap is only 30 feet away from the location of the L25 well.

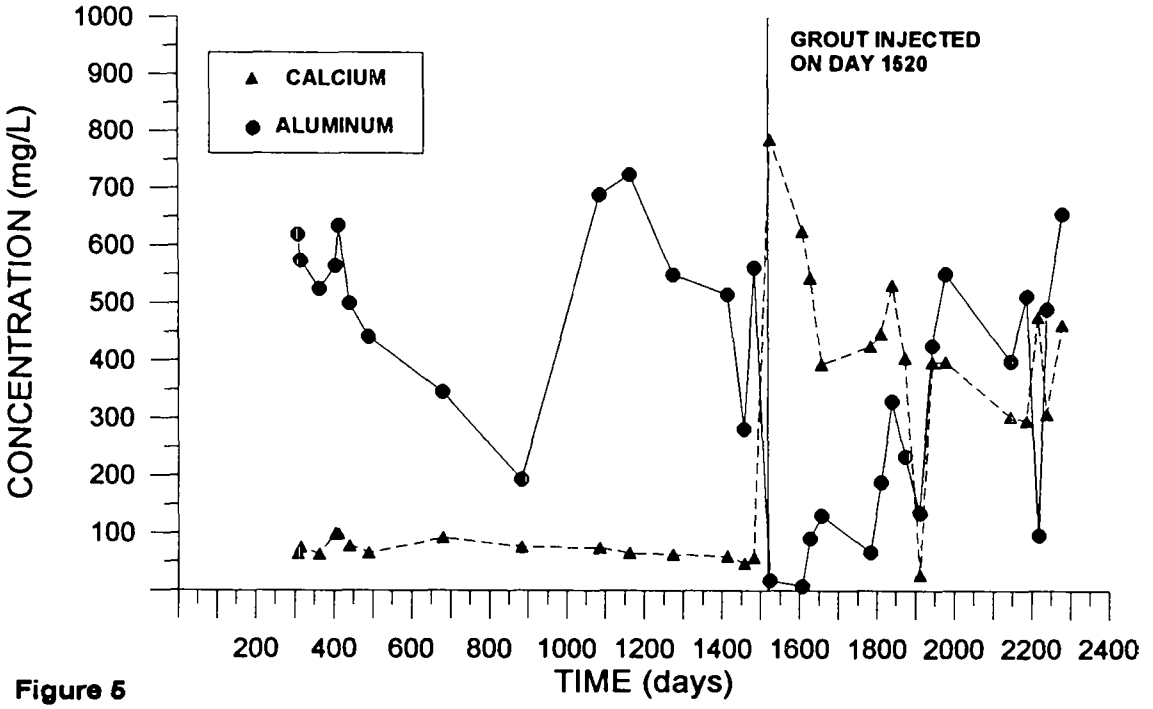

Figure 5

The purpose of the pit floor paving was to prevent the AMD from reacting with the clays to leach aluminum. The area where this was attempted exhibited high permeability. Unfortunately, this application occurred near the end of the project when the ash was in short supply. Consequently, several of the wells installed for this part of the project were not grouted.. Post grouting Al concentrations are 20% lower in L25 than they were pre-grouting. The data reflecting the presence of supernatent were excluded for making this determination. A downgradient monitoring well did not reflect this change, however. This is likely due to the limited aerial extent of the pit floor paving. Because of the limited extent of this application, insufficient data exists for the authors make recommendations on its use or non-use.

Conclusions

The pods of pyritic material were treated with FBC ash grout in three ways: 1) injection only, 2) capping only, and 3) both injection and grouting. Based on the water chemistry, the combination of injection grouting and capping produced the most favorable results, followed by injection only. Capping produced the least favorable results. The combined approach inhibits contact between water, oxygen, and pyrite by limiting infiltration as well as diverting lateral flow around the pods. Injection limits contact via lateral flow, but may not inhibit vertical infiltration. Capping is most applicable in situations where the pods are located high in the spoil and above the level of water table fluctuations within the backfill.

The inability to control final grout placement is a major drawback of the injection process. Pseudo karst conditions become established during backfilling operations (Hawkins, 1993). Because the gout is a viscous fluid, it will tend to flow into high permeability zones when pumped into spoil under pressure. If the permeability within the pod is low, the injected grout may flow away from the pod instead of filling the voids within the pod as intended or else the well will accept very little grout. When this happens, AMD abatement will be limited or will not occur at all.

The placement of the grout in the capping operation is controlled by the operator and is a direct approach. The ability to control infiltration zones is dependent upon the area of excavation and grouting. This approach is appropriate where the pyritic material cannot come into contact with water moving along the pit floor.

Use of the FBC ash grouting techniques on this site resulted in an overall improvement in water quality. Although percent reduction in mean concentrations vary, concentrations of the comon AMD parameters generally decreased by 30 to 40% and reduction of trace was usually higher. This is significant because the grout application occurred on only 5% of the site. The research effort was to demonstrate that FBC ash grout could effectively reduce the severe AMD production within the pods of pyritic material buried within the spoil. The noted water quality improvements indicated this goal has been met; however, the degree of improvement is somewhat less than what the researchers had hoped for. Any changes in water quality which resulted from the grouting are expected to be permanent because of the pozzolanic nature of the grout It was known that the entire site generated AMD and there was no pretense of eliminating all AMD production. Hopefully the reduction in pollutant loading discharging from the site will be sufficient to provide for stream recovery. Final stream evaluation remains to be completed.

Despite less than total success at AMD abatement, the authors view injection grouting as a viable AMD abatement technique worthy of application on sites which meet certain criteria. This technology is perhaps best indicated for those sites which would normally produce net alkaline drainage but improper placement of refuse or pit cleanings has resulted in an acidic discharge. In addition to reclaimed sites, the use of FBC ash is recommended on active surface mines and refuse disposal sites as a preventative measure. FBC ash grout can be used in a controlled approach on active sites. The ash grout can be applied directly to or mixed with refuse and pit cleanings to create monolithic structures capable of diverting water away from the pyritic materials.

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REMEDIATION OF ACID MINE DRAINAGE WITHIN STRIP MINE SPOIL BY SULFATE REDUCTION USING WASTE ORGANIC MATTER [1]

Jeralee Stalker, Arthur W. Rose and Louise H. Michaud[2]

Abstract: Many treatment options for AMD, like wetlands and anoxic limestone drains, are limited by acidity, metal loadings, flow rate or areal requirements so as to be inapplicable at many sites. *In-situ* bacterial sulfate reduction is proposed as a solution for certain settings. Requirements for successful *in-situ* bacterial sulfate reduction include dissolved sulfate, an organic substrate, permanent anaerobic conditions, a mixed culture of bacteria, appropriate nutrients, and a sufficient AMD contact time. These requirements can be provided within mine spoil by injection of waste organic matter into an extensive zone of saturated spoil.

Laboratory experiments on cheese whey, lactate, non-degraded sawdust, partially degraded sawdust, pulped newspaper and mushroom compost have all yielded sulfate reduction, increased alkalinity and iron sulfide precipitate in AMD with pH < 4.0. The addition of a small amount of dolomite to the organic matter creates alkaline microenvironments that facilitate the initiation of sulfate reduction. The rates of sulfate reduction using cellulose materials are slow but the rate for milk products is much more rapid.

A field test utilizing partially degraded sawdust is underway. A total of 11.3 tons of sawdust mixed with 5% dolomite, 5% sewage sludge and a mixed bacterial culture was successfully injected into 4 drill holes in mine spoil as 13% w/v suspension. The spoil had enough coarse porosity for injection into the saturated subsurface at about 300 L/min. Data on *in-situ* SO_4 reduction rates and water quality are being collected in preparation for a full remediation program at the site, which has an extensive zone of saturated spoil 10-20 m thick.

Additional Key Words: acid mine drainage, sulfate reduction, *in-situ* remediation, organic addition

Introduction

Although generation of acid mine drainage (AMD) at new surface coal mines has been largely eliminated by changes in federal and state regulations, many previously mined sites release severe AMD that contaminates streams in western Pennsylvania and other coal mining areas. "Passive" methods such as wetlands, compost wetlands, anoxic limestone drains (ALD) and alkalinity producing systems (APS) are beginning to provide methods for treating these AMD flows (Hedin et al., 1994). However, aerobic wetlands satisfactorily mitigate only AMD containing net alkalinity. Weltands also require a large area and become less effective in cold weather. ALD's can neutralize a maximum of only about 250 mg/l acidity, and require considerable area and length for large flows, as do compost wetlands and APS. This paper describes the injection of waste organic matter into strip mine spoil to promote bacterial sulfate reduction as an additional passive method. Figure 1 schematically illustrates a situation in which *in-situ* sulfate reduction may perform satisfactorily.

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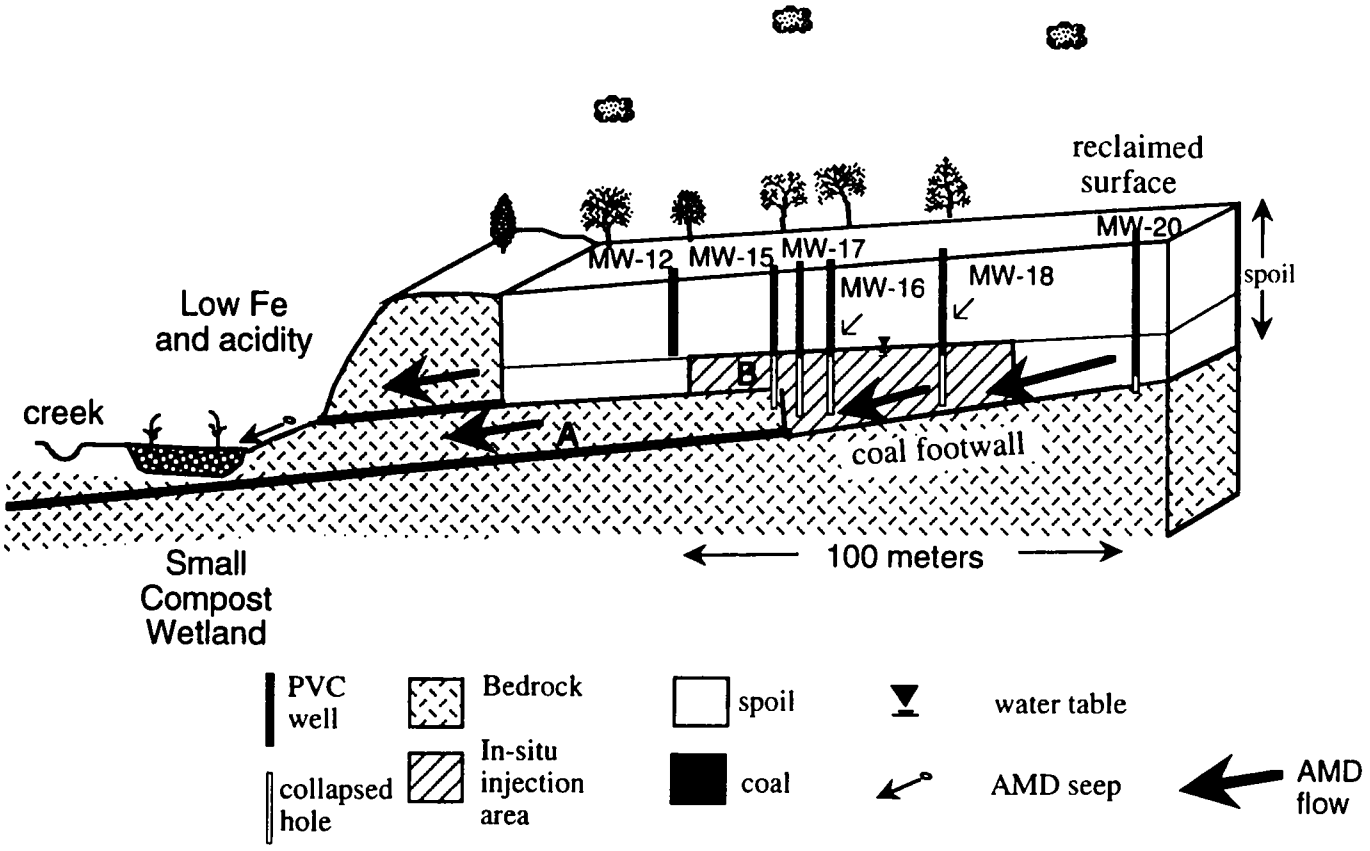

Figure 1. Schematic diagram of the area of in-situ sulfate reduction. Wells at the site are indicated. A hypothetical wetland is included in the diagram. Thick arrows indicate possible ground water flow paths. A) AMD may flow between the Mercer and Brookville coals in a sandstone unit and emerge as seeps along the crop line. B) This may be an area of little flow. Due to the shallow water table in this area, anaerobic conditions may not be maintained here.

To investigate the applicability of *in-situ* sulfate reduction, laboratory experiments were conducted on several waste organic materials to determine rates of sulfate reduction and to investigate limitations on the process, such as pH. Based on successful results of the laboratory experiments, a field test at a reclaimed strip mine was designed and implemented, and is being monitored.

Sulfate Reduction

Sulfate reduction is accomplished by several species of bacteria that obtain metabolic energy by reacting sulfate with simple organic compounds such as lactate, pyruvate, ethanol or acetate (Brock et al, 1994). They require an O_2-free environment because aerobic bacteria deplete the levels of these organic substrates to levels too low for utilization by sulfate reducing bacteria (SRB). In general, the SRB produce H_2S (or HS^-) and alkalinity (HCO_3^-). The H_2S reacts with dissolved Fe to precipitate FeS or a similar compound.

SRB cannot break down complex organic substrates; they rely on other bacteria, mainly fermenting bacteria, to provide substrates like acetate and lactate from larger organic molecules such as glucose or cellulose (Chapelle, 1993).

An example of the reaction accomplished by SRB on a lactate substrate at slightly acid pH (4 to 4.5) follows:

$$SO_4^{2-} + 2CH_3CHOHCOO^- + 4H^+ = H_2S + 2CH_3COOH + 2H_2CO_3 \quad (1)$$

Given the pK's for dissociation of lactic acid (3.8), acetic acid (4.75) and carbonic acid (6.5), the nature of the products depends on pH. At alkaline pH (pH>6.5), the products would be acetate (CH_3COO^-) and HCO_3^-, and no H^+ would be consumed but alkalinity would be produced. Some SRB are able to utilize acetate to reduce sulfate (Brock et al., 1994):

$$SO_4^{2-} + CH_3COO^- + 3H^+ = H_2S + 2H_2CO_3 \quad (2)$$

The total oxidation of lactate by sulfate is thus:

$$3SO_4^{2-} + 2CH_3CHOHCOO^- + 8H^+ = 3H_2S + 6H_2CO_3 \quad (3)$$

If H_2S does not degas or complex with other species, it can dissociate to HS^- and/or react with dissolved Fe^{2+} to precipitate FeS:

$$H_2S = H^+ + HS^- \quad (pK = 7.0 \text{ at } 25°C) \quad (4)$$

$$Fe^{2+} + HS^- = FeS + H^+ \quad (5)$$

An overall reaction generating FeS from SO_4 and lactate at pH 4.75 to 6.5 is:

$$3SO_4^{2-} + 2CH_3CHOHCOO^- + 3Fe^{2+} + 2H^+ = 3FeS + 6H_2CO_3 \quad (6)$$

Under certain conditions, sulfate reduction has the capability of producing pyrite from AMD, though the reaction is generally slow compared to FeS precipitation. In fact, bacterial sulfate reduction during sedimentary diagenesis was responsible for the formation most of the original pyrite in the rocks that produce AMD. Pyrite has been observed in several AMD localities undergoing sulfate reduction (Hedin et al., 1988).

In contrast to anoxic limestone drains, sulfate reduction is not limited in the amount of alkalinity that can be produced by SRB. The main limitations are the rate of the process, which may be relatively slow, especially if fermentative bacteria are necessary to break down cellulose or other complex organic compounds to simpler coumpounds metabolizable by

SRB. The pH may also be a limitation, since SRB prefer pH values above 5 (Zajic, 1969).

Materials and Methods-Laboratory Study

Prior to the *in-situ* test of sulfate reduction at the Pot Ridge mine, several organic substrates were tested in the laboratory. Initial substrate selection was based on a combination of characteristics including availability, cost, longevity and previous work indicating ability of SRB to degrade the substrate. Tuttle et al, (1969a, 1969b) showed that flow of AMD through a sawdust pile accomplished some remediation, and that cellulose could reduce SO_4 in lab experiments. Compost wetlands and APS accomplish some of their effect by sulfate reduction. Composted organic matter, hay, spent mushroom compost and other inexpensive organic materials have been used as organic substrates for sulfate reduction (Bechard et al., 1994, Bechard et al., 1995, Eger and Wagner, 1995, Dvorak et al., 1991, McIntire et al., 1990, Hammack and Edenborn, 1992, Borek et al., 1995). Waste whey from cheese production was initially tested as a substrate. Later substrates tested included fresh and degraded sawdust, newspaper, and spent mushroom compost.

In initial experiments with sawdust and pH 3.85 AMD, sulfate reduction had not started at day 217, but addition of 10% by weight dolomite led to fluctuating decreases in sulfate in 6 days and sustained decreases in sulfate in about 50 days. The dolomite particles apparently furnished alkaline microenvironments in which the bacteria could thrive. Therefore, dolomite particles (100 mesh) or alkaline fly ash was added to later experiments.

Intial experiments also indicated that low redox conditions are absolutely necessary to stimulate sulfate reduction. Several types of plastic tubing were used in systems for circulating AMD through substrate-filled columns, but all were found to be permeable to O_2 , resulting in Fe-oxide precipitation and slow to negligible sulfate reduction. The most useful data was obtained in glass systems completely closed to air except during brief addition of fresh AMD.

Bacteria from waste water at the Penn State Sewage Treatment Plant were initially cultured with cheese whey and an $FeSO_4$ solution. After sulfate reduction was observed in this medium, the culture was divided into several closed containers and fed lactate and $FeSO_4$ periodically. Later experiments were inoculated with 10 ml of this culture.

The lactate and whey experiment included 1 liter of synthetic AMD ($FeSO_4 \cdot 7H_2O$) and whey initially, then reagent grade calcium lactate. This experiment was circulated through a flask filled with glass beads (Figure 2A) by bubbling a constant stream of ultra high purity (UHP) N_2 through tubing connecting the bottom and top of the column. Samples were collected weekly from one of the top openings under a stream of UHP N_2 using a pipet.

The experiments with solid substrates were run in glass ion chromatography columns (2.2 cm diam x 50 cm length) (Figure 2B) with 125 mL of AMD from the Pot Ridge site and variable amounts of substrate (Table 1). Sawdust, spent mushroom compost and newspaper were tested. Table 1 indicates the organic substrates tested, the alkaline material added to the experiment, the volume of organic substrate added and the initial sulfate concentrations. Weekly, 20 ml was drained from the bottom and analyzed, and 20 ml of fresh AMD was added to the top of the columns.

The pH, Eh, and temperature were measured *in-situ* for the lactate experiment and immediately after the effluent was drained from the columns for the solid substrates. Alkalinity

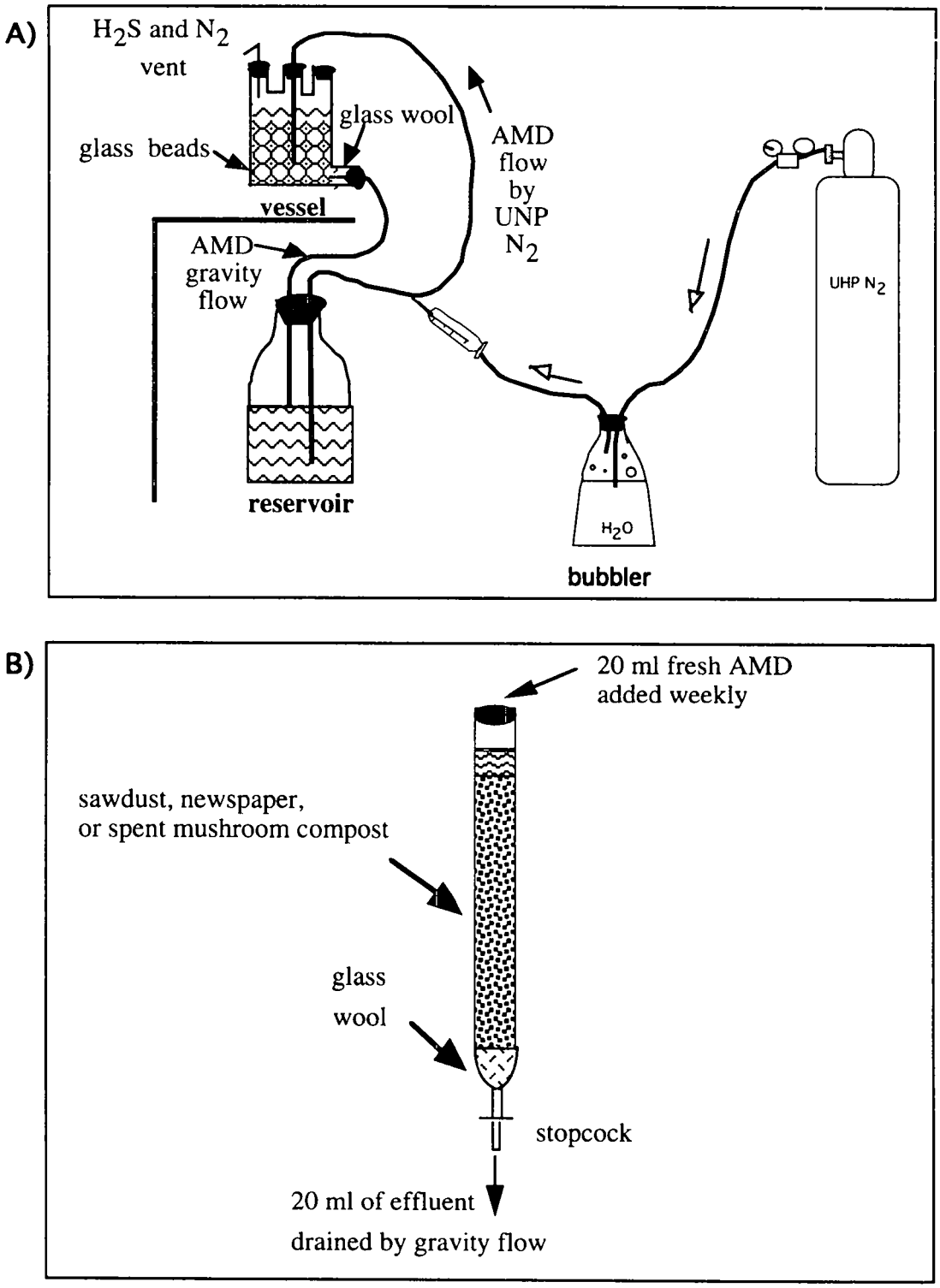

Figure 2. A) Reaction vessel and reservoir . B) Column design. Weekly, 20 ml of effluent was drained from the stopcock and fresh AMD was replaced at the top.

Table 1. Organic substrates, alkaline and AMD sources for each experiment.

Organic substrate	Alkaline source	Amount of Substrate	SO_4 mg/L	AMD source and amount
Fresh sawdust	FBC fly ash (10%)	3.84 cm^3	641 mg/L	150 ml AMD
Degraded sawdust	FBC fly ash (10%)	24.05 cm^3	641 mg/L	150 ml AMD
Spent mushroom compost	limestone	22.65 cm^3	641 mg/L	125 ml AMD
Newspaper	dolomite	5.37 cm^3	641 mg/L	125 ml AMD
Lactate	------	2400 mg/L	1585 mg/L	1L $FeSO_4$

was determined immediately after sampling and was followed by hot acidity. Methods are summarized in Table 2. Samples were diluted for ICP analysis using a ratio of 6 ml sample: 1 ml concentrated HCl: 23 ml distilled deionized water. All samples were filtered through a .45 micron filter directly after dilution and prior to chemical analysis by ICP.

Results of Laboratory Studies

In general, sulfate reduction was observed for all substrates after varying lag periods. For example, Figure 3 illustrates the behavior of SO_4, Fe and pH in the experiment using lactate. Initially, the low pH solution and high redox conditions inhibit sulfate-reducing bacteria. In a system closed to the atmosphere, O_2 is utilized rapidly by aerobic organisms, lowering the Eh. Fermentation then breaks down the complex organics matter into simpler substrates. Sulfate reducing bacteria utilize the simple organics via equation 6 or a similar reaction involving simple organic matter and sulfate. The bacteria thus create alkaline microenvironments within low pH AMD. Figure 4 summarizes results in the form of percent gains and losses for SO_4 and Fe for each substrate normalized for changing influent concentrations and different

Table 2. Analytical Methods

Method	Procedure
Alkalinity	Method 410 (Fishman and Friedman, 1989)
Acidity	Method 305.1 of E.P.A. (1979) using boiling and H_2O_2 to oxidize Fe
pH	Combination Ag/AgCl electrode standardized with pH 4 and 7 buffers connected to an Orion 205A or a Fisher Scientific Accumet pH Meter
Eh	platinum electrode with an Ag/AgCl reference electrode
Mn, Al, Fe,S Ca, Mg, Na, K	Plasma Spectrometry (ICP)

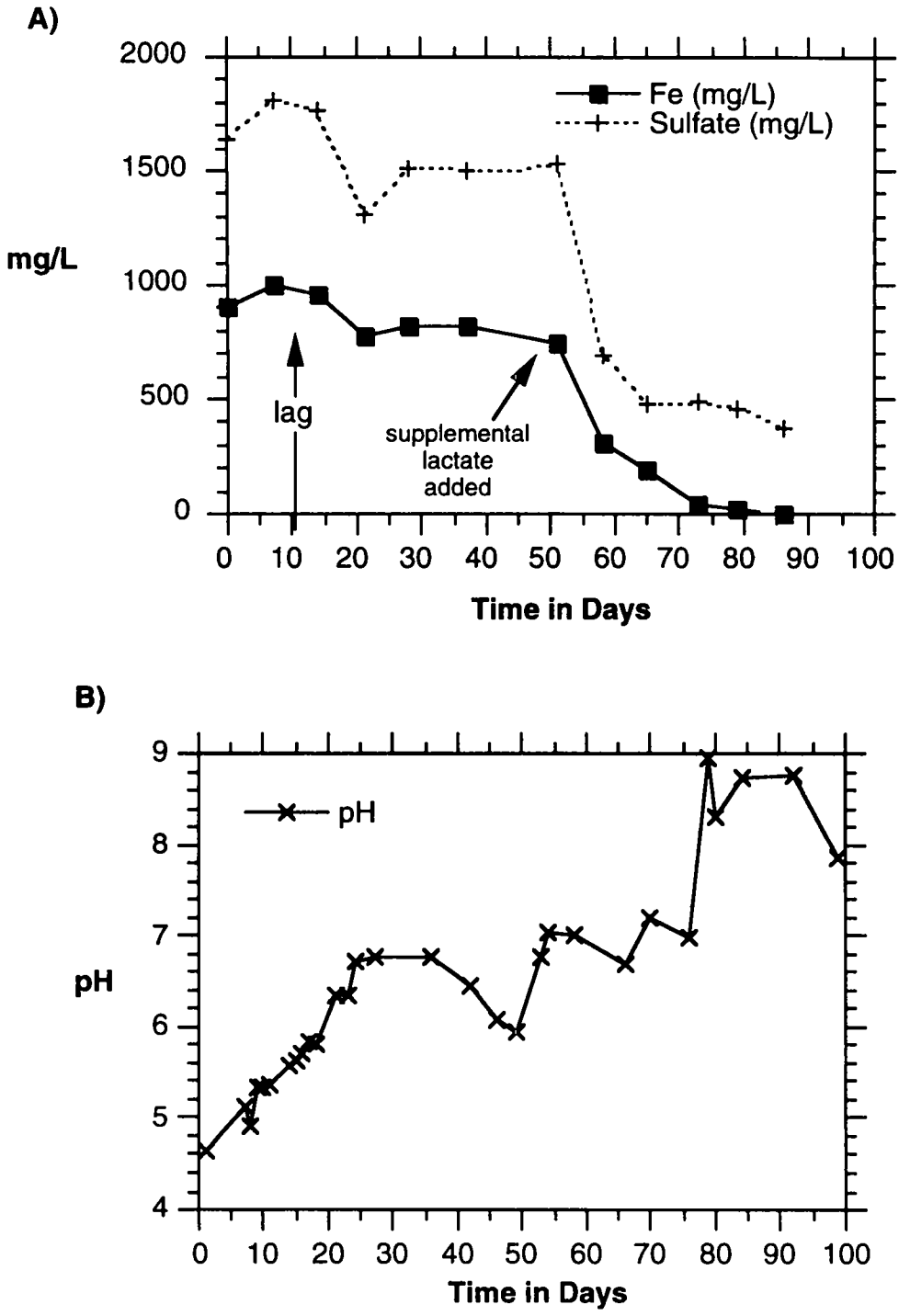

Figure 3. Plot of sulfate, iron (A) and pH trends (B) over time for the lactate experiment as an example of the raw data. The lag time before sulfate reduction is between day 0 and day 15. Sulfate decreases rapidly after this period. The ceasing of sulfate reduction from day 30 to day 55 represents a period of negligible organic matter concentrations. Sulfate decreases again when lactate is added on day 46.

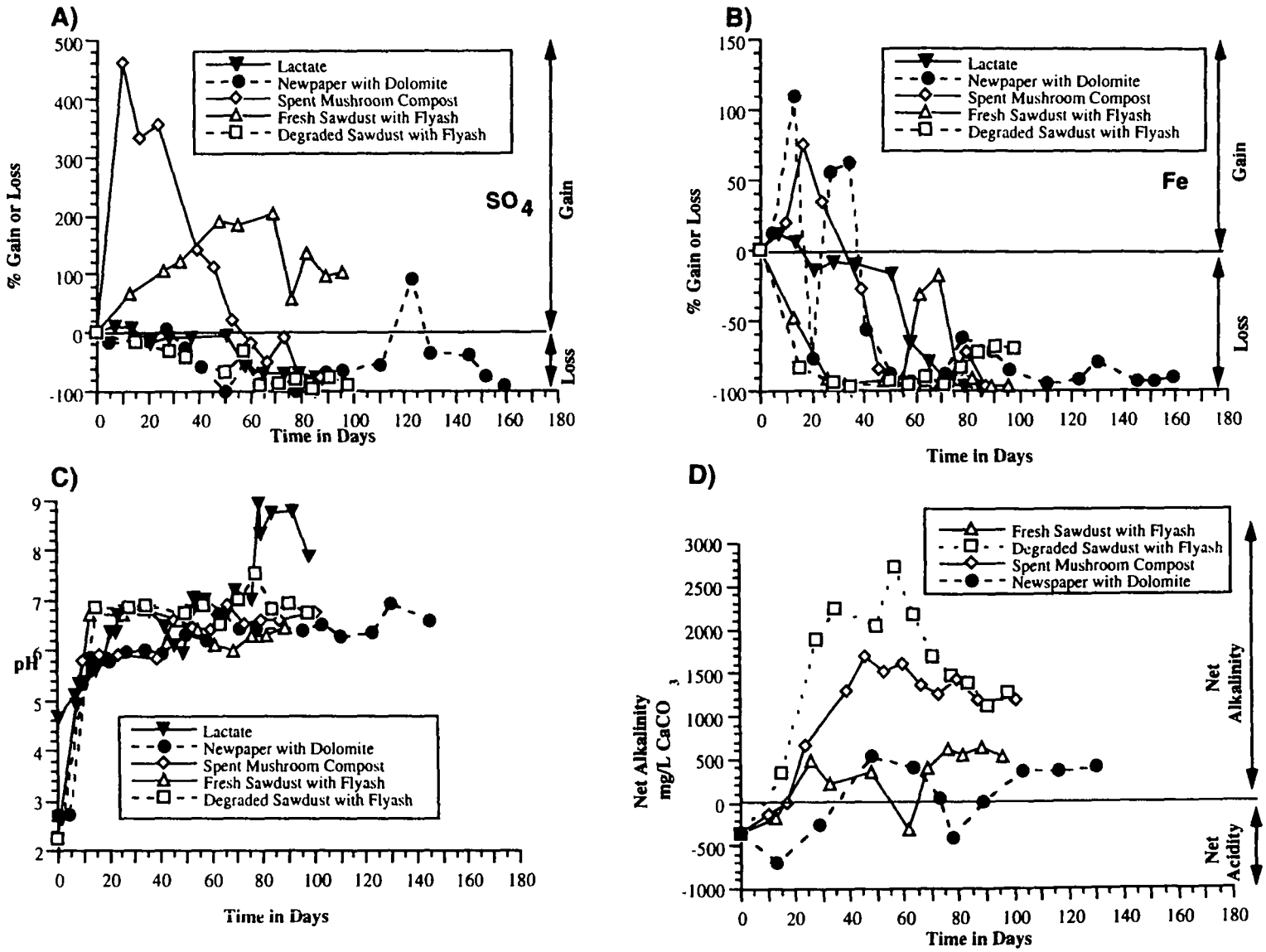

Figure 4. Experimental data for various organic substrates tested in the lab. A) and B) Gains and losses for sulfate and iron are expressed in % relative to the influent concentrations; C) pH over time; D) Net alkalinity over time.

substrate volumes. The influent concentrations of Fe and SO_4 for the lactate experiment were 870 mg/L and 1585 mg/L respectively. For the other experiments which used solid substrates, the influent concentrations of Fe and SO_4 were 38 mg/L and 641 mg/l respectively.

Several experiments showed increases in sulfate and other solutes during an early leaching period. An increase in SO_4 was observed in the mushroom compost due to the dissolution of gypsum and in the fresh sawdust amended with fly ash due to the leaching of sulfate from the fly ash. No excess sulfate was observed for the degraded sawdust with fly ash; however, cation concentrations for Ca, Mg, K and Na increased in both substrates suggesting that these constituents were being leached from fly ash in both degraded and fresh sawdust substrates. The cation leaching period lasted for the entire length of the experiment for both sawdust substrates. Although some leaching of Fe is indicated in Figure 4 for the newspaper with dolomite and the spent mushroom compost, nearly 100% of the Fe was removed in all experiments to once sulfate reduction began. Erratic patterns for some substrates may be due to disturbance of the anaerobic environment.

Average sulfate reduction rates were calculated for each of the column experiments using the decrease in sulfate concentrations over time per gram of substrate (Table 3). Rates were calculated for the lactate experiment using the decrease in sulfate concentrations over time per liter of AMD. Gypsum dissolution in the sawdust experiments that contained fly ash yielded excess SO_4 in these columns. In order to obtain sulfate reduction rates that reflected this additional SO_4 concentration, the analytical data was corrected using the molar ratio of $Ca:SO_4$ from control experiments (Stalker, 1996). The sulfate reduction rates for fresh and degraded sawdust were calculated using corrected SO_4 values. The sulfate reduction rates obtained for the degraded sawdust with fly ash experiment are probably underestimated, as both the sulfate leached from fly ash and the sulfate in influent AMD are reduced. Rates of sulfate reduction for the mushroom compost and newspaper with dolomite experiments were determined during the later portion of the experiments, after leaching. The alkalinity and pH trends over time indicate that all experiments went from acidic to near neutral pH levels with a concomitant rise in alkalinity (Figure 4). Alkalinity was not measured during the course of the lactate experiment, but a net alkalinity of 260 mg/l $CaCO_3$ was measured at the culmination of the experiment.

All experiments produced effluents with high alkalinity. However, since some alkalinity could be furnished by carbonate dissolution or consumed by precipitation of Fe and Al hydroxides, the source of alkalinity deserves consideration. Based on the increased Ca and decreased SO_4 in solution, dissolution of added dolomite, and alkaline components of fly ash and mushroom compost is estimated to contribute 70-

Table 3. Calculated sulfate reduction rates for the experimental organic substrates.

Substrate	Average SO_4 Reduction Rate
Lactate	197 mmol/L/day
Newspaper with dolomite	69 nmol/g/day
Spent Mushroom Compost	60 nmol/g/day
Fresh sawdust with fly ash	20 nmol/g/day
Degraded sawdust with fly ash	148 nmol/g/day

90% of the alkalinity, especially during the early stages of the experiment. However, some alkalinity was attributable to sulfate reduction in each experiment.

Site of the Field Test

The Pot Ridge mine located near Dunlo, PA, U.S.A. was the focus of the field study. The mine is a 198.3 hectare strip mine that was reclaimed approximately 10 years ago. Forty-four highly acidic AMD seeps emerge near the cropline of the Brookville coal and flow into nearby Paint Creek.

In the central part of the Pot Ridge mine, ground water is ponded against an internal highwall bounding an up dip zone where the deeper Mercer coal as well as the Brookville and Clarion coals were mined (Figure 1), and a down dip mine where only the upper two coals were mined (Figure 1). A saturated zone up to 13 m thick is ponded in the up dip mine over an area of about 400 m x 300 m. Rainfall infiltrating into the spoil in this up dip area becomes acid in the unsaturated zone, migrates downward and down dip in the spoil to the ponded zone, and then slowly leaks out through the crop barrier and a barrier between the mines to emerge as a major group of seeps.

This saturated zone appears to satisfy the requirements for *in-situ* sulfate reduction if organic material is introduced. The thick zone of ponded groundwater should lead to a thick anoxic layer overlain by a thin aerobic layer just beneath the water table in which dissolved O_2 is consumed by aerobic bacteria. Flow through the ponded zone is slow, estimated at .4 m/day giving time for sulfate reduction. Although the water table fluctuates by 1 to 2 m over the year, the ponded zone appears permanent, which is essential to prevent re-oxidation of FeS and regeneration of AMD.

The results from the laboratory experiments were used to implement a field test for *in-situ* sulfate reduction. Nearly 11.3 tons of sawdust mixed with 5 % by volume dolomite and 5 % by volume sewage sludge (to supply nutrients) were injected as a slurry with 13% solids into the saturated spoil at the Pot Ridge mine. A total of about 20 liters of an SRB culture was mixed into the slurry as it was injected. The slurry was injected through the drillstem into 4 drill holes spanning about 33 m (Figure 1). Previous efforts to complete drill holes using casing with coarse perforations were unsuccessful due to caving before the casing could be inserted. The slurry was mixed in a hydroseeder and pumped into the drill hole with a grouting machine or the hydroseeder pump at rates of about 300 L/min (75 gal/min). Little or no back pressure was encountered; the slurry seemed to flow into the hole essentially by gravity, indicating high porosity and large pores.

The sawdust was derived from a pile left from a sawmill near Portage, PA and was considerably degraded and mixed with soil (79.8% loss on ingnition, LOI) in comparison to fresh sawdust that was 99.7% LOI. Because of appreciable stone, bark and other coarse pieces, it was sieved to less than 1 inch to remove these materials.

The extent of spoil containing the sawdust slurry is not clear, but little sawdust could be noted in holes 10 m from the injection holes. If the spoil has 30% porosity, calculations indicate that 2.7% of the porosity is filled with sawdust.

Preliminary results after about 7 months indicate an inital rise in pH and alkalinity, and appreciable decrease in SO_4 in the injection area compared to an up dip well (Figure 1). Table 4 presents *in-situ* data for a monitoring well (MW-17) in the injection area and an up dip well (MW-20).

Table 4. Results for an *in-situ* injection well (MW-17) and an up dip monitoring well (MW-20). MW-17 was dry between days 51-107 due an extremely dry summer. Day 0 is July 26, 1995.

Injection well (MW-17)

Days	m SWL	pH	mS/cm Cond.	mg/L $CaCO_3$ Acid.	Alk.	mg/L SO4	Fe	Mn	Al	Ca	Mg	TOC
37*	13.96	7.40	2560	0	686	1010	155	65	.1	352	154	251
51*	14.42	6.60	1090	0	407	1019	177	54	9.3	295	175	155
107*	13.87	5.66	1350	408	180	595	239	83	.8	191	147	83
145		4.60	1840	440	8.0	1534	86	90	17	171	197	2.1
189		3.30	2480	440	0	784	37	93	24	164	199	3.5
210**		3.70	1275	--	--	--	--	--	--	--	--	--

Up dip well (MW-20)

Days	m SWL	pH	mS/cm Cond.	mg/L $CaCO_3$ Acid.	Alk.	mg/L SO4	Fe	Mn	Al	Ca	Mg	TOC
37*	18.50	3.90	6600	1540	0	4085	381	131	67	329	320	5.0
51*	18.93	4.00	6450	1660	0	3651	446	124	94	212	338	3.7
62		4.20	6690	1230	18	3088	404	132	84	258	337	4.9
76*	19.45	4.40	6500	1364	28	3082	477	128	91	261	338	6.4
90		5.00	5280	680	22	2400	247	114	14	306	252	3.2
107*	18.47	4.18	2950	1412	0	1275	467	126	100	236	202	NA
111		4.90	5960	962	18	3238	319	132	23	338	326	3.2
145		4.80	6370	1080	20	2946	284	126	13	363	323	3.0
189		3.20	7440	1940	0	1772	467	159	94	256	456	3.8
210**		4.60	3400	--	--	--	--	--	--	--	--	--

* Denotes PSU sampling dates. All other dates sampled by Mahaffey Labs, Grampian, PA.

** Monitoring is continuing and the samples from day 210 are being analyzed.

If the difference in SO_4 concentrations between the up dip well and the injection well was due to SO_4 reduction and the precipitation of FeS, then equal molar decreases in Fe would be expected. Redox potential was low enough (140 mV) within the injection area to support sulfate reduction, but persistent high concentrations of Fe suggest that sulfate reduction may not have been responsible for the large difference in SO_4 concentrations between MW-20 to MW-17. A decrease in TOC over time in MW-17 may suggest bacterial activity or dilution in the injection area. Increases in alkalinity and pH suggest that some acidity-consuming reactions are occurring. In addition to sulfate reduction the dissolution of dolomite is probably occurring. The possibility also exists that the organic matter in the sewage sludge adsorbed Fe, other metals and/or sulfur compounds. This could explain why the ratio of decreases in Fe to SO_4 are not concomitant.

A significant problem may be that MW-17 had only about 0.7 m of water at most sampling dates, whereas most sawdust was injected from 1 to 8 m deeper. Sulfate reduction may have been occurring beneath the completed portion of the well where anaerobic conditions are inferred to be prevalent. Figure 1 indicates that the geometry of the injection area may greatly influence the test results.

Another possible explanation for lack of clear evidence for SO_4 reduction is that deep water in MW-20 does not flow to the other monitoring wells. The water that is sampled from MW-20 may leak through sandstone units associated with the Brookville and Mercer coals and emerge as seeps to the northwest without mixing with overlying water. Because of this MW-20 samples may not reflect the up dip chemistry of the monitoring wells.

A third possibility is that there is little flow in the injection area, especially at MW-15 where the ponded area thins over the Brookville coal pit floor.

Discussion

Although the most rapid rate of sulfate reduction was obtained with whey (and lactate), this material has the disadvantage of being liquid, so it would presumable flow out of the spoil with the groundwater and would have to be replaced continuously. The possibility of organic outflow has been raised as a concern, though if appropriate concentrations were added the effluent concentration should be negligible. However, because of these potential problems, this option was not pursued, though it has potential. Discussions with milk processors indicate that whey is difficult to dispose of, and considerable quantities might be obtained.

Addition of alkalinity can stimulate the initiation of sulfate reduction. However, leaching from the alkaline material may initially increase concentrations in the effluent. For example, some fly ash and mushroom compost contains readily soluble $CaSO_4$. Over time the leaching will slow and eventually concentrations will decrease to less than or equal to influent concentrations. Initial alkalinity and increased pH in both the lab and field experiments is most likely the result of carbonate dissolution. Subsequent alkalinity generation is mostly the result of sulfate reduction.

Clearly, fresh sawdust did not support sulfate reduction as well as the degraded sawdust. This was also evidenced by Tuttle et al. (1969b) where partially degraded wood dust yielded more consistent sulfate reduction rates over a longer period of time than fresh sawdust in flask experiments. Here, the degraded sawdust with fly ash provided enough carbon to reduce not only all of the generated SO_4 from the fly ash, but also some of the influent SO_4 as well. This observation suggests that the degree of degradation of sawdust can affect sulfate reduction rates. Sulfate reduction using fresh sawdust may be limited by lignin, which protects plant cells from degradation. Also, the degraded sawdust may have been significantly broken down by fermentation of cellulose into simpler organics that SRB can use, but not so degraded that all of the reactive carbon is gone. Newspaper, mushroom compost and lactate all support sulfate reduction at similar rates.

Based on the average rate of SO_4 reduction obtained in the lab experiments, the reduction of sufficient SO_4 to create alkalinity adequate to neutralize the very high acidity of 1500 mg/L $CaCO_3$ found at the Pot Ridge mine would require a contact time of 500 days assuming 30% spoil porosity, 2.7% of porosity filled with sawdust and 2 moles of alkalinity for each SO_4 reduced. This very long contact time is not practical at the Pot Ridge site. However, several possible means of improving this performance can be suggested. Injection of higher sawdust concentrations, to perhaps 10% of the porosity, should proportionately increase the reduction effect. Also, the possibility of using milk processing wastes, which react more rapidly, exists.

An important question remaining is whether cellulose-type substrates will continue to degrade and react at rates similar to those measured. Eger and Wagner (1995) found that sulfate reduction markedly decreased in the third year of experiments with compost and sawdust and ceased in the fourth year, despite considerable carbon still available. Middelburg (1989) summarizes research on sulfate reduction in marine sediments indicating that the rate of organic carbon

decomposition in recently deposited marine sediments decreases with time according to the equation:

$$\log k = -0.95 \log t - 0.81 \qquad (7)$$

where k is the rate of carbon decomposition by sulfate reduction (yr^{-1}), and t is time (yr). If this relation applies, then the rate of organic-sulfate reaction is predicted to decrease by a factor of about 9 between year 1 and year 10. To the extent that the organic matter in marine sediments is land-derived (which some undoubtedly is), then this rate of decrease might apply to the AMD system. This first order relationship follows the multi-G model which asserts that organic matter consists of many groups of compounds with different degradation rates (Westrich and Berner, 1984; Jorgensen, 1979). Marine organic matter tends to be more reactive than land plant wastes and the pH in the marine environment is higher than most AMD environments. Fe occurs as solid Fe-oxides rather than in solution in AMD, so the rate may decrease at a slower rate for the conditions applicable in this study. Tarutis and Unz (1994) showed similar trends for organic matter in wetlands exposed to AMD. Also, the rate of AMD generation probably drops off with time, so the required rate may be smaller.

Based on these data, the reaction rates we have derived should probably be considered a maximum rate, with a progressive decrease over a period of years. Probably only a field experiment over a long time period will be able to determine the long-term behavior of sawdust or similar materials exposed to AMD.

The sulfate reduction rates obtained in this study for cellulose and related materials are probably not the optimum that can be attained. The breakdown of the wood material by fermenting bacteria is probably the rate limiting step. Further experiments are needed to find fermenting bacteria that can accomplish this reaction more rapidly at acid pH values.

Conclusions

Bacterial sulfate reduction is capable of removing iron from AMD solutions and furnishing alkalinity to neutralize the acidity. Considerable sulfate is also removed. The process has no inherent maximum concentration that can be removed, in contrast to anoxic limestone drains. A variety of organic substrates can be used, including several forms of organic waste products such as milk processing wastes, sawdust, compost, and paper. If a sizable zone of anaerobic water-saturated spoil exists or can be created at a reclaimed surface mine, then AMD generated in the overlying unsaturated spoil can be partly or completely remediated within the spoil, thus obviating the need for surface treatment.

Experiments indicate that bacterial sulfate reduction can be initiated even at pH less than 3, though the rate is slow until about pH 5 is reached. Initial results from a field test show that slurries of organic particles can be injected into strip mine spoil in large quantities. Initially, there was a large rise in alkalinity and pH in the injection area, however these parameters have leveled off to pre-treatment values. It is possible that the monitoring wells are not deep enough to penetrate the anaerobic zone. Deeper wells are being drilled and sampled to ascertain the effects of sulfate reduction on AMD at depth in the spoil.

Acknowledgments

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IMPROVING GROUND WATER QUALITY IN THE BACKFILL WITH ALKALINE ADDITIONS

by

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Abstract. Skyline Coal Company, a wholly owned subsidiary of Cyprus Amax Coal Company, initiated surface mining of the Sewanee seam of coal in Sequatchie County, Tennessee, in September, 1987, pursuant to the terms of a duly approved surface mining permit. The mine was named after an adjacent creek, Glady Fork. The engineering and geological studies conducted in the preparation of the surface mining permit application indicated that the mine could be operated with a minimum of concern for the generation of adverse impacts to the hydrological regime outside of the permit area. On July 5, 1990, however, the mine was cited for polluting Glady Fork with iron laden seeps. Shortly thereafter, comprehensive hydrogeological, geochemical, and operations management investigations were launched. Furthermore, laboratory testing procedures were scrutinized and modified to address the "siderite masking" of overburden data generated from standard acid-base accounting techniques. These investigations culminated in the preparation of a detailed Toxic Materials Handling Plan (TMHP) which incorporated "state of the art" ground water management plans and water quality assurance programs. In light of the lithologic distribution of the problematic strata and the reliance on both blast casting and dragline equipment for overburden movement, the TMHP placed a heavy emphasis on the timeliness of certain activities, the collection of detailed pre-mining information, and the application of heavy doses of alkalinity in the form of crushed limestone. The permit area has been mined out and is currently in the final reclamation phases of pit closure. Data collection from various ground water sources support the validity of the concepts and practices undergirding the TMHP. Significant alkaline recharge of the backfill ground water resource has been realized. More importantly, the alkaline additions of the TMHP have yielded substantial reductions in both iron (Fe) and manganese (Mn) concentrations in the post-mine ground water regime.[1][2][3]

Introduction

Activities aimed at the prevention or abatement of poor quality mine drainage must be based upon a well designed, fully implemented Toxic Materials Handling Plan (TMHP). The plan should be site specific in nature. Its overall effectiveness in meeting desired abatement objectives will be contingent upon the prudent factoring of key hydrogeologic, geochemical, and operational considerations into its design and field application. In this case, alkaline additions through imported limestone were a key component to the success of a TMHP developed specifically for the Glady Fork Mine. This paper will document the merits of adding limestone amendments to the backfill in coordination with surface coal mining and reclamation activities. Emphasis is placed on the benefits

[1]Paper presented at the 1996 Annual Meeting of the American Society for Surface Mining and Reclamation, Knoxville, May, 1996.

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[3]Publication in this proceedings does not prevent authors from publishing their manuscripts, whole or in part, in other publication outlets.

derived from adding the alkaline materials along identified hydrologic flow-paths in the backfill.

Statement Of Purpose

The purpose of this paper is to share the experiences of Skyline Coal Company with the coal mining industry. In particular, the writers wish to address the following points of significance:

- The methodologies involved with limestone incorporation to the backfill in synchronization with the surface mining and reclamation process require a detailed knowledge of the operation of the mine.

- Limestone additions, with respect to both quantity and placement in the backfill must be based on overburden geochemical data. These data must be routinely gathered in advance of mining.

- Time spent in identifying the post mining ground water regime and direction of flow is essential in developing concepts and defining the needed "building - blocks" to achieve overall effectiveness of the TMHP.

- The benefits of alkaline additions can be directly measured or observed by monitoring the ground water quality of the backfill.

Background

Skyline Coal Company is a small surface operation (<500,000 tons/year) located in Sequatchie County, approximately 15 miles northwest of Dunlap, Tennessee. The subject mine is the Glady Fork Mine. Mining was initiated at the Glady Fork site in September, 1987, in the lower third of the mining permit by use of endloaders and trucks. A box cut was prepared using that equipment. Once the box cut was in place, a walking dragline was placed into the sequence, and the front end loaders and overburden trucks were subsequently phased out. Cast-blasting was later introduced to the operations and subsequently the remainder of the permitted area has been developed with a combination of both dragline and cast-blasting as the primary mode for overburden movement.

The coal-bearing sequence developed at Glady Fork are Pennsylvanian Age rocks of the Crab Orchard Group. *Figure 1* shows the overburden strata to consist of the Newton Sandstone and Whitwell Shale Formations. The coal seam being mined is the Sewanee Coal Member of the Whitwell Shale Formation. The surface mine is in a broad upland divide area of the Cumberland Plateau. It lies between major drainage systems which dissect the plateau surface. The major stream adjacent to the former mine site is Glady Fork Creek.

In mid-1990, Skyline Coal began experiencing water problems at Glady Fork. The water concerns were in the form of off-site seepage emanating from the mine site into the adjacent creek. The seepage consisted of slightly acidic to alkaline, manganese (Mn) enriched, ferruginous waters. Seepage flows were defined as originating from the mine disturbance areas with movement to the creek via natural fracture systems or other stratigraphic flow paths existing in the stream buffer zone and stream bed itself. Although the overall impacts of the seepage to the adjoining streams were limited to aesthetics (red staining) and temporary benthic aquatic habitat concerns (iron sediment coatings), the discharge was defined as a pollution source to waters of the State and therefore implementation of remedial action plans was required to eliminate or minimize degradation potentials.

The range in primary water quality parameters of the seeps at the

time of discharge were as follows: pH = 3.4 to 7.5; alkalinity = 0 to 121 mg/l; total iron (Fe) = 4.8 to 48.6 mg/l; total manganese (Mn) = 2.3 to 34 mg/l; and, sulfate (SO_4) = 8 to 812 mg/l.

Skyline embarked on an intensive investigative program so that enough information could be collected to identify the cause of the apparent disparity between the projected and actual water quality parameters. This information was also deemed necessary in order that a reliable materials handling plan could be developed which would satisfy both the regulatory authorities and corporate custodians. This program included extensive core drilling, testing of overburden samples, and modifying the testing procedures to account for "siderite masking" of acid/base overburden characterization objectives (Wiram, 1992), re-examining the hydrogeological and geochemical structure of the overburden, and consulting with experts in the field of mine drainage. Contrary to earlier permit findings, the additional site investigations identified potential acid-producing strata that were previously ill-defined or non-existent based upon acid/base laboratory procedures that did not account for the abundance of siderite ($FeCO_3$) common to the overburden shale sequence.

Using modified analytical procedures to account for the "siderite masking", three stratigraphic horizons in the overburden were identified as being potentially acid-producing in character. The three horizons were (listed in descending order):

1) upper shales (<5.0') at the top of the Whitwell Shale Formation;
2) lenticular sandy shales (0 - 20') within the Whitwell; and,
3) pit floor or underclay of the Sewanee Seam.

A fourth type of material, the "coal cleanings" (an admixture of coal and underclay), generated in the coal extraction process, was also identified as being acid-producing in character and was subsequently targeted for proper handling and disposal during mining.

Although the different materials were demonstrated to possess pyritic materials and a lack of inherent neutralizing potentials, the acid producing character of the various strata was only sporadic in nature. With the exception of the "coal cleanings", it was not uncommon to see both acid and alkaline characteristics within the respective strata.

Typical ranges in % pyritic sulfur ($\%P_S$) contents and modified net neutralization potentials (MNNPs, expressed as tons of $CaCO_3$ /1000 tons material (Wiram, 1992)) for identified acid producing units were as follows:

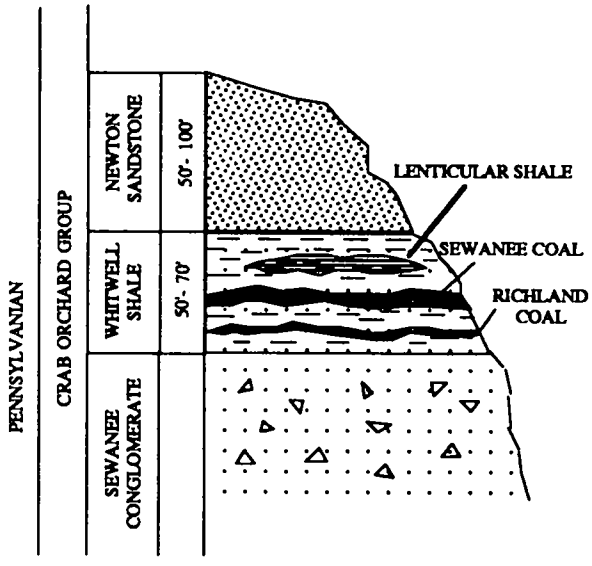
1) Upper shales: $\%P_S$ = <0.1 to 1.64, and MNNP = -26 to +55.
2) Sandy shale: $\%P_S$ = <0.1 to 1.78, and MNNP = -48 to +14
3) Underclay: $\%P_S$ = 0.12 to 0.78, and MNNP = -19 to +20

The typical range in both $\%P_S$ and MNNP observed for the coal cleanings were 0.73% to 1.60% and -45 to -24 tons $CaCO_3$/1000 tons material respectively.

Throughout the reserve area, the lower Whitwell Shale interval above the coal seam demonstrated consistently low $\%P_S$ and was alkaline in character. The Newton Sandstone, the rock unit comprising the majority of the stratigraphic column of the coal sequence, was shown to be basically inert from the standpoint of acid or alkaline producing potentials due primarily to its indurated, quartzitic cementation. An exception to the general rule for the sandstone interval existed when, on an occasional basis,

the quartzitic cementation was replaced by sparry calcite. Although the calcite cementation was sporadic in occurrence, when present, such sandstone material demonstrated excess alkalinity characteristics with MNNP values commonly in the range of 100 to 130 tons of $CaCO_3$/1000 tons material.

Figure 1
Stratigraphic Sequence in Permit Area



Eventually, these investigations, mine planning efforts, negotiations with the regulatory authorities, and protracted hearings at the administrative law level, generated a TMHP which was attributed the highest probability of success. The balance of this presentation describes the results of the application of this TMHP.

The Mining Method

Skyline's method for selectivity handling and disposing of potentially acid-producing materials involved a combination of operating techniques. Overburden movement was achieved by a combination of direct cast-blasting, tractor pushing, and dragline placement with the mine's Bucyrus-Erie Model 1300 stripping machine.

Skyline modified some phases of its traditional mining methods to accommodate the TMHP developed for the site. An important element of change was that Skyline moved the backfill-side operating bench level of the dragline from 50 feet or less off the pit floor to 85 feet above the coal. That change enhanced the machine's capabilities of handling an increasing volume of shale (i.e., problematic shale) to be placed on the bench pad as well as expanding the machine's ability to place the best shale materials in the graded backfill. Both activities were deemed important to the success of the TMHP.

The mining sequence was initiated with the cleaning of the mined out pit, placement of cleanings against the toe of the machine bench, and where appropriate, constructing a hydrologic flow-path (chimney drain) to move subsurface water off the dragline bench itself. As other sections of this paper will discuss, this phase of the mining/reclamation sequence was also the recipient of alkaline additions. The pre-blast configuration as well as the generalized geochemical characterization of the overburden are depicted in *Figure 2*.

Figure 2
Configuration of Pit Prior to Blast Cast

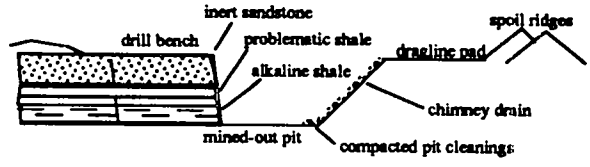
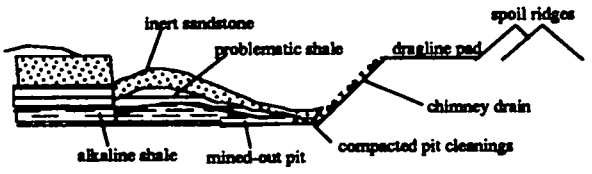


Figure 3 illustrates the position of the muck pile (shot overburden) following a normal cast-blast shot. (The shape and the distribution of the shattered rock ("the bread loaf") was established by field measurements. Thereafter, the services of blasting specialists using high speed movie cameras were also secured. An examination of the film using the "stop frame" capability of a special projector supports these initial field

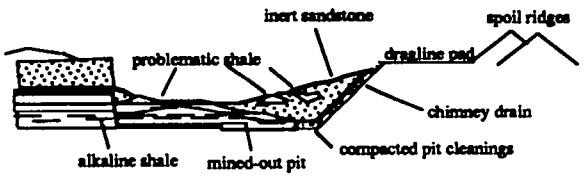
observations.) Note that the problematic shale extends across the former pit floor. However, the bulk of the acid-forming shale lies toward the highwall side of the pit and thus in a more advantageous position for selective handling with the dragline. Lastly, it is important to note that the shot rock underlying the entire length of the muck pile consists of alkaline shale materials derived from the basal section of the original overburden profile.

Figure 3
After Blast Cast



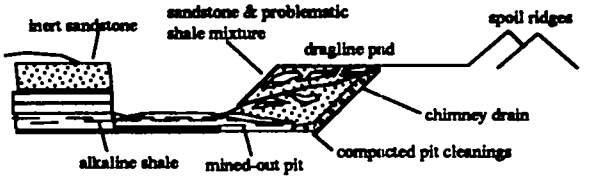
Once the overburden material was cast, tractors were used to push the upper sections of the muck pile ("the bread loaf") from the highwall side toward the adjacent backfill bank. Eventually the tractors would cut into the shattered problematic shale and the pushed material would become an admixture of problematic shale and inert sandstone. This unique configuration is depicted in *Figure 4*.

Figure 4
After Tractor Push



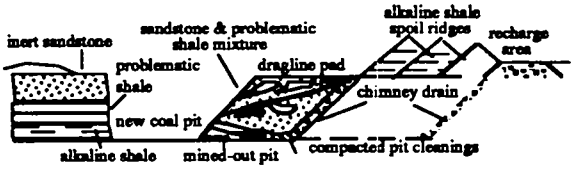
Following the tractor push of primarily inert rock materials, the dragline was used to build its dragline bench out of mostly problematic shale materials. As a practical way of providing in-field identification of the position of the problematic shale materials in the muck pile following cast blasting, stakes with notes affixed were placed along the dragline bench. Information pertaining to estimated depths to the base of the problematic materials was provided by these notes. The dragline operators would rely on the information on those stakes to control stripping depths to the base of the problematic shale. *Figure 5* shows the position of the pad relative to the other components of the mine backfill.

Figure 5
Configuration After Building Dragline Bench



Final pit excavation consisted of placing the alkaline shale materials in spoil ridges behind the machine on the dragline bench. As shown in *Figure 6*, only an extremely small portion of problematic shale materials had the opportunity of actually being cast into the spoil ridges. Overburden sampling of the graded spoil ridges at this mine proved the validity of the method. As the dragline bench has progressively moved higher into the backfill, the geochemical quality of the graded backfill interface steadily improved.

Figure 6
Final Overburden Distribution



Tractors regraded material no more than three spoil ridges behind the active cut. *(See Figure 6.)* This material was graded in lifts approximately 8 feet thick. The spoil ridges were continually worked with

tractors until the area was regraded to the approximate original contour, or slightly elevated due to the swell of the overburden. Final grading was conducted parallel to the contour in sloped areas to minimize erosion and to provide a more stable surface for redistribution of topsoil.

As a summary comment, the majority of the problematic shale ended up on or directly beneath the dragline bench. At a minimum, Skyline targeted placing 90% of the acid- producing shale materials on or below the dragline bench horizon. An estimated 30 - 40% of the acid-producing materials remained permanently buried in the cast-blast profile. As will be discussed later, limestone applications above and below the direct cast-profile were designed to prevent and/or substantially minimize acid production from this zone of the backfill.

Design Objective Of Alkaline Additions

The limestone applications had to meet both interim and long-term requirements. A synopsis of the key amendment objectives follows:

1. To subdue bacterial pyritic oxidation activities and neutralize acidities generated from unavoidable reactions occurring when potentially acid-producing materials (i.e., Whitwell interface shale and lenticular sandy shale deposits, coal cleanings, underclay materials, etc.) are exposed to the atmosphere between stripping cycles to control similar reactions in these materials once disposed in the backfill.

2. To focus alkaline recharge capabilities along pre-defined hydrologic flow-paths (i.e., dragline bench, graded backfill/topsoil interface, pit floor, etc.) to enhance a long-term alkaline availability in the backfill for the purpose of counteracting unfavorable acidity build-up. (Coarse-sized ($1^1/_2$" x 0") limestone was used in order to sustain longevity of the alkaline resource.)

3. To enhance the chemical stability of siderite ($FeCO_3$) common to the shale materials comprising the overburden sequence. Limestone additions were expected to provide high alkalinity/acidity ratios in the backfill and to thus maintain favorable pH/alkaline conditions which would prevent and/or minimize the mineral dissolution of siderite constituents in the backfill.

4. To complement the in situ alkaline shale materials at the base of the muck pile. The combination of imported limestone and native alkaline shale at the base of the muck pile was to ensure the availability of alkaline products of sufficient quantity and distribution to effectively accommodate the chemical reactions and subsequent metal-loading activities that could accompany natural backfill weathering processes (i.e., "wetting" and "drying" cycles) associated with backfill ground water re-establishment.

Key Methods, Parameters, And Assumptions

1. Overburden was sampled ahead of mining on a close-spaced grid (approximately 500 feet) using air-rotary or core recovery methods. Individual samples were collected at 5-foot vertical intervals in each exploratory hole.

2. All amendment rates associated with alkaline additions to the backfill were determined by utilizing the modified neutralization potential (MNP) (Wiram, 1992) procedure for the purpose of eliminating "siderite masking". MNP was expressed in terms of tons $CaCO_3$/1000 tons of material.

3. The standard multiplication factor of 31.25 times the pyritic sulfur content was used in calculating potential acidities (PAs) of the various overburden materials. Potential acidities were expressed in terms of tons $CaCO_3$/1000 tons of material.

4. Modified Net Neutralization Potential (MNNP) was defined as the difference between MNP and PA (MNNP = MNP - PA).

5. In defining its lime requirements, Skyline Coal used a 2,300 tons/ac.-ft. factor for estimating in situ tonnages of overburden shale and pit cleanings to be encountered in stripping. This is a "Caterpillar Performance Handbook," (Edition 17, October, 1986) value. A 250 foot radius around each hole was used to estimate the total quantity of alkalinity required.

6. Limestone application rates were based on "negative MNNP" zones identified through the 5-foot sampling and testing interval in each borehole. (These were also denoted as "net deficient" zones in other communications on the subject.) Each zone was presumed to be 5 feet thick and to occupy an area reaching half the distance to each adjacent borehole (typically 250 feet). Zones demonstrating negative MNNP values equal to greater than 5 tons $CaCO_3$/1000 tons of material were selected for the subsequent calculation of limestone application rates for the area around the borehole. The limestone application rates for each 5-foot zone demonstrating negative MNNP's equal to or greater than 5 tons of $CaCO_3$/1000 tons of material were then accumulated to arrive at the total quantity of limestone applicable t the area around each borehole.

7. The alkalinity of "net neutral" zones in each exploratory hole was not factored into the limestone quantity estimation calculations.

8. Limestone application rates were based on a stone quality of 90% purity.

Alkaline Addition Placement In The Backfill

The "building blocks" needed to develop a workable TMHP relative to the following issues began to accumulate as the investigations proceeded. These "building blocks" included the following:

1. The water quality characteristics of the earlier mined and backfilled portions of the mine, as evidenced by both post-mine ground water data and off-site seepage water quality;

2. The refinement of pre-mine/post-mine positioning of potentially acid-producing materials in the stripping sequence; and,

3. The establishment of a basic understanding of the hydrologic flow-paths in the backfill.

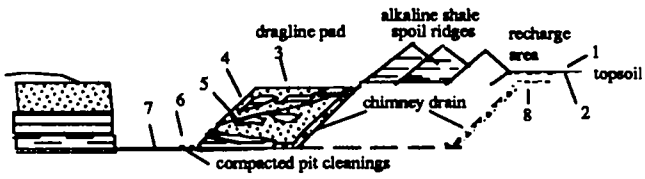
It quickly became evident that alkaline additions to the backfill would play a major part in the TMHP. These additions were to have the dual role of minimizing and/or preventing acidic reactivity in the backfill and to enhance the alkaline-loading of the reemerging ground water presence in order to meet the long-term post-mine drainage objectives following reclamation. The overall concepts for the design and planned installations of alkaline-loading mechanisms for the TMHP are credited primarily to the past research works and technical papers of Caruccio and Geidel, 1988 and 1989. The ideas and technical support for incorporating the limestone materials into the backfill in conjunction with the stripping and reclamation process stem primarily from the works of Byerly (1990) and Infanger and Hood (1980).

Eight zones in the backfill were targeted for alkaline additions. As

shown in *Figure 7*, the zones of alkaline additions are (in descending order from the reclaimed surface):

1) replaced topsoil;
2) topsoil-graded backfill interface;
3) backfill-side dragline bench;
4) pit-direct cast overburden;
5) pit-direct cast understory;
6) coal cleaning;
7) pit floor underclay/shale; and,
8) surface alkaline recharge structures (SARS).

Figure 7
Alkaline Addition Areas



Because of the substantial quantities of limestone to be applied to the backfill to account for the neutralization deficiencies identified for the involved strata and to compensate for possible unforeseen short-comings in the mine's overall success at achieving total selective handling in the backfill, Skyline split the limestone application into three equal portions. Each split was placed within a specific position within the backfill. Correct proportioning in the field was assured by use of "zone of influence" boundaries marked along the dragline bench.

The three sections of the backfill receiving the limestone amendments are as follows:

1) topsoil/backfill interface;
2) backfill dragline bench; and
3) direct cast profile.

The portion of lime placed on the topsoil/backfill interface was in addition to the amount of lime already added to compensate for possible "hot-spots" identified on the graded backfill in advance of final soil replacement.

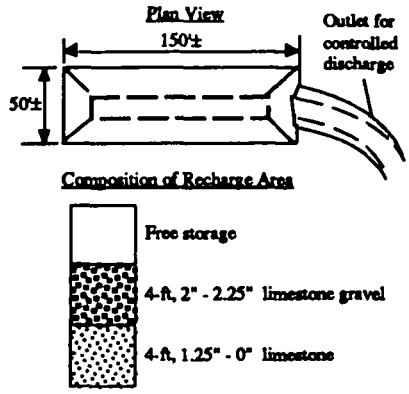
Skyline further split the lime targeted for the "direct cast profile" of the muck pile into two separate applications. Half of the required lime amount was dumped on the pit floor and spread in a uniform lift. The remaining half of the lime planned for the direct cast profile was placed and spread over the tractor push area of the muck pile.

Backfill Alkaline Recharge Structures

To enhance the introduction and subsequent movement of alkaline waters into and through the backfill, Skyline constructed a network of surface alkaline recharge structures (SARSs). *(See Figure 8.)* The SARSs were installed during final reclamation and were designed to afford maximum watershed inflow into the backfill. Furthermore, wherever possible, these surface features were positioned to overlay chimney drains installed in the backfill. In that manner, the potential for the alkaline/oxygen loading of the backfill ground water was enhanced. The key objective of this initiative was to induce metal precipitation within the backfill.

Typically, each SARS was first excavated and shaped to match the existing contours of the reclaimed land. Usually, this would result in a backfill excavation with volume of approximately 1.8 acre-feet. Limestone with a calcium content of 90% or greater was placed into the excavation. The first four feet of limestone consisted of a 1.25" x 0" "crusher-run" produce. The next four feet consisted of 2" x 2.5" gravel. The resulting loosely packed, bowl-shaped, recharge areas were anticipated to have a water holding capacity of 1.3 acre-feet, almost double the volume expected from the design precipitation event (2-year, 15 minute event).

Figure 8
Detail of Alkaline Recharge Facility



Ground Water Monitoring Results

The application of the TMHP is now running into its fourth year. *Figure 9* shows the general location of where the TMHP was implemented at Glady Fork. The TMHP was initiated in August 1992. Skyline Coal continued full implementation of the plan through final pit closure in 1995.

Prior to entering into a discussion of the water quality data of the mine site, it is important for the reader to understand that the basis for this paper is "ground water well" data and not "seepage" data. This is a practical off-shoot of the company's seepage abatement program which was initiated in October, 1990. At that time, the ground and surface water paths to the identifiable seeps to Glady Fork were intercepted, and as a result, the seeps have ceased to exist. The multi-source water captured in the process is being pumped back to the mine site. This method of control of the former seeps is expected to continue until the company installs a passive, wetland-based, water control system. Among the many objectives of this wetland is the accommodation of those seepage flows, should they in fact re-emerge following final reclamation of the mine site.

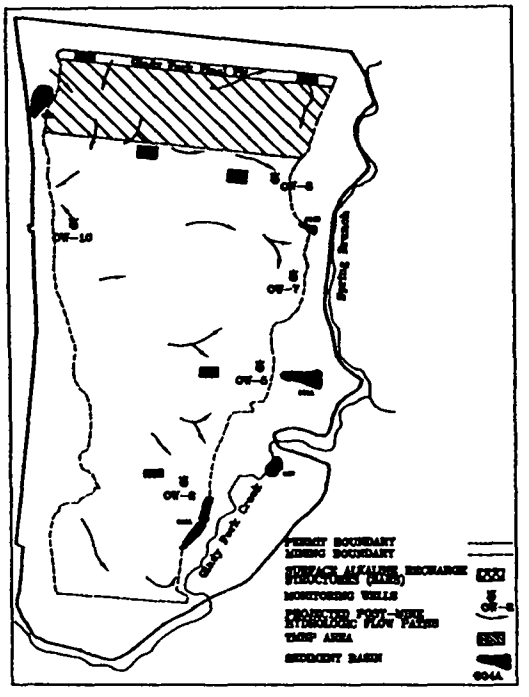
A review of the water quality trends from the majority of the backfill monitoring wells indicates an overall amelioration of the ground water associated in areas mined prior to the application of the TMHP. The improvement in ground water quality in the areas mined prior to full implementation of the TMHP initiative primarily reflects the combined benefits of contemporaneous reclamation and the backfill's inherent capabilities to control the chemical reactions which generate poor quality water. The water quality trends within and adjacent to those areas that had the direct benefits of the TMHP clearly demonstrate the prevention of significant deterioration of the ground water. The following discussions and graphic presentations summarize these observations.

Water quality data generated from five backfill ground water wells are presented here. The five wells (listed in order from south to north through the mined reserve) are OW-2, OW-5, OW-7, OW-10, and OW-8. *Figure 9* shows the respective positions of the ground water wells relative to the projected post-mine ground water flow paths in the backfill. Also, delineated on the generalized map are the TMHP and non-TMHP areas as well as locations of the installed SARS. Some discussion of the location of ground water wells relative to the SARS, TMHP area, and post-mine hydrologic flow paths is required.

- Wells OW-2, 5, 7, and 10 are similar in that they fall well beyond the boundary of the fully implemented TMHP area. These four wells reflect non-TMHP backfill conditions for approximately 90% of the total area mined at Glady Fork.As shown on *Figure 9*, ground water wells OW-2 and 5 share additional likenesses in that both are located down gradient from SARSs. The SARS up-gradient of OW-2 was installed in March, 1991. The

SARS located above OW-5 was constructed in October 1991.

Figure 9
General Location Map of Ground Water Wells, TMHP Area and SARS



- OW-7 is located approximately 1,300 feet northeast of OW-5 and along the easternmost highwall perimeter of the mined area. The well is positioned over a structural low area on the former pit-floor and is within the projected post-mine hydrologic flow-path from the TMHP area in the backfill.

- OW-8, located approximately 1,300 feet south of the final pit, is located immediately down gradient of the TMHP area and is positioned hydrologically in the backfill to effectively monitor the overall influence of the fully implemented TMHP on the ground water regime. Although OW-7 lies down gradient of OW-8, both wells are hydrologically connected in that the location of each well falls within the overall post- mine hydrologic flow pattern.

- OW 10 is offered here as a stark contrast primarily to OW-7 and OW-8. This observation well was deliberately placed in a shallow backfill area, known for its preponderance of problematic shales, and its anticipated high position in the ground water regime of the backfill. It was concluded early on, that this well would at best monitor the results of contemporaneous reclamation and possibly the influence of previous backfill/interface liming activities only. The area around this well was not the beneficiary of the direct application of the TMHP, and it had little or no chance of enjoying TMHP benefits through ground water migration.

Five ground water quality parameters are discussed here for the purpose of demonstrating the overall effectiveness of the TMHP in meeting the desired post-mine reclamation goal of minimizing backfill ground water deterioration. The five parameters to be highlighted are alkalinity, pH, sulfate (SO_4), total manganese (Mn), and total iron (Fe). With the exception to the sulfate, all the parameters chosen for comparative evaluations align with the mine's current post-mine NPDES effluent standard requirements. For sake of brevity, the comparative discussions offered below address the overall performance of the different quality parameters from a time-trend perspective. The graphic presentations provided depict a statistically computed trend line reflecting overall performance of the respective parameters as measured since initiation of monitoring at the individual wells involved.

The trend lines of the various water quality parameters presented here have been developed using a statistical software program within Harvard Graphics 2.3. The trend lines of the parameters have been statistically derived by a least squared fit of the data. The sample population of each parameter ranged from a minimum 26 to 49 data points depending on the age of installation of the respective ground

at OW-7. The ground water accumulated in the vicinity of OW-7 has not only been the beneficiary of alkaline-loading from the up-gradient TMHP area, it has also been the recipient of alkaline-loading from both surface alkaline recharge as well as backfill/interface liming initiatives. These additional sources of alkaline-loading explain the observed high alkalinity levels recorded at OW-7.

Observation wells OW-5 and OW-2 are functionally similar in that both measure backfill ground water conditions at a considerable distance (3,000 feet and 4,500 feet respectively) from the TMHP area. However, both wells are positioned down-gradient of SARSs. They demonstrate alkaline-recharge in those areas of the backfill where basically no limestone additions have been made. Data derived from both wells show favorable increases on alkalinities with time. Average alkalinity levels observed for ground water at OW-5 and OW-2 are 159 mg/l and 140 mg/l respectively.

The elevated alkalinity trend observed for OW-5 reflects the dissolution of limestone within the SARS under the influence of both natural precipitation and operational pump-back recharge to the backfill. Surface water pump-back activities to the SARS above OW-5, began in late summer, 1994, and has continued to the present. Both surface and spoil ground water which accumulates in Basin 002A is routinely pumped back into the backfill to prevent basin discharge of possible non-compliant effluent. The pump-back mechanism allows for natural "air-stripping" of both iron and manganese prior to re-entry to the backfill ground water resource.

Contrary to OW-5, the overall trend line for OW-2 is flat to slightly negative. The character of this trend line reflects the long-term influence of continuous backfill dewatering activities associated with pumpage at Basin 002A and past sediment pond modifications. Changes in pond structure were implemented in the Fall of 1991 for the dual purposes of controlling pond discharge and to establish a framework for ultimately developing a permanent passive wetland system for the backfill discharge area. Basin 002A modifications resulted in significant lowering of water levels in the pond and related backfill ground water. Since modification, Basin 002A has been under a continuous pump-back mode. The continuous pumping at 002A has resulted in considerable ground water fluctuations in the adjacent backfill over time. The sustained mechanical manipulations of the ground water is primarily responsible for the trend character observed for alkalinity at OW-2.

Ground water in the vicinity of OW-10 displays a slight negative trend in alkalinity. This trend undoubtedly reflects the consumption of alkalinity in response to past and/or current natural acidification/neutralization activities in the backfill. Despite the apparent decline in natural alkalinity depicted by the graph, the inherent backfill carbonate resources appear to be effective in maintaining favorable alkalinity levels in the ground water.

Backfill Ground Water - pH

The benefits of the alkaline additions to the backfill can also be discerned from a trend line graph of pH values *(See Figure 11)* for the ground water collected from the selected wells.

The pH values at both OW-7 and OW-8 ground water wells have remained within the 6.0-7.0 range. The slight negative trend in pH observed for OW-8 reflects pit-dewatering/final closure activities. No pH conditions have been recorded which might indicate adverse

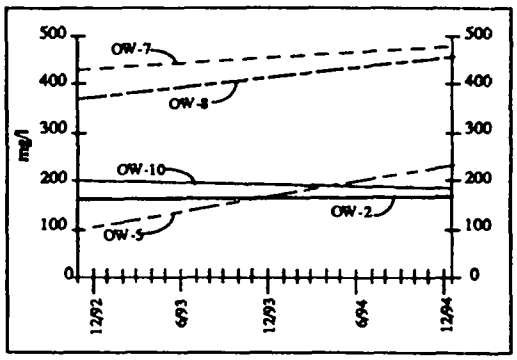
water monitoring wells involved. For example, the trend line derived for OW-8 was based upon the least number of data points (i.e., 26) per parameter due to the fact that the monitoring well in question is the younger of all the wells. Ground water well OW-2 is one of the older wells installed at Skyline Coal's Glady Fork Mine, therefore, a greater population density of data points (i.e., 49) exists for this well. The frequency of data collection of all the ground water monitoring wells discussed in this paper is monthly. Because of the population density of the data involved and the desire to present legible graphics absent data cluttering, a decision was made to present only trend line information of the various parameters at this time. It is recognized that the trend line presentations cannot be subjected to rigorous interpretation without further statistical analysis beyond which is currently offered here, however, the information provided does afford the reader with a preliminary overview of the favorable geochemical responses of adding alkaline materials to the backfill for the purpose of improving post-mine ground water quality conditions. Such transfer of information despite its preliminary nature is the ultimate objective of this paper.

Backfill Ground Water - Alkalinity

Figure 10 depicts the various alkalinity trends established for the selected wells in the backfill.

The elevated positions of the trend lines for the hydrologically connected wells OW-7 and OW-8 clearly illustrate the positive effects of the alkaline additions. Although not discernible from this simplified graph, up until March, 1994 (the date of initiation of pit dewatering for pit closure),the slope of the alkalinity trend for OW-8 was greater than that of

Figure 10
Glady Fork - Backfill Ground Water - Alkalinity



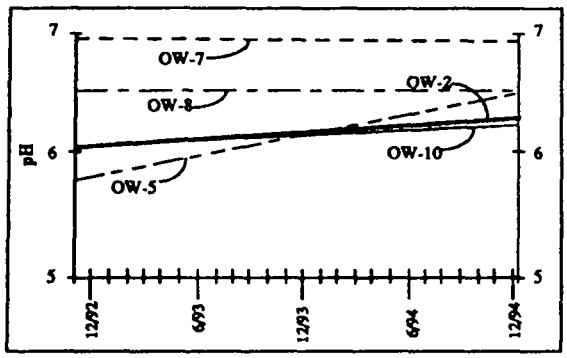
OW-7, and individual readings actually were higher that those of OW-7. Pit dewatering activities associated with final pit closure, however, lowered the next series of readings for OW-8 by an estimated 50 to 60 mg/l during the summer and fall of 1994. This temporary operational impact explains the currently observed position of the OW-8 alkalinity trend below that of OW-7. Despite these unavoidable operational manipulations, the average alkalinity value for ground water in the vicinity of OW-8 is an impressive 433 mg/l. The maximum recorded alkalinity for the 2 1/2 year time period of monitoring is 537 mg/l.

As previously mentioned, both OW-7 and OW-8 wells are positioned within the primary post-mine hydrologic flow-path in the backfill. OW-7 is located in an area which had not been subject to the TMHP. OW-8, on the other hand, is located within a TMHP area. OW-7 pre-dates OW-8 and is hydrologically down-gradient of OW-8. Despite the fact that OW-7 lies approximately 2,100 feet from OW-8, its high ground water alkalinity values reflect down-gradient migration of alkaline charged waters from the TMHP area.

North-to-south ground water movement has resulted in a significant "pooling" of alkaline-loaded waters in the structural low area in the backfill

acidification activities in this part of the Glady Fork backfill.

Figure 11
Glady Fork - Backfill Ground Water- pH



The steady pH performance observed at OW-7 reflects the cumulative benefits of both recharge contributions from the SARSs and subsurface ground water recharge from the TMHP area. OW-7's distance from any significant draw-down impacts associated with pit/dewatering activities provides a further explanation of the consistency of the pH values observed at this well location.

OW-5 is located in backfill that had not been the recipient of the TMHP. It is located down-gradient of an SARS. Considering the past backfill-dewatering activities and the subsequent manipulations of the ground water regime nearby (in the vicinity of OW-2) it is apparent that the surface alkaline recharge structure up-gradient from this well has played a beneficial role in maintaining steady state pH conditions in the backfill. Evidence indicates that the backfill ground water conditions have gone from acidic pHs (<6.0) to more favorable pHs (>6.0) in less than a 2 1/2 year period following reclamation.

A major segment of the monitoring well-screen for OW-10 is positioned in pit cleanings. The steady pH recorded for OW-10 (pH = 6.0± range) reflects the presence of natural alkaline resources in the backfill and the benefits derived from the inundation of coal-cleanings positioned at the base of the backfill.

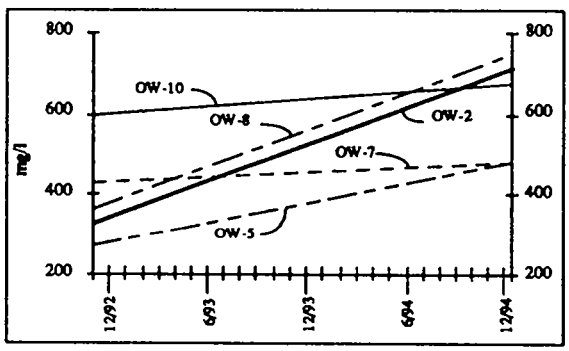
Backfill Ground Water - Sulfates (So_4)

The sulfate data is openly presented here for the purpose of allowing the readers to formulate their own opinions regarding possible geochemcial reactions occurring in the backfill, in particular, the case for suppression of pyrite oxidation. Commonly, observed increases in sulfate concentrations are automatically attributed to pyrite oxidation in the backfill and that sulfate concentrations are expected to decline with time if indeed pyrite oxidation is being prevented. However, the authors suggest that caution be used in the broad application of this "axiom" of AMD technology in the interpretation of the sulfate trends being presented here as well as any other backfill water quality investigations. There exist other possible sources of sulfate productivity in the backfill besides pyrite oxidation that can and most likely are contributing to the observed sulfate increases to date in the Glady Fork Mine study. In light of the marine/freshwater paleo-environment source (i.e., distributary channel/backwater embayment setting) of the former sediments comprising the overburden in the Glady Fork Mine area, other mineralogical sources of sulfates can be expected to be present. When considering the degree of overburden fragmentation involved with the mining process coupled with the past and present ground water mechanical manipulations (i.e., sediment pond dewatering, continuous spoil pump-back activities, final pit closure dewatering, etc.) of the backfill, natural increases in sulfate concentrations are expected to occur. The mere cyclic "wetting and drying" and subsequent "flushing" phenomena associated with the mechanical manipulations of the backfill ground

water resource yields sulfate loading from multiple mineralogical sources.

Figure 12 depicts the various trend-lines for sulfates measured at the respective well locations in the backfill. All of the sulfate trends illustrate increases in concentrations with time. The only differences in sulfate performance between wells are in degree, as indicated by positioning of respective trend-lines on the graph, and the rate of productivity (kinetics of reactivity), as measured by the respective slopes of the individual trend lines.

Figure 12
Glady Fork - Backfill Ground Water - Sulfate (SO_4)



The sulfate trend-line for OW-7 shows elevated sulfate conditions in the backfill. The flattened slope of the line, however, reflects rather stable conditions relative to actual acid/base reactivity in the backfill. Another possible explanation for the observed flattened slope is that equilibrium conditions have been reached between a mixture of waters having varying degrees of sulfate concentrations. The sulfate performance observed here reflects the cumulative effects of recharge to the backfill ground water originating from the previously identified up-gradient SARS and the migration of ground water from the TMHP area.

The sulfate trend observed at OW-8 exhibits a response to pit-dewatering/pit closure activities. Up to March, 1994, the sulfate levels recorded at OW-8 were historically lower than those observed at OW-7 and the overall slope aspect of both trend-lines were essentially the same. Shortly after the initiation of full scale pit pumping in early March of, 1994, sulfate concentrations at OW-8 increased steadily. This is attributed to the "flushing" of the backfill materials as ground water levels in the backfill near that hole were caused to fluctuate. One significant observation that can be made from the data is that sufficient alkaline materials are present in the backfill to mitigate the effects of the acidic reactions. Thus both acid-loading and metal-loading have been minimized or prevented.

The slopes observed for the sulfate trends measured at both OW-5 and OW-2 somewhat mirror the trending sulfate levels observed for OW-8. As with OW-8, the overall sulfate trends for both wells reflect significant influences from past and present mechanical manipulations of the ground water regime. Surface water pump-back activity to the SARS up-gradient of OW-5 has induced frequent "flushing" of the backfill. Both pump-back and backfill-dewatering has significantly influenced the overall sulfate-loading in the vicinity of OW-2. Undoubtedly in both situations, some backfill acid/base reactivity has accompanied such mechanical influences. The ground water quality data supports the conclusion, however, that the overall impacts of undesired reactions have been minimized by the alkalinity releases from the SARS and the natural alkaline resources of the backfill materials.

In comparison to measurements made at all other observation wells at Glady Fork, the sulfate concentrations at OW-10 have been consistently higher. The elevated sulfates coupled with the slight increase in sulfate productivity

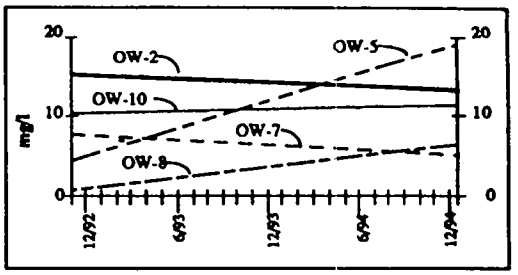
over time is associated with a backfill situation where greater than normal proportions of pyritic-bearing backfill and coal-cleanings are present and the benefits derived from fully implemented TMHP are lacking.

Backfill Ground Water - Metal Loading Fe And Mn

The benefits of alkaline additions is further evidenced by the significant reductions in metal-loading to the backfill ground water regime. Trend comparisons of metal concentrations of both Fe and Mn reveal the positive effects of the alkaline additions in preventing or significantly minimizing development of undesirable acid conditions and restraining metal dissolution in the backfill. Although the company collects both total and dissolved metal data routinely, only total values are shown here. Total iron and manganese data are presented here for the purpose of illustrating the "worst case" scenario of backfill ground water conditions. It is understood that suspended solids can conceivably create a negative bias in the metals data which is independent of the alkaline additions to the backfill, however, the idea is to present favorable conditions resulting from use of total metal information knowing that use of dissolved metal data would only show a much better case. *Figures 13 and 14* illustrate the overall trends of both metals for each of the wells under consideration here. The following discussions briefly highlight key observations made from the trend analyses provided.

Manganese (Mn). Compared to background Mn levels in the majority of the non-TMHP areas of the backfill, a significant reduction in Mn-loading is observable at OW-8. Using a calculated mean of 14 mg/l for Mn data generated at OW-10 for comparison purposes, a 3.5X to 4X fold reduction in manganese dissolution has been achieved at OW-8.

Figure 13
Glady Fork - Backfill Ground Water - Total Manganese (Mn)



This favorable reduction is noted despite the fact that the trend shown for OW-8 is negative. Over the 2.5 year time-period of monitoring at OW-8, the Mn levels have fluctuated from 1.8 to 8.1 mg/l and averaged 3.7 mg/l. As depicted in *Figure 13*, the Mn trend-line at OW-8 is climbing. This negative trend is skewed upward by recent pit-dewatering/final pit closure activities. Prior to the pit dewatering events at Glady Fork, the Mn performance fell in the range of 1.8 to 4.0 mg/l. These values are consistent with post-mine NPDES effluent discharge standards and are anticipated once again following final pit closure.

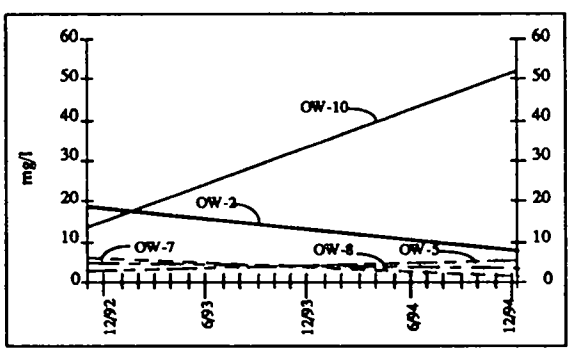
The progressive decline in Mn concentrations observed at OW-7 reflect the positive effects of highly alkaline, low Mn-bearing ground water generated in the TMHP area migrating southward toward OW-7. At OW-7, a 2.5X to 3X fold reduction in Mn levels can be observed when compared to ground water quality at OW-10. From initial monitoring at OW-7, the Mn levels have varied from 1.80 mg/l to 9.30 mg/l. The last year values cluster around 5 mg/l and are trending downward.

The continued mechanical manipulation and associated "flushing" of the backfill as a result of pump-back activities and pit dewatering have influenced the overall Mn trend-lines for both OW-5 and OW-2. Their concentrations of Mn are high. A favorable trend in Mn reductions at OW-

2 was noted following the halt of pump-back and backfill dewatering activities. This suggests that this parameter will be reduce with time once passive wetland systems are finally established and backfill pumping activities are permanently terminated.

Iron (Fe). The alkaline additions element of the TMHP has played a significant role in reducing the concentrations of iron and subsequent acid productivity in the backfill. Although it cannot be definitely proved by the information gathered to date, the positive trends in reduction of both acidity and iron levels lends support to the proposition that the alkaline additions have played a role in suppressing pyrite oxidation in the backfill. The effects of the alkaline additions on such backfill reactivity are best illustrated by observing the major reductions in Fe loading shown in *Figure 14*.

Figure 14
Glady Fork - Backfill Ground Water - Total Iron (Fe)



Using OW-10 data for comparisons, a 10X fold reduction in iron levels observed at both OW-7 and OW-8 can be tied directly to the influence of the limestone additions of the TMHP. The average Fe values observed for both OW-8 and OW-7 are 3.4 mg/l and 2.8 mg/l respectively.

As mentioned in the preceding discussions of Mn-loading, the on-going pit-dewatering/pit closure activities have caused "flushing" of the backfill and as a result have impacted the ground water quality in the vicinity of OW-8. However, despite an observed "spike" in Fe levels following the draw-down events, the overall trend is toward significant reduction of ferruginous water in the backfill.

Once final reclamation at Glady Fork is complete and the backfill ground waters receive the full benefits of the "workings" of the TMHP, Fe concentrations are expected to drop and remain below 3.0 mg/l iron on a permanent basis. Backfill water quality data collected during the interim of final pit inundation (March 1993 - March 1995) provides a technical basis in support in of such a favorable post-mine ground water quality projections. Prior to final pit dewatering, the iron levels at OW-8 averaged 2.4 mg/l. With final pit closure the post-mine ground water quality should stabilize in the 2.4 to <3.0 mg/l range.

The on-going pump-back/surface alkaline recharge activities associated with OW-5 provides a plausible explanation for the slight negative trend in iron-loading associated with this well. A reversal in trend can be expected once these operations are halted. Iron concentration trends depicted for OW-2 provides a technical basis for that expectation. As indicated in *Figure 14*, OW-2 Fe concentrations show a steady reduction over time. This favorable trend was accelerated following completion of pump-back and backfill -dewatering activities in the vicinity of that well.

When all mechanical manipulations of the backfill ground water regime are terminated, it is expected that continued declines in iron-loading will be achieved. Such favorable trends in the backfill ground water will ensure long-term effectiveness of passive wetlands designed for handling post-

mine water quality discharges from the southern end (Pre-TMHP area) of the Glady Fork Mine.

Conclusions And Recommendations

In review of the overall design objectives of the TMHP, in particular the element of limestone additions, the ground water data generated from monitoring the ground water in the Glady Fork backfill indicate significant benefits have been derived. In fact, the monitoring data reflect that all of the basic design criteria and related objectives built into the TMHP are being met. The following summary highlights salient observations, interpretative inferences, and conclusions drawn from the monitoring data collected to-date.

- The alkaline additions have made significant inroads toward achieving major reductions in the concentrations of acidity, iron, and manganese. Iron concentrations in amended backfill are up to 10 times less than those in unamended backfill. While manganese levels are 2.5 - 3.0 times lower.

- The consistently high alkalinity levels recorded and the observed steady rise of ground water alkalinity with time provides substantial evidence that limestone placement along pre-determined hydrologic flow-paths in the backfill has generated the desired alkaline-loading. In areas where limestone has been added, observation wells had alkalinity concentrations that were 2 to 3 times greater than those of a well (OW-10) located in the backfill where no alkaline amendments have been made.

- Favorable alkaline-loading accompanied by reductions in metal concentrations are observed to be associated with the SARS. The ground water data from points down-gradient of the SARSs show steady increases in alkalinity and corresponding decreases in metal levels (both Fe and Mn).

- Because of the observed significant reductions in levels of acidity, iron, and manganese, it can be inferred (not proven) that limestone additions to the backfill have been effective in subduing bacterial pyritic oxidation activities during and following mining. The observed persistent high alkalinities, lowered metal contacts, and the complementing favorable pH conditions in the backfill have undoubtedly minimized or prevented pyritic oxidation reactions. Further, the same favorable water quality parameters have undoubtedly assisted in maintaining a more favorable stability field for siderite in the backfill.

Suggested Future Investigative Efforts

All of the favorable trends referenced above point to achievement of the desired long-term goal of preventing damage to the environment.

The results of the ground water data generated to-date raise technical questions, which if answered through additional on-site investigations, would expand the knowledge base for achieving workable solutions to effective AMD abatement and control. The following are suggested investigative efforts.

- The high alkalinities (400-500 mg/l range) observed as a result of the limestone additions to the backfill, suggest that the basic geochemical mechanisms of the passive anoxic limestone drains (PALDs) are at work at the Glady Fork Mine. These high levels of alkalinity can only be achieved through the accelerated dissolution of the added limestone. This in turn suggests the involvement of CO_2 in the backfill. Further investigative efforts are needed to document the role of CO_2 in the

backfill and its ability to enhance the solubility of limestone.

- Further studies are needed to document the effectiveness of the limestone additions along defined hydrologic flow-paths in the backfill. Such investigative efforts should focus on development of technical guidelines which will facilitate the application of this attractive alkaline-loading method.

- Investigation and formal documentation of the workings of the SARSs are in order. Technical guidance documents should be developed subsequent to this investigation so that the application of such facilities to other mines can be better evaluated.

Conclusions

In conclusion, this paper provides documentation of the benefits of planned placement of alkaline additions to the backfill. The limestone materials added to the Glady Fork Mine in combination with the synergistic workings of all elements of the TMHP have been effective in minimizing and/or preventing acid drainage development in the backfill. Further, the fully implemented TMHP has been effective in substantially curtailing the dissolved metal concentrations in the backfill ground water regime.

Post-mine NPDES effluent standards will be achieved for the areas under the immediate influence of the TMHP. This is supported by the current ground water quality trends in the backfill. Therefore, in the unlikely event that seepage discharges were to occur from the TMHP section of the backfill, off-site damage in the form of AMD will not occur.

With respect to the southern portions of the mine where the TMHP was not applied, the benefits of alkaline-loading and metal-loading reductions derived from the TMHP area, the alkalinity contributed by the SARS, and the inherent alkaline resources within the backfill material itself positions these portions of the mine for meeting post-mine NPDES discharge objectives.

Acknowledgments

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THE USE OF SEDIMENT SAMPLING TO EVALUATE THE SUCCESS OF HYDROLOGIC RECLAMATION [1]

by

ROBERT G. LIDDLE [2]

Abstract Surface water monitoring efforts usually involve random, infrequent, grab sampling of the water column. This method may not detect irregular or infrequent pulses of pollutants and may be ineffective in assessing the physical and chemical quality of the benthic zone. The benthic zone can accumulate metals and often supports the aquatic insects essential to a healthy aquatic ecosystem. Sediment analysis was conducted on two mine-project sites in Kentucky and Tennessee to supplement existing chemical and biological sampling of streams and lakes. Sediments were analyzed for 15 metals and trace elements (Al, Sb, Ar, Be, Cd, Cr, Cu, Fe, Pb, Mn, Ni, Se, Ag, Tl, and Zn) and fine-particle size analysis was done using air elutriation methods. Each sediment sample was analyzed using total metal extraction, EPA's Toxicity Characteristic Leachate Procedure (TCLP), and a four-step sequential extraction procedure. The four-step sequential extraction procedure provided the most meaningful indication of metals and trace elements that could be of concern. The results showed distinct sediment chemistry and particle-size distribution between mined and unmined streams. Sediment chemistry mirrored pore-water chemistry and aided interpretation of the stream water and biological data in mined and unmined streams that was not otherwise apparent by sampling the water column alone. Sequential sediment analysis and fine-particle size analysis shows promise in assessing the success of hydrologic reclamation at mined sites.

KEY WORDS: Trace Elements, Water Quality, Coal, Mining

Introduction

Hydrologic Reclamation includes, but is not limited to, the restoration of baseline water quality and sediment quality to support the aquatic ecosystem and other potential water users. Assessing hydrologic impacts and hydrologic reclamation typically includes sampling of the water-column and comparing it with ambient criteria or water-quality standards. This method alone may ignore the condition of the benthic zone that can be critical to the aquatic ecosystem. In addition, the spacial and temporal nature of water column sampling may not represent the acute or chronic water chemistry of the stream affecting aquatic life.

This report summarizes the results of stream and lake sediment sampling programs at watersheds having active and proposed coal mines. The results were used to compliment the other physical, chemical, and biologic surveys done at the sites. It was intended that the sediment sampling would reduce the time and cost of obtaining ambient water conditions over water column sampling alone and be more representative of in-field conditions. The physiochemical environment at the sediment level is generally ignored in typical coal mine stream sampling. Yet the benthic zone is host to much of the food source used by higher aquatic life such as fishes. The bottom dwelling aquatic insects rely on the sediment pore-water quality as much as the water column itself.

The effects of mining on sedimentation loads has been well documented (Walters, 1995; EPA, 1981). Stream pollutants and sediments from mining may occur in

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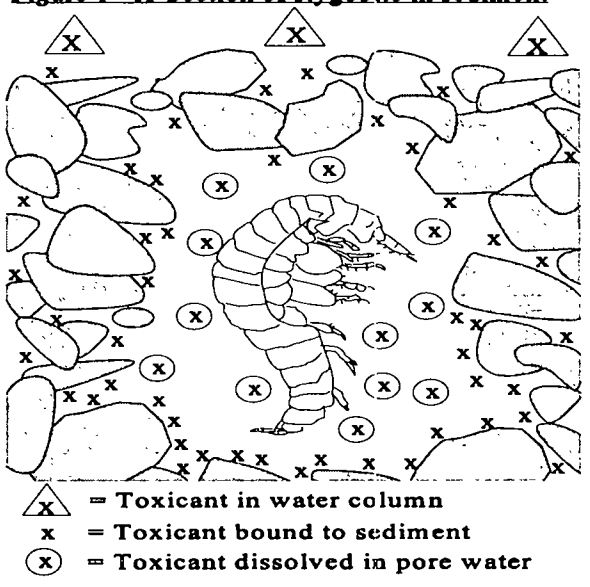
pulses due to the nature of sediment distribution and transport and also operational considerations such as the periodic opening and closing of treatment system discharge valves. Periodic random water column sampling may or may not represent the true pollutant load placed on a stream since it ignores bedload sediment (Horowitz, 1991). Frequent and composited stream water sampling is needed to characterize the stream-water-column environment fully; however, use of diffusive gradients in thin films (DGT) technology is helping fill the gap (Zhang, 1995, p. 3391). Sediment sampling aids in filling in the gaps between periodic water-column sampling and presents a more complete picture of the quality of the aquatic environment. A previous regional study in Tennessee has shown that water sampling alone only identifies 75 percent of the mine impacted drainages (O'Bara, 1993, p. 87). Another regional study in Virginia concluded benthic impacts from copper and zinc were occurring in coal mined drainages despite regulatory water column monitoring (McCann, 1993).

Several literature reviews on recent sediment research are provided by Fuller (1995, p.164) and Clement (1985, p. 236R). Sediment sampling can include both bioassay (Ingersoll, 1995A) and direct chemical analysis. Stream and overbank sediment sampling is useful in evaluating mining-related stream impacts (Ridgway, 1995; Wielinga, 1994; Prairie, 1994; Coupe, 1993).

Metal and trace element contaminants may be dissolved or settle as particulate matter. They may be adsorbed to sediments, precipitated, coprecipitated, bonded organically, undergo cation exchange, or be incorporated into the crystal lattices of sediments that coat the bottom of the stream channel. The type of sediment dictates the ability to bind with other metals. Horowitz (1991, p.41) found the relative capacity of collectors in freshwater systems were: Amorphous Iron Oxides > Total Extractable Iron > Total Organic Carbon > Reactive Iron > Clay Minerals > Total Extractable manganese > Manganese Oxides. Horowitz (1991, p. 69) also studied the relative importance of selected physical and chemical factors on sediment trace-element chemistry; he found: Amorphous Iron Oxides > Surface Area > Total Organic Matter (LOI) > total Extractable Iron > Other Organic Matter > Percent less than 63 μm > Reactive Iron > Total Organic Carbon > Percent less than 125 μm > mean grain size > percent clay minerals > percent less then 16 μm > percent less than 2 μm > Total Extractable Manganese > Manganese Oxides.

The sediment pore-water quality is a function of both the sediment and the overlying water quality (Burton, 1992). Figure 1 shows this relationship. Unless the benthic zone is restored after mining, the aquatic ecosystem will not be restored to baseline levels.

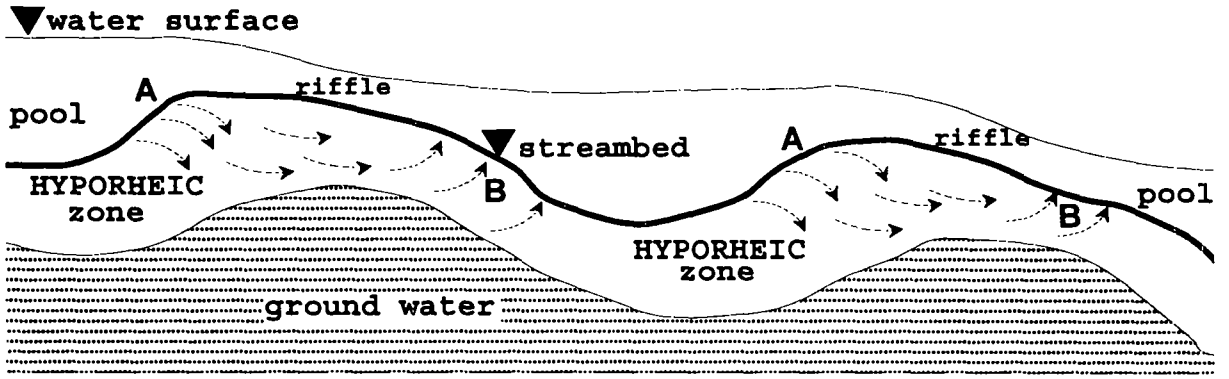
Figure 1 - X-Section of stygobite in sediment



After Burton, 1992

The sediment zone contains a reservoir of metals, trace elements, and nutrients that can be re-suspended in the water column by diagenic processes (Buffle, 1994, Horowitz, 1991, p. 49) such as floods or bioturbation, changes in physiochemical conditions (pH, eH, DO, temperature, *etc.*), and biochemical reactions (bacteria, humic substances, *etc.*) that increase toxicity or solubility (e.g., methylization of mercury). The sediment layer is often the location of the ground water and surface-water interaction zone, termed hyporheic zone (See Figure 2), which fluctuates seasonally and supports a diverse ground-water ecosystem (Hendricks, 1992; White, 1992, Gilbert, 1994, Dahm, 1992). Polluted ground waters recharging a stream in this zone can adversely affect the interstitial waters of the sediment without detectable change in the water quality of the water column. Therefore, the use of benthic and interstitial

Figure 2 - Conceptual Model of Hyporheic Zone



A = Zone of Down-welling **B** = Zone of Up-welling → = Direction of water
After Hendricks, 1992. (Vertical exaggeration)

invertebrates, such as stygobites, can also be used as indicators of ecosystem damage, along with sediment analysis (Plenet, 1992; Gilbert, 1994).

Heavy metals and trace elements such as Al, Cd, Cu, Se, Pb, Ni, and Zn are known to be associated with coal mine drainages (EPA, 1981; Liddle, 1994; Liddle, 1995). Often these minor elements are not monitored since it is incorrectly assumed they either do not exist in high concentrations or they are removed during the iron and manganese treatment process. For example, metal transport through the spoils to streams via fractures is unaffected by surface-water treatment ponds, yet this groundwater is seldom monitored at minesites unless it forms springs. Another example is aluminum, which is often reported in the literature as having a low (<1 mg/l) concentration at mine sites. However, it was not until August 1988 that EPA promulgated the aquatic freshwater criteria of 0.087 mg/l for aluminum (53 FR 33178). And recently the EPA is considering reducing the drinking water standard for arsenic from 0.040 mg/l to 0.006 mg/l, because of carcinogistic concerns (Eaton, 1994). Since many trace-element water-quality criteria have not been determined until recently, and EPA criteria are very low, one must guard against assuming only iron and manganese are of concern. The sediment-analysis methods presented here show techniques that can be used to determine the amount of metals available for bioaccumulation and the amount of metals easily taken back into solution in the pore water or water column. The absence of these metals in the sediments is a good indicator they are not significant in the water column.

Methods

At project site #1, sediment, biological, and water sampling was conducted in four watersheds with similar geology and topography and with little or no non-mining development: 1) 18.4 km^2 (7.1 mi^2)Little Yellow Creek in north Tennessee; 2) nearby 25.9 km^2 (10 mi^2) Little Clear Creek in southern Kentucky; and 3) Bennett Fork, and Langley Branch). Little Yellow Creek watershed is proposed for surface coal mining and the others have been mined since the late 1970's. As such, they are excellent locations to evaluate true baseline conditions with post-SMCRA mining impacts. Little Yellow Creek and Little Clear Creek historically contained a large population of the endangered Blackside Dace. However, the mined watershed has been without the Blackside Dace since 1985, according to stream surveys. Mining is suspected of causing this extirpation. Complete information on the hydrology of the sites is available from OSM (Liddle, 1995A). A second project site is Bruce Creek above Cove Lake State Park. The stream was sterilized by metal contamination in 1993 emanating from an anoxic drain in coal spoils (Sugar Ridge Mine, 200 acres) north of Knoxville, TN. While water quality has since improved due to treatment, the question of restoration of the benthic zone remains unanswered.

About 15 to 20 sediment samples less

than 4 cm deep were taken at each site and composited. Five sites were sampled on each stream at the base of a riffle zone during low streamflow (Burton, 1992, p. 89; USGS, 1995, p. 20; USGS, 1970, p. 53). These five composites were then combined to produce one sample felt to represent the stream. It was important to take many samples, because stream sediments from coal mines are spatially and seasonally variable (Gunten, 1994) and lab analysis was to be done on only a few grams of each sample. Sediments were stored at or near freezing until analyzed, which took several months. This may have produced errors; recent research has shown samples should be frozen and analyzed immediately to prevent biochemical changes (Becker, 1995). The less toxic the sediments the more susceptible they are to changes.

The sampling procedures followed recommendations of Burton (1992, p. 32), Ingersoll (1995) and the specifications of ASTM standards D3974-81 and D3976-92 (ASTM, 1993, Vol. 11.02). ASTM Standards D4822-88, D4823-88, D5074-90, and D3976-92 were also followed depending on the stream being studied (polluted vs. unpolluted). Sampling and laboratory practices generally followed the USGS procedures (USGS, 1982; USGS, 1993).

Particle size analysis was done using sieves and air elutriation methods to obtain very fine particle analysis to 0.0015 mm. Samples were analyzed in three ways: 1) total metals were extracted using hydrochloric and nitric acid digestion; 2) the EPA Toxicity Characteristic Leaching Procedure (TCLP); and 3) a 4-step sequential extraction method. The sequential extraction method is based on Tessier's (1979) work, and has been modified to fit geochemical extractions of inorganic metals and trace elements common in mining investigations (Tessier, 1979; Chao, 1983; Chao, 1984; Horowitz, 1988; Horowitz, 1990; and Horowitz, 1991a). The sequential extractions were done on samples screened to less than 0.63 micron to allow clearer comparisons based on particle size (See Horowitz, 1991a). No grinding or crushing of particles was done.

Results of Grain Size Analysis

The sediment grain-size distributions showed that the mined watershed of Little Clear Creek had more fine clays and silts than the unmined watershed of Little Clear Creek (See Figure 3). Field investigations included suspended sediment sampling during storm events to develop a sediment graph for both streams. The results revealed that during storm events the coal-mine haul roads were a significant source of sediment loads to the stream since most of the haul-road drainage entered the Little Clear Creek untreated.

The grain-size distributions for Bruce Creek revealed that stream sediments had less clays at greater distances away from the mine site. Since clays have the highest surface area per particle, and sediment surface area is directly proportional to the capacity to capture and retain metal and trace elements (Horowitz, 1991, p. 16) the grain-size distribution can shed light on the potential for metal retention.

Results of Sediment Chemistry Analysis

The total digestion of the samples with nitric and sulfuric acid did not show any meaningful trends between mined and unmined streams or downstream from the anoxic drain at Bruce Creek. This is because the amount of metals contamination was very low. In high contamination sites such as hazardous waste sites, concentrations of metals are large enough to show differences between natural and contaminated sites. The EPA TCLP extractions confirmed that none of the sediments came close to meeting EPA toxic criteria. The metals contamination at coal mines would be expected to be more subtle. The total digestion of sediments does not distinguish between metals held in the crystal lattice of the rocks, metals bound organically, metals precipitated, or metals adsorbed. Consequently the total metals results show more about the mineral content of the sediments than about the bioavailability of metals.

The total metal extraction (TME) analysis is somewhat useful as a quality check on sequential extraction methods. If a total extraction with hydrofluoric and nitric acid is conducted on ground sediments, then all metals including those in the crystal lattice should be released. (Note Hcl, HNO_3, H_2SO_4 alone will not extract all metals without HFl) One could then analyze the remaining sample after the sequential extractions to determine what metals were not released. The quality check then becomes:

$$TME = E_1 + E_2 + E_3 + E_4 + E_n \ . \ . \ . + TME_e$$

Where E_1 to E is the mass of metal released from the first to last sequential extract, respectively, and TME_e is the mass of metal remaining on the extracted sediment after all sequential extracts are completed. In this study, our contract laboratory would not do the HFl extractions due to safety concerns, therefore we were unable to obtain this quality check step. Instead, splits were run on the samples to ensure reproducible results.

The sequential extractions were designed to mimic the physiochemical processes that would strip sediments of metals and trace elements from successively higher energy orders. Use of sequential extractions, while not an exact science, has been used extensively by the metal mining community for geochemical prospecting (Chao, 1983, 1984). Extraction #1 is designed to extract exchangeable, adsorbed, and carbonate metals using 1 gram residue per 8 ml 1M Sodium Acetate adjusted to pH 5 with Acetic Acid, agitated for 1 hour. Extract #2 is designed to release metals easily bound to manganese oxides and reactive iron using 1 gram of residue per 50 ml of 0.1 M Hydroxylamine Hydrochloride in 0.1 M HNO_3, shaken for 30 minutes at 25°C. Extract #3 is designed to release tightly bound metals from amorphous iron oxides and manganese using one gram residue to 250 ml solution of 0.25 M Hydroxylamine Hydrochloride in 0.25 M HCl, heated and shaken at 50°C for 30 minutes. Extract #4 should release organically bound metals using 3 ml of 0.02 M HNO_3 and 5 ml of 30% H_2O_2, adjusted to pH 2 with HNO_3, heated and occasionally shaken at 85 °C for 2 hours. Then adding a 3 ml aliquot of 30% H_2O_2 adjusting to pH 2 with HNO_3 again and heated and agitated for 3 more hours. After cooling, add 5 ml of 3.2 M Ammonium Acetate in 20 % v/v HNO_3 , then dilute with distilled water to a 20 ml volume, shake at 25°C for 30 minutes, then analyze. Each extract was separately analyzed for Al, Sb, Ar, Be, Cd, Cr, Cu, Fe, Pb, Mn, Ni, Se, Ag, Tl, and Zn.

The results of total digestion of the samples for the first project site are listed below in mg per kilogram dry weight (Figure 1). The results show the difficulty in comparing multiple sources of sediments using total sediment analysis. For some parameters such as arsenic, the unmined watershed was highest. In unmined Fern Lake the lead concentration was highest probably due to the use of lead fishing sinkers. Mining affected streams were higher in zinc, nickel, copper, and chromium. However, this method may be useful for screening out elements that are too low to consider further such as Ar, Ag, Se, Tl, Sb, Be, and Cd.

Table 1 - Total Digestion of Sediment

	L. Yello Creek	Fern Lake	Benn-ett Fork	L. Clear Creek	Lang-ley Br.
mg/kg	UNMINED		MINED		
AL	2290	14 K	2210	3410	6580
Sb	0.5	0.5	0.5	0.5	0.7
Ar	9.4	5.6	4.2	0.7	2.2
Be	0.4	1.1	0.2	1.0	0.8
Cd	0.1	0.2	0.2	0.1	0.02
Cr	5.6	8.7	8.3	8.3	9.9
Cu	5.6	20.6	15.7	10.8	15.8
Fe	12 K	40 K	9900	12 K	26 K
Pb	4.5	204	6.9	8.6	144
Mn	260	347	254	426	1220
Ni	10.4	30.3	15.7	13	32.6
Se	0.3	0.2	0.2	0.2	0.2
Ag	0.1	0.1	0.1	0.1	.001
Tl	0.1	0.2	0.2	0.1	0.1
Zn	29	107	40	49	83

The results of the first sequential extractions in mg/kg dry weight are as follows:

Table 2 - SEQUENTIAL EXTRACT #1

	Yello Cr	Fern Lake	Benne -t Cr	L. Clear	Lang-ley
mg/kg	UNMINED		MINED		
Al	263	192	34	20	87
Cu	8000	7310	1540	1010	480
Fe	331	1210	35	16	61
Pb	832	1550	29	69	47
Mn	840	304	121	509	613
Ni	12	8	4	5	10
Zn	165	1100	304	200	99

These results show concentrations in the 63μm fraction of sediment and are useful in correlation studies. The USGS has used similar procedures to find relationships between chemical and physical parameters with sediment chemistry on a regional basis. If a good correlation is found, then concentrations of the elements can be predicted for similar regions. Examples of the USGS national sediment models are:

$$Cu = 0.52\ LOI + 0.351\ Al + 0.395\ Ti + 0.875$$

and

$$Ni = 0.963\ Al + 0.248\ TOC + 0.003$$

(See Horowitz, 1991, p. 79)

To evaluate the total effects of the metals in the environment, one must consider the total metal available, not just the 63 μm fraction. Since the previous data shows mined watersheds were higher in clays, the total contribution of metals must be recalculated, based on the percentage of clays to the total sediment. One could reanalyze the total sediments, but research has shown the contribution from sands and silts to be negligible in most cases (Horowitz, 1991). The results of this comparison follow:

Table 3 - SEDIMENT TOXICITY: Extracts 1 + 2

	L. Yello	Bennett	L. Clear	Langley
mg/kg	Un-Mined	MINED WATERSHEDS		
Al	90	57	188	266
Fe	275	192	830	551
Mn	84	46	279	405
Zn	37	99	105	71
Cu	872	1120	643	373
Pb	105	12	50	46
Clay	8.9%	28.5%	37.8%	52.2%

Assuming the sediments have similar bulk densities and assuming the first two sequential extractions represent the bioavailable fractions; the results show that the mined watersheds are higher in some metals. Lead is higher in the unmined watershed, owing to lead fishing tackle, while zinc and aluminum are generally higher in mined watersheds. Some element concentrations are not significantly different in the sediments (e.g., copper). Pore water samples taken by centrifuging sediments from Langley Branch below a strip mine revealed dissolved aluminum concentrations of 1.68 mg/l, well above the EPA chronic aquatic standard of 0.087 mg/l (53 FR 33178, 8/30/88), while the maximum concentration in the water column was only 0.02 mg/l. The dissolved iron concentration in pore water was 2.17 mg/l, which is above the 1.0 mg/l EPA threshold (EPA, 1985); the maximum iron content in the water column was 0.028 mg/l. The zinc pore water concentration was 0.053 mg/l, which is near the EPA aquatic life chronic criteria of 0.059 mg/l (52 FR 6214, 3/2/87). The water column concentration of dissolved zinc was 0.03 mg/l. The sediment analysis failed to correlate with the elevated lead concentrations of 0.006 mg/l in the pore water, which is above the EPA chronic aquatic criteria of 0.0013 mg/l (50 FR 30791, 7/29/85); the dissolved lead concentration in the water column was 0.002 mg/l.

At the second project site at Bruce Creek, sediment samples were taken at treatment ponds below an anoxic drain and in several places in the receiving stream. The purpose was to assess how far downstream the metal contamination had spread. Water from the anoxic drain flows from 0.25 to 2 cfs and contains several toxic concentrations of metals, as compared with aquatic receiving-stream standards. The following table compares concentrations from the anoxic drain with receiving-stream standards:

Table 4 - Anoxic Drain Quality

	Total Metals	Dissolved Metals	Aquatic Criteria[1]
Cu	20 μg/l	1 μg/l	6.5 μg/l
Al	1810 μg/l	940 μg/l	87 μg/l
Cd	26 μg/l	0.5 μg/l	4.6 μg/l
Fe	85 mg/l	40 mg/l	1.0 mg/l
Mn	35.4 mg/l	35 mg/l	none
Ni	470 μg/l	430 μg/l	88 μg/l
Zn	388 μg/l	190 μg/l	59 μg/l
Hg	2.3 μg/l	-	.012 μg/l

[1] EPA Freshwater Aquatic Chronic Criteria.

Although the water is now being treated under a bond forfeiture situation, the stream was already contaminated to the point where no aquatic life was found (Turner, 1995). Recent stream-water-quality sampling of the water column about 0.2 km (1/8 mile) downstream

revealed aluminum, copper, iron, nickel, and zinc are still at the EPA aquatic life chronic criteria. Water samples taken about 3.2 km (2 miles) downstream showed aluminum, nickel and zinc were still at EPA freshwater aquatic life chronic criteria. The results of the sediment analysis showed decreasing metal concentrations as one sampled downstream:

Table 5 - Sediment Toxicity: Extracts 1 + 2

mg/kg	Upper pond sedimnt	Lower pond sed.	1.75 mi down stream	2 mi. Down stream
Al	575	399	242	22
Cu	584	664	466	43
Fe	4975	2223	433	74
Mn	5726	1113	725	234
Ni	104	7.5	4.5	1.5
Zn	153	221	113	10
Clay	80 %	35 %	22 %	3 %

If one looked at total metals analysis, and even TCLP data, the results would not show this trend. For example, aluminum total digestion results for the upper pond to the lower segment were 4080 mg/kg, 8460 mg/kg, 5640 mg/kg, and 3070 mg/kg respectively. Total zinc concentrations were 180, 39, 37, and 49 mg/kg respectively. Similar results were noted with Cu, Mn, and Ni. Since biological data shows aquatic life survival increases the further downstream from the anoxic drain one samples, and the water column sampling shows a similar trend in improvement, one could logically hypothesize that the availability of metals in the sediments would also increase. Pore-water sampling and biological assays would be needed to support this hypothesis.

Additional information was obtained from the sediment data. Sequential extract #11 implied that the unmined watersheds had higher available metals (i.e., easily adsorbed metals and carbonates) than the mined watersheds studied. This could be the result of the mined watersheds adding iron to the stream that resulted in higher-energy binding of metals and trace elements to the sediment. The second extractions showed much more metals in the mined watersheds. The EPA TCLP extractions did not mimic any of the sequential extractions, as might have been expected. A comparison of sequential extractions from both project sites indicates metals would likely be a problem at levels greater than 100 mg/kg total.

Evaluation of the sediment quality data currently is done using comparisons of affected and unaffected reaches. The EPA has only recently established sediment-quality criteria for several organic constituents; metal criteria are still lacking, and bioassay has been historically used in evaluating dredged sediments (EPA, 1987, Zarba, 1988, Mancini, 1992, and Iannuzzi, 1995). Some States have developed threshold criteria based on comparative sampling (Kelley, 1984) and Canada has issued Provincial Guidelines for Sediment Quality (Prairie, 1994).

CONCLUSIONS

The EPA TCLP extractions were not very useful for estimating pore-water chemistry or as indicators of affected drainages. This test was designed based on municipal waste leachate testing (Kimmell, 1988). Most coal mine drainage will not result in metal contamination that would be high enough to be classified as "toxic" according to this EPA test.

The total metals analysis was not useful in predicting either the availability of metals and trace elements or comparisons of affected verses unaffected drainages. The total metals analysis may serve as a check on the results of sequential extractions.

Sediment sequential extractions shows promise in identifying the availability of metals and some trace elements. Sequential extractions #1 and #2 were particularly useful; other extracts would be useful in other investigations such as geochemical prospecting or organic contaminants. Sequential extractions are better suited for investigating metals of elevated concentrations (>100 mg/kg) and are less valuable in looking at trace amounts of elements. The OSM is now using a five-step sequential extraction procedure to better define the speciation of metals in sediments.

Partical-size analysis of sediments is important for assessing the quality of the sediments and to evaluate the results of sediment metal extractions.

A lab experienced in sediment analysis should be used so that samples can be quickly analyzed upon collection, otherwise errors can occur. Short delays in getting a sample analyzed, even if the sample is frozen, has been shown to affect the toxicity of the sediment (Becker, 1995).

Additional research is needed in sampling and preparation of both pore-water and sediment from typical high gradient first and second-order streams. More research is needed in establishing standard sequential extraction methods that replicate bioavailable fractions typical of coal mining geochemistry (Horowitz, 1991). More comparative studies are needed of sediment quality, pore-water quality, water quality, sediment bioassay, and aquatic bioaccumulation at mine sites. Additional work is needed in establishing sediment-quality standards.

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SESSION 6: ACID MINE DRAINAGE PASSIVE TREATMENT - DRAINS

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ACID MINE DRAINAGE TREATMENT WITH OPEN LIMESTONE CHANNELS[1]

by

Paul F. Ziemkiewicz[2], David L. Brant[2], and Jeffrey G. Skousen[3]

Abstract. Acid mine drainage (AMD) is often associated with mining of pyritic coal and metal deposits. Typical AMD associated with coal mines in the eastern U.S. can have acidity and iron concentrations ranging from the teens to the thousands of mg/l. Aluminum and manganese can be present in concentrations ranging from zero to the low hundreds of mg/l. Much attention has been devoted to developing inexpensive, limestone (LS)-based systems for treating AMD with little or no maintenance. However, LS tends to coat with metal hydroxides when exposed to AMD in an oxidized state, a process known as "armoring". It is generally assumed that once armored, LS ceases to neutralize acid. Another problem is that the hydroxides tend to settle into plug the pore spaces in LS beds forcing water to move around rather than through the LS. While both are caused by the precipitation of metal hydroxides, armoring and plugging are two different problems. Plugging of LS pores can be avoided by maintaining a high flushing rate through the LS bed. Armoring, however, occurs regardless of water velocity. This study investigated the influence of armoring on LS solubility and the implications of armoring and plugging on the construction of open (oxidizing) LS channels for treating AMD. We evaluated the AMD treatment performance of armored and unarmored LS in oxidizing environments both in laboratory and field studies. The results showed ferric and aluminum hydroxide floc remained suspended in solution until the LS was allowed to dry. As the floc dried, the LS became armored. The laboratory study treated AMD with armored LS (ALS) from two field sites and unarmored LS (ULS). ALS dissolved 25 to 33% more slowly than ULS. The field study surveyed 2- to 8-year-old rock-lined waterways constructed for erosion control. One waterway was constructed of sandstone rip-rap and seven others were constructed with LS. The results indicated that OLC's, though armored, continued to reduce acidity at rates similar to those of the laboratory study. The results were used to verify a dissolution kinetics model which predicts the required dimensions of an OLC for treatment of given flows and acidities.

Additional Key Words: Abandoned mine lands, chemical water treatment, passive treatment, reclamation

Introduction

Acid mine drainage (AMD) continues to be one of the largest problems facing the mining industry. AMD originates from active and abandoned mines as pyrite (FeS_2) or other metal sulfides associated with the mineral deposit are exposed to oxidizing conditions. Upon exposure, the sulfide minerals progress through a combination of auto-oxidation and microbial oxidation reactions to produce large amounts of acid, iron and sulfate. This acidity then dissolves other minerals, releasing ions such as manganese and aluminum. The resulting solution is AMD. Upon reaching a stream, AMD alters the chemical balance: it consumes alkalinity, introduces metal ions and generally degrades its biological productivity. If sufficiently severe, AMD will also render the receiving waters unfit for human, agricultural, industrial or recreational use (Atlas and Bartha 1987).

AMD can be treated with alkali chemicals and this is the method of choice for most active mines. This is expensive and must continue long after mining has ceased. An alternative is passive treatment. Passive treatment refers to any zero to low maintenance AMD treatment scheme. These systems are of increasing interest as state, industry and federal partnerships are formed to rehabilitate watersheds damaged by historic mining. Passive systems offer low maintenance, inexpensive, and long-term solutions to AMD remediation (Brodie 1990, Hedin 1989). Anoxic limestone drains (ALDs), wetlands, or a combination of both are the most often used passive systems (Faulkner and Skousen 1994). Wetlands are effective in handling low acid loadings but often encounter difficulties or fail under high loading. Problems with ALDs occur when ferric iron, aluminum,

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or ferrous iron and dissolved oxygen are present. Under these conditions, metals will precipitate and armor the LS reducing its dissolution efficiency compared to unarmored LS. Precipitates may occlude all of the pore volume within the drain preventing water from contacting the LS.

Studies by Pearson and McDonnell (1974, 1975a, 1975b) found that armored LS (ALS) dissolved at about 20% the rate of unarmored LS (ULS). Ziemkiewicz et al. (1994) conducted a preliminary study of OLCs on field sites and developed a spreadsheet to estimate LS volumes and channel dimensions for treating AMD. Open limestone channels will become armored, presumably reducing the limestone dissolution rate to 20% of ULS. But unlike ALDs, plugging of LS pores can be controlled by maintaining high flows and the armoring effect can be accounted for by adding a design factor of five (Ziemkiewicz et al. 1994).

This study compares the AMD treatment efficiency of armored and unarmored LS in the laboratory. In addition, a field study was conducted to survey existing open LS-lined waterways to evaluate the effects of armored LS on AMD treatment.

Experimental Design

Laboratory Study

The lab study was conducted using containers (2 liter, high-density polyethylene) filled with 2.3 kg (5 lbs) of 5 - 10 cm (2 - 4 in) ALS or ULS (Figure 1). One of five sources of AMD or deionized water (blank) (1.2 liters) was added to each of the containers. The two sources of ALS were from Robinson Run (RR) and Dola, WV. The ULS was from the Deer Valley formation provided by Action Mining in Somerset County, PA. The five AMD sources (all Pittsburgh coal bed) were: Maidsville Seep near Morgantown, WV; Shaw Mines Run and Weir-11, near Meyersdale, PA; Coal Run, near Salisbury, PA; and a synthetic AMD containing only sulfuric acid and deionized water. Only 13 of the 18 possible LS/water combinations were used in the study (Table 1), and each of the selected combinations had 3 replications.

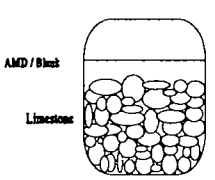

Figure 1. Containers used for the laboratory study filled with limestone and AMD or deionized water.

The method for estimating the solubility of ALS vs. ULS was adapted from Watzlaf and Hedin (1993). Water samples (60 mls) were collected with 60-ml plastic syringes from containers in duplicate (one sample for general water chemistry and one for metals analysis) at the following time intervals after water introduction: 0 hr, 1 hr, 2 hrs, 4 hrs, 6 hrs, 12 hrs, 18 hrs, and 24 hrs. The samples were filtered (0.45 micron) and metal analyses samples were acidified with 1 ml of concentrated nitric acid prior to submission to the NRCCE Analytical Laboratory for analysis. The parameters tested were: pH (electrode), electrical conductivity (conductivity bridge), alkalinity and acidity (Brinkman auto-titrator), and concentrations of iron, aluminum, manganese, calcium, magnesium (Leaman Labs inductively coupled plasma spectrometry), and sulfate (Milton Roy Spectronic 20) (Clesceri et al. 1989).

Field Study

The field study surveyed existing rock-lined waterways on abandoned mine lands (AML) sites containing AMD (Table 2). These waterways were constructed for erosion control or stream bank stabilization only. One waterway was constructed with sandstone and the other seven waterways were made with LS. Two water samples (250 mls each) were collected at identified distances along the channels (one for general water chemistry and one for metals analysis) and analyzed as described above (Clesceri et al. 1989). The samples were field filtered (0.45 micron) and acidified with 1 ml of nitric acid for metals analysis or cooled to 4°C for general chemistry. Flows were measured with a Marsh - McBirney model 2000 Flo-Mate electromagnetic flow meter for larger flows (> 95 l/min or 25 gpm) or a calibrated bucket and a stopwatch for smaller flows (< 95 l/min or 25 gpm). Distances were measured with a 100-foot surveying rope.

The results of water quality analyses from field channels were plotted against our kinetics spreadsheet (RBOLD) designed to predict the dimensions required to treat AMD with OLCs (Ziemkiewicz et al. 1994). RBOLD translates ALS dissolution kinetics into spreadsheet which estimates the reduction in acid load for a given LS channel, or estimates the size of a channel required to achieve desired acid load reductions.

Description of Field Sites

NRCS Coral/Graceton Site

The Coral/Graceton site is located adjacent to U.S. Route 119 immediately northeast of the towns of Coral and Graceton in Indiana County, PA. The channel is 220 m long, 3 m wide and 0.1 m deep (720 x

Table 1. Water and limestone combinations used in the laboratory study. Each combination checked had three replications.

Water	Robinson Run ALS	Dola ALS	ULS
Maidsville	X		X
Shaw Mines		X	X
Weir-11		X	X
Coal Run		X	X
Synthetic AMD		X	X
Deionized Water	X	X	X

9 x 0.5 ft) on a 10% slope, and was constructed with sandstone. The flow of AMD through the channel, measured at the source and mouth of the rock lined waterway, was 1323 l/min (350 gpm) and the acidity at the source was 550 mg/l.

Morgantown Airport Site

The Morgantown Airport site is located adjacent to U.S. Routes 119/857 east of Morgantown, WV. There are two channels (both LS and heavily armored) at this location. The first channel (West) is 46 m long, 1.3 m wide and 0.1 m deep (150 x 4 x 0.5 ft) on a 14% slope. The second channel (East) is branched, with the first branch being 21 m long (70 ft) and the second branch being 27 m (90 ft) long (same widths and depth as West channel) both on 20% slopes. The flow of AMD in the West channel was 113 l/min (30 gpm) and the acidity was 410 mg/l. The total combined flow in the East channel was 76 l/min (20 gpm) and the acidities were 355 mg/l for the first branch and 335 mg/l in the second. The flow rates were equal at the sources and the mouths of each channel.

NRCS Eichleberger #2 Site

The Eichleberger #2 site is located in Bedford County, PA, 6.5 km southeast of the village of Coaldale. The channel is 49 m long, 2 m wide and 0.1 m deep (160 x 6 x 0.5 ft) on a 20% slope, and was constructed with LS that became heavily armored. The flow through the channel was consistent at 378 l/min (100 gpm) and the acidity at the source was 510 mg/l.

Pa DER Site

The Pennsylvania Dept. of Environmental Resources site is located in Bedford County, PA, 1.6 km west of the village of Defiance. This channel is 11 m long, 1 m wide, and 0.1 m deep (37 x 3 x 0.5 ft) on a slope of 60% with a flow of 95 l/min (25 gpm). The acidity was 2600 mg/l at the source. This channel is also constructed of LS that became heavily armored.

PA Game Commission Site

The Pennsylvania Game Commission site is also a small channel of 11 m long, 1 m wide, and 0.1 m deep (35 x 3 x 0.5 ft) on a slope of 45%. It is located on the northeast side of Vintondale in Cambria County, PA. The flow is 484 l/min (128 gpm) and the acidity is 330 mg/l at the source. This channel was constructed of LS and became armored after construction.

Cottage Town Site

The Cottage Town site is located 1.6 km west of Cairnbrook in Somerset County, PA. The channel is 137 m long, 1.3 m wide and 0.1 m deep (450 x 4 x 0.5 ft) on a 9% slope with a flow of 302 l/min (80 gpm) throughout the entire length. LS was used for the construction of the channel. The LS was heavily armored and the AMD had an acidity of 32 mg/l at the source.

NRCS - Opawsky Site

The Opawsky site is located in Armstrong County, PA, 1 km south of Mosgrove. This site was different from the other sites due to the construction of a wetland 46 m (150 ft) that was installed at the bottom of the upper section of OLC. The top portion of the channel was constructed of LS for 46 m long, 2 m wide and 0.3 m deep (150 x 6 x 2 ft) on a slope of 9%. The LS was armored and the flow of AMD throughout the entire system was 907 l/min (240 gpm). This portion of the channel entered a wetland that covered an area of 350 m^2 (7.6 m by 46 m). The lower 137 m (450 ft) of the channel was also constructed of LS that was armored. The acidity at the source was 30 mg/l.

Results

Laboratory Study

The initial acidity of the Maidsville seep (2080 mg/l) was reduced to 925 mg/l (56% reduction) after 24 hours with RR ALS (Figure 2). This compares to ULS that eliminated 65% of the acidity after 24 hours. The ALS from Dola completely eliminated the Shaw Mines'

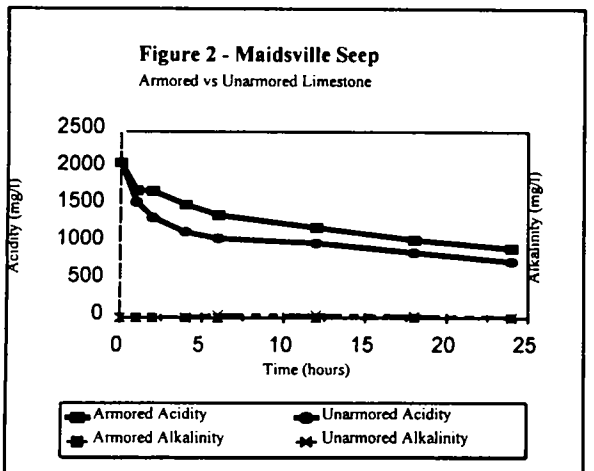

Figure 2. Acidity reduction and alkalinity generation of Maidsville Seep AMD with armored and unarmored limestone.

initial acidity of 518 mg/l in the containers after 6 hrs (Figure 3). Unarmored LS achieved 100% treatment after 4 hrs. Alkalinity production leveled off after 12 hours for both armored (75 mg/l) and unarmored (120 mg/l) LS.

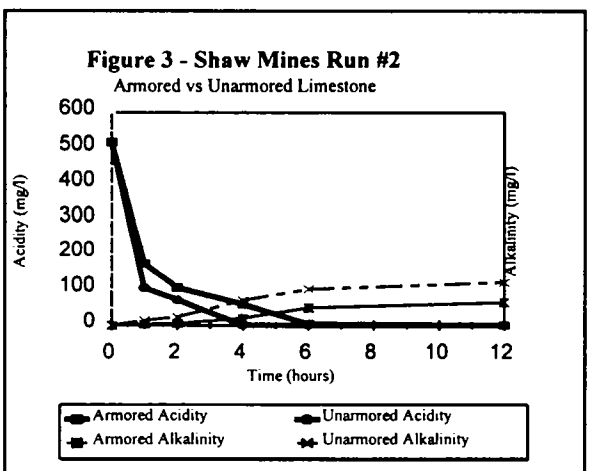

Figure 3. Acidity reduction and alkalinity generation of Shaw Mines AMD with armored and unarmored limestone.

The initial acidity of Weir-11 (1370 mg/l) was reduced to 20 mg/l (99% reduction) after 21 hours using Dola ALS (Figure 4). Unarmored LS treated all the acidity and produced 50 mg/l alkalinity during the same time period. Coal Run, a stream contaminated by a turn of the century deep mine, had an initial acidity of 905 mg/l (Figure 5). The Dola ALS and ULS both completely neutralized the acidity of the water after 21 hrs. Both types of LS produced net alkaline water during the same time period, producing 61 mg/l for the Dola ALS and 84 mg/l for ULS.

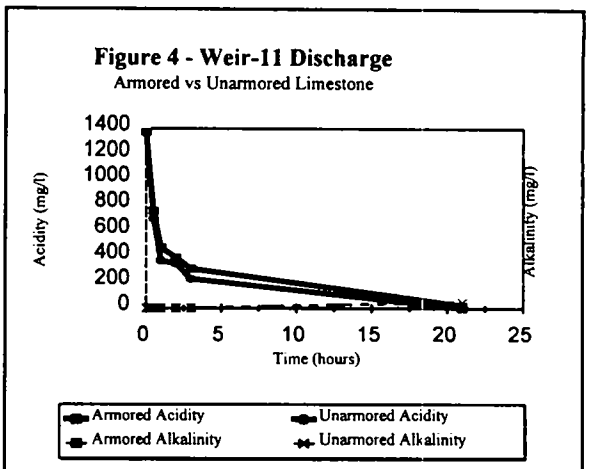

Figure 4. Acidity reduction and alkalinity generation of Weir - 11 AMD with armored and unarmored limestone.

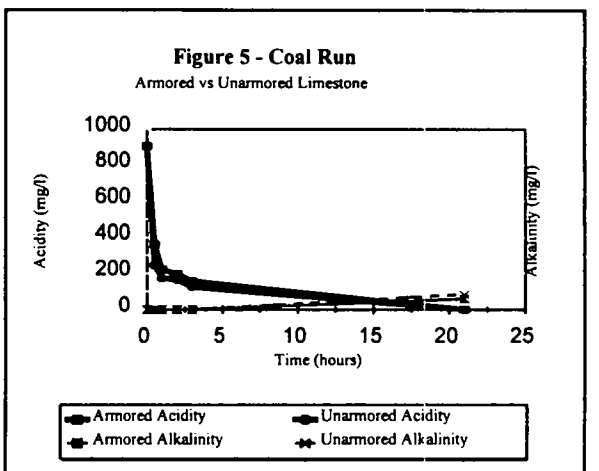

Figure 5. Acidity reduction and alkalinity generation of Coal Run AMD by armored and unarmored limestone.

Deionized water was added to the three LS types used in the study to isolate the effect of armoring in the absence of acid leachate. The results indicate that the ALS initially produced some acidity but that the solutions became alkaline within the first hour. The ULS produced the highest alkalinities at 50 mg/l while Dola ALS produced 30 mg/l and RR ALS gave 40 mg/l (Figure 6).

The three LS types were treated with a synthetic AMD (0.02M H_2SO_4). The acidity was completely neutralized within 2 hrs with ALS and within 1 hr for ULS. Alkalinity generation leveled off at 67 mg/l after 18 hrs for ALS and 85 mg/l after 4 hrs for ULS (Figure 7).

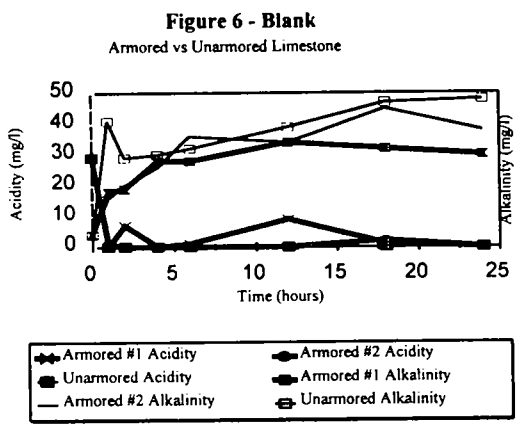

Figure 6. Acidity and alkalinity generation with deionized water on armored and unarmored limestone.

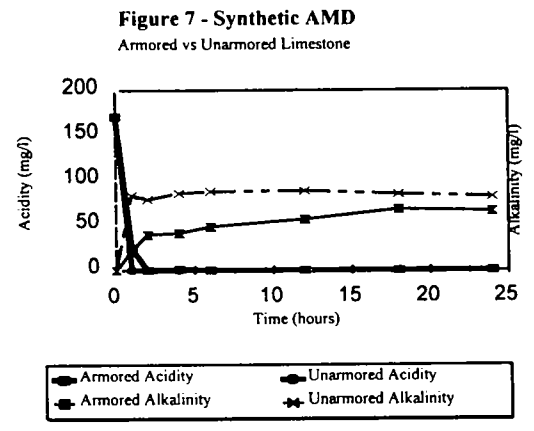

Figure 7. Acidity and alkalinity generation of Synthetic AMD with armored and unarmored limestone.

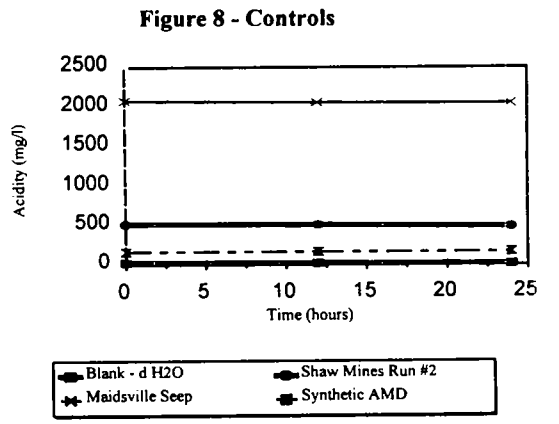

Figure 8. Acidity changes over time with three AMD sources without limestone.

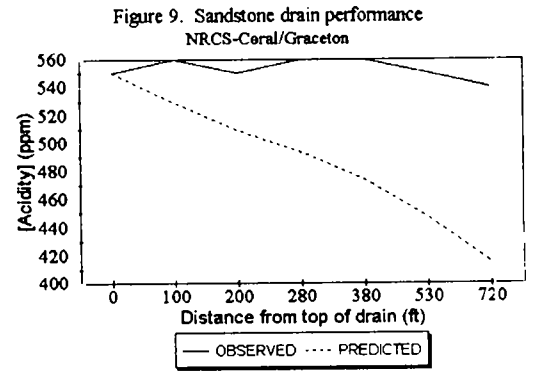

Figure 9. Observed and predicted acidity reductions from a sandstone drain at the Coral/Graceton site in Indiana County, PA.

Four water types used in the study were placed in cells and monitored over a 14 hr period to confirm whether any changes occurred in acidity levels on exposure to air. Figure 8 indicates that no changes occurred.

Field Study

The NRCS Coral/Graceton waterway was constructed with sandstone to serve as a control to LS channels. The prediction line was based on the use of LS. The resulting acidity reduction on the site was 0.0028% per ft (Table 2), much less than the predicted 0.034% per ft if it would have been constructed with LS (Figure 9).

The Morgantown Airport West channel performance was better than predicted (Figure 10). The actual acidity reduction was 0.0800% per ft (Table 2) compared to the predicted reduction of 0.032% per ft. The Morgantown Airport East channels also performed better than predicted with an acidity reduction of 0.0780% per ft compared to a predicted reduction of 0.020% per ft (Figure 11).

The NRCS Eichleberger #2 channel also performed better than expected with an actual acidity reduction of 0.2250% per ft compared to a predicted reduction of 0.104% per ft (Figure 12). The PA DER site is a very short channel with high acidity, but removes 0.1080% of the acidity per ft. This was an order of magnitude better than the predicted acidity reduction of 0.010% per ft (Figure 13).

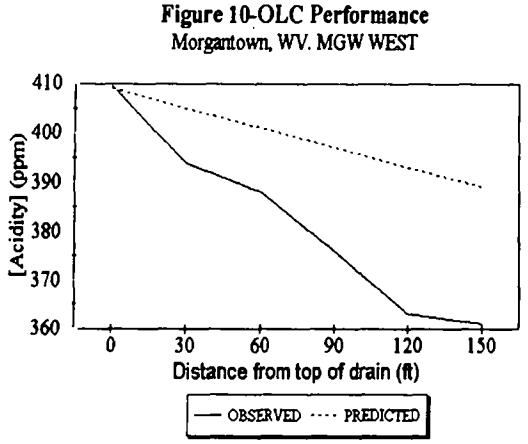

Figure 10. Observed and predicted acidity reductions of an open limestone channel at the Morgantown Airport west drain.

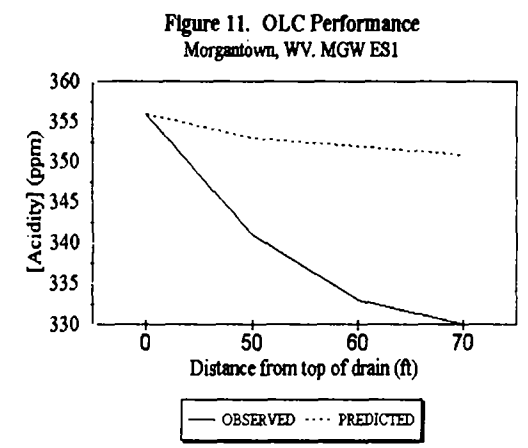

Figure 11. Observed and predicted acidity reductions of an open limestone channel at the Morgantown Airport east drain.

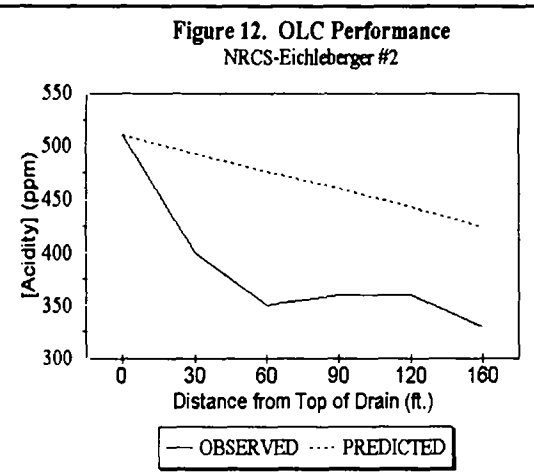

Figure 12. Observed and predicted acidity reductions of an open limestone channel at the NRCS Eichleberger #2 site in Bedford County, PA.

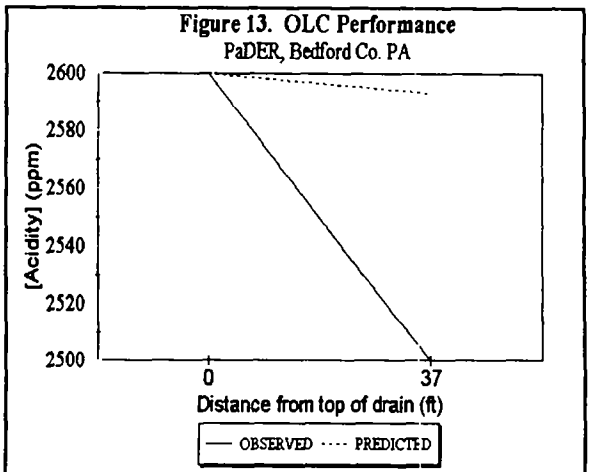

Figure 13. Observed and predicted acidity reductions of an open limestone channel at the Pa DER site in Bedford County, PA.

The PA Game Commission OLC is also a very short channel (Figure 14), but it shows an impressive performance (1.7710% acidity removal per ft compared to a predicted performance of 0.044% per ft acidity removal). The steep grades of these two channels really increased water velocities and enhanced LS dissolution.

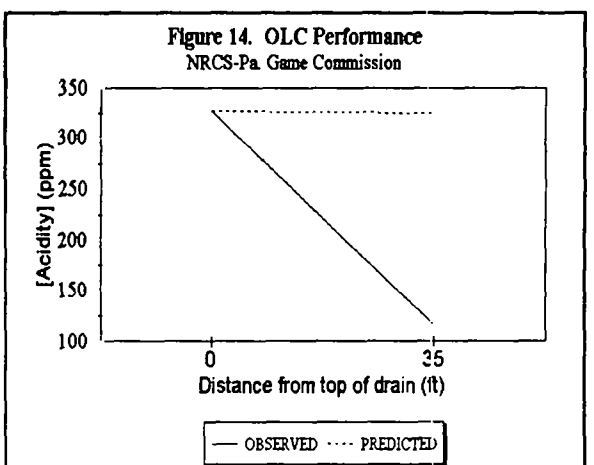

Figure 14. Observed and predicted acidity reduction of an open limestone channel at the PA Game Commission site in Cambria County, PA.

The Cottage Town site has a small amount of acidity entering the channel (Figure 15) and exhibits an acidity removal better than that predicted over the first 110 m (360 ft) of the channel (0.0870% per ft compared to 0.035% per ft predicted). Acidity increases over the last 27 m (90 ft) of the channel probably due to a small source of AMD entering at the base of the channel. But

this brings the overall acidity removal closer to the predicted value (.0290% compared to .0350% per ft of channel).

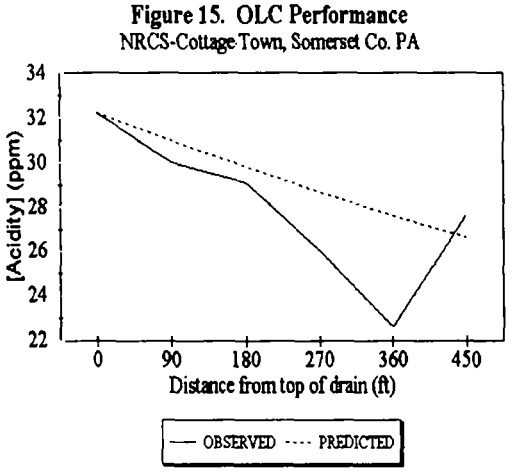

Figure 15. Observed and predicted acidity reductions of an open limestone channel at the NRCS Cottage Town site in Somerset County, PA.

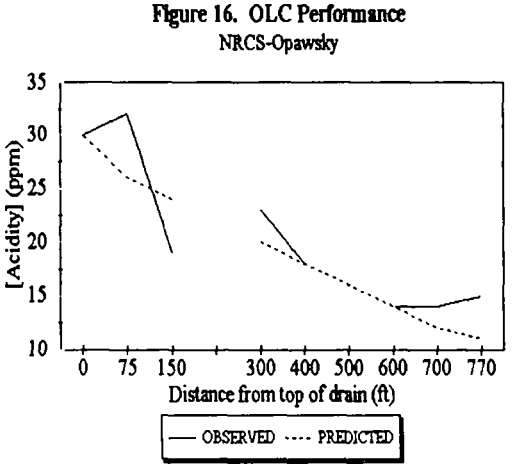

Figure 16. Observed and predicted acidity reduction of an open limestone channel at the NRCS Opawsky site in Armstrong County, PA.

The NRCS Opawsky site's performance was slightly worse than predicted (0.33% acidity removal per ft compared to a predicted removal of 0.42% per ft) but still removed 50% of the acidity (Figure 16).

Discussion and Conclusions

In the laboratory study, ALS treated acidity one-third to one-fourth as fast as ULS. This factor also applies to alkalinity production from these two types of LS. These values are close to the one-fifth factor reported by Pearson and McDonnell (1974). Acidity reduction of OLCs in the field varied between 4% and 62%, and acid reductions per ft of channel were between .029 and 1.77% (Table 2). The steeper channels performed better than the two channels with shallower (9%) slopes. In the sandstone channel, acidity decreased by only 2% and by a factor of .0028% per ft of channel.

The results confirmed the logarithmic acidity decay curve with ALS reported by Pearson and McDonnell (1974). Thus acidity removal by ALS is proportional to the increment of channel length and cross sectional area, regardless of initial acidity. In other words, a fixed proportion of acidity is removed by ALS per ft of channel (width and depth included). Acidity loss is rapid at first then gradually slows down.

OLC's work best on steep slopes. The key factor in designing OLCs is to prevent iron and aluminum flocs from settling out and plugging the LS pores in the channel. One LS channel not reported here was found on a nearly flat slope (1 to 3%). It was filled with floc and was ineffective in treating acidity. The successful channels generally had slopes above 10% and used coarse LS (15-to 30-cm sized material or 6- to 12- in sized material). Both slope and size of LS can maximize void space and water velocity thereby inhibiting floc settling. Evidence of the effect of slope on ALS dissolution is seen on the PA Game Commission and PaDER sites that had LS channels constructed on slopes > 40%.

Each of the passive treatment systems (aerobic wetlands, anaerobic wetlands, ALDs, APS, and OLCs) have an area of application (see Faulkner and Skousen 1995 for descriptions and applications of each). It will be difficult to achieve effluent limits by passive water treatment in most cases by using any one method alone. However, coupling these systems could allow some acidity reduction and metal precipitation with one system, then routing the water into another system for additional acidity and metal removal. The primary application of most passive treatment systems will be on watershed restoration projects, AML sites, and perhaps for pre-treatments for active treatment systems using chemicals. OLCs are particularly useful in steep terrain where long (300 to 1000 m) channels are possible, and they offer a unique treatment where no other passive system is likely to be appropriate. OLCs will produce metal flocs, so settlement basins should be incorporated in the design. Larger OLCs should have settling ponds or wetlands placed at intermediate points (flat channel segments) to remove the precipitates and help prevent plugging.

Table 2. Characteristics and performance of a sandstone open channel and eight OLCs at field sites in Pennsylvania and West Virginia (SS=sandstone, LS=limestone).

Site	Flow	Length	Rock type	Slope	Acidity Initial	Acidity Final	Acid Loss	Rate of Acid loss
	(gpm)	(ft)		(%)	(---- mg/l----	)	(%)	(%/ft)
Coral/Graceton	350	720	SS	10	550	540	2	.0028
Morg Airport E	20	90	LS	20	355	330	7	.0780
Morg Airport W	30	150	LS	14	410	360	12	.0800
Eichleberger #2	100	160	LS	20	510	325	36	.2250
Pa-DER	25	37	LS	60	2600	2500	4	.1080
PA Game Com.	128	35	LS	45	330	125	62	1.7710
Cottage Town	80	450	LS	9	32	28	13	.0290
Opawsky	240	150	LS	9	30	15	50	.3333

The age of the channels we studied varied from 2 to 8 yrs and none of these channels had required maintenance. If constructed correctly, OLCs should be nearly maintenance free and less expensive to construct than other AMD treatment systems.

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AN APATITE DRAIN: NEW METHOD FOR IRON AND ALUMINUM REMOVAL FROM FROM HIGHLY CONTAMINATED ACID MINE DRAINAGE[1]

by

Jung-Chan Choi and Terry R. West[2]

Abstract. The apatite drain system is a new method for acid mine drainage (AMD) treatment. Laboratory tests and the field experiment showed that this technique can be used for removal of high concentrations of iron and aluminum in the AMD with pH less than 4.0. In addition, this system removes both ferric and ferrous irons whereas constructed wetlands system remove only ferric iron when it is converted from ferrous iron in the wetlands. The geochemical model was then used to simulate the changes in concentration of aqueous constituents and in pH, and the potential mineral phase. The results of MINTEQ2 supports the conclusion of experiments. Apatite removed iron, sulfate, and aluminum from AMD and pH was inversely proportional to apatite added until it had reached the equilibrium point (pH=4.09 at 25°C). Nevertheless, the role of apatite as a precipitant and a pH buffer was restricted by the solubility of it.

INTRODUCTION

AMD forms when certain sulfide minerals are exposed to the atmosphere and oxidized in the absence of calcareous material. Although commonly associated with coal mines, the problem also occurs in areas where pyrite or marcasite becomes exposed during land disturbance such as in metallic sulfide mines, highway road cut, subway tunnels, etc.

In the process of AMD forming, sulfide minerals oxidized to form a series of soluble hydrous iron sulfates. Usually, these compound appear as white and yellow crusts on the weathered rocks or sediment surfaces. Natural waters readily dissolve the salts and hydrolyze to form acidic drainage with high concentration of sulfate and ferrous iron. Subsequently, the ferrous iron is oxidized to the ferric state, complexing with water to form a ferric hydroxides (yellow boy) that impart the red and yellow color characteristic of acid mine drainage.

Conventional AMD treatment is a simple process: the water is neutralized, typically to a pH of 8 to 9, and then aerated to oxidize the iron to the ferric state if ferrous iron exists in AMD, causing precipitation of $Fe(OH)_3$ (yellow

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boy sludge). The water is then separated from the sludge in a series of settling basins or ponds and is discharged.

Alkaline reagents used in conventional AMD treatment are lime (CaO), limestone ($CaCO_3$), sodium hydroxide (NaOH), and sodium carbonate (Na_2CO_3). However, these reagents are expensive, potentially dangerous, and when misused can results in the discharge of excessive alkaline water. It is obvious that conventional treatment of run-off and seepage is not a long-term solution, because formation of AMD can persists for many decades.

Staged aerobic constructed wetlands offer a potential low-cost, natural, low maintenance, and long-term alternative to conventional treatment of AMD. In an aerobic wetlands system, oxidation reactions occur and the metals precipitate as oxides and hydroxides. Constructed wetlands contain cattails growing in a clay or spoil substrate. However, plantless systems have also been constructed and function in a similar way to systems containing plants. However, aerobic ponds are not recommended when the water entering the wetlands system has a pH less than 4. At such a low pH, iron oxidation and precipitation reactions are quite slow, and significant removal of iron in the aerobic pond would not be expected (Hedin et al., 1994).

The Anoxic Limestone Drain (ALD), which consists of a shallow, limestone-filled trench excavated into the source material and sealed from the atmosphere, passively introduces buffering capacity, as alkalinity, into the AMD and may comprise the initial component of a staged wetlands system. Changes in pH due to acid production from Fe hydrolysis in the wetlands are buffered due to the high alkalinity in the influent (Brodie, 1991). Dissolution of limestone added 100-200 mg/l of alkalinity as $CaCO_3$ to the water (Kleinmann et al., 1990).

However, not all water is suitable for pretreatment with the ALD. The primary chemical factors believed to limits the utility of the ALD are the presence of ferric iron, aluminum and dissolved oxygen (DO). When acidic water containing any ferric iron or aluminum contact limestone, metal hydroxides will form. No oxygen is necessary. Ferric hydroxide can armor the limestone, limiting its further dissolution. Whether aluminum hydroxides armor the limestone has not been determined. The buildup of both precipitates within the ALD can eventually decrease the drain permeability and cause plugging. The presence of dissolved oxygen in mine water will promote the oxidation of ferrous iron within the ALD, and thus potentially cause amoring and plugging.

APATITE DRAIN SYSTEM

Seeps of some Indiana Abandoned Coal Mine sites show high concentration of ferric iron and aluminum and very low pH because refuse piles have weathered for several decades. Therefore, constructed wetlands system with ALD is not appropriate.

Laboratory tests by the authors showed a high potential for apatite where limestone has failed. Apatite removed iron and aluminum as an metal phosphate at low pH. According to the laboratory test using pebble to sand-sized apatite (50 mm to 0.6 mm) with aerobic condition, apatite removed iron from 100 to 1,200 mg/l and aluminum from 0 to 800 mg/l, and removal was inversely proportional to flow rates. Flow rate ranged between 1.17×10^{-4} to 1.43×10^{-3} cm/sec. In addition, pH increased to values as high as 3.2 in the effluent water from pH of influent water ranging from 2.1 to 2.2 (Choi and West, 1995a).

According to this result, an apatite drain was constructed on September 30, 1994 at the Green Valley Abandoned Mine site near Terre Haute in west central Indiana. The primary objective of this experiment is to evaluate the long-term ability of the apatite drain to mitigate AMD under field conditions. The drain was 9 m long, 3.3 m wide, and 0.75 m deep, and receives AMD seepage from reclaimed gob

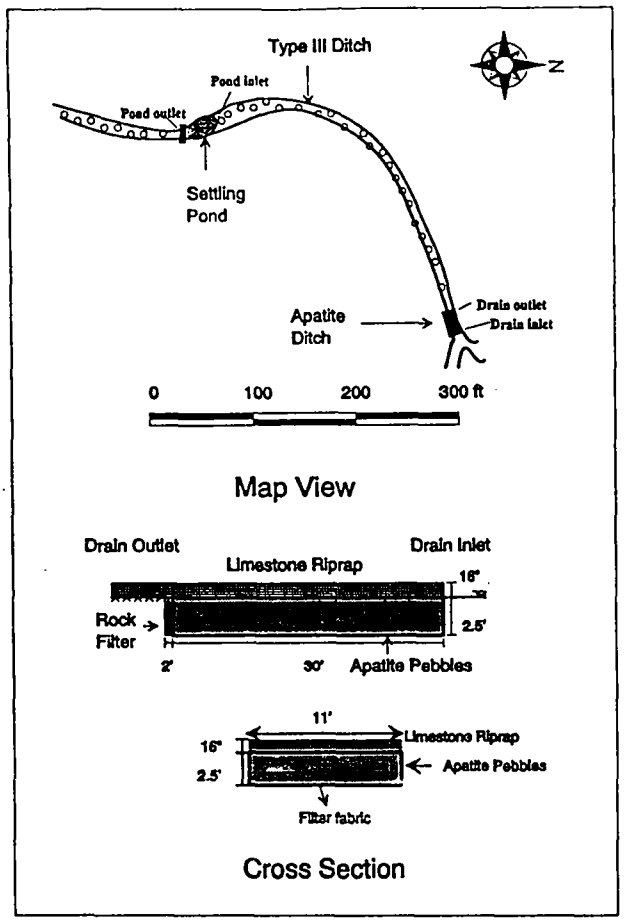

Figure 1. Schematic of an apatite drain design

piles (See Figure 1). The apatite drain was covered with limestone riprap (D_{50}=36 cm) and filter fabric (coefficient of permeability>10^{-3} cm/sec) to protect the drain system from stormwater and siltation. Limestone riprap was also used as a rock filter at the end of the apatite drain. The drain consists of about 50 metric tones of 50 mm to 0.6 mm sized marine apatite (francolite) which came from Payne Creek Phosphate Mine in central Florida. A gabion structure was constructed downstream of the drain to create a settling pond to collect precipitates.

Apatite removed iron from 800 to 4,200 mg/l, aluminum from 120 to 830 mg/l and sulfate from 2,090 to 13,430 mg/l between the section of drain inlet and pond outlet during the nine month monitoring period (See Figure 2).

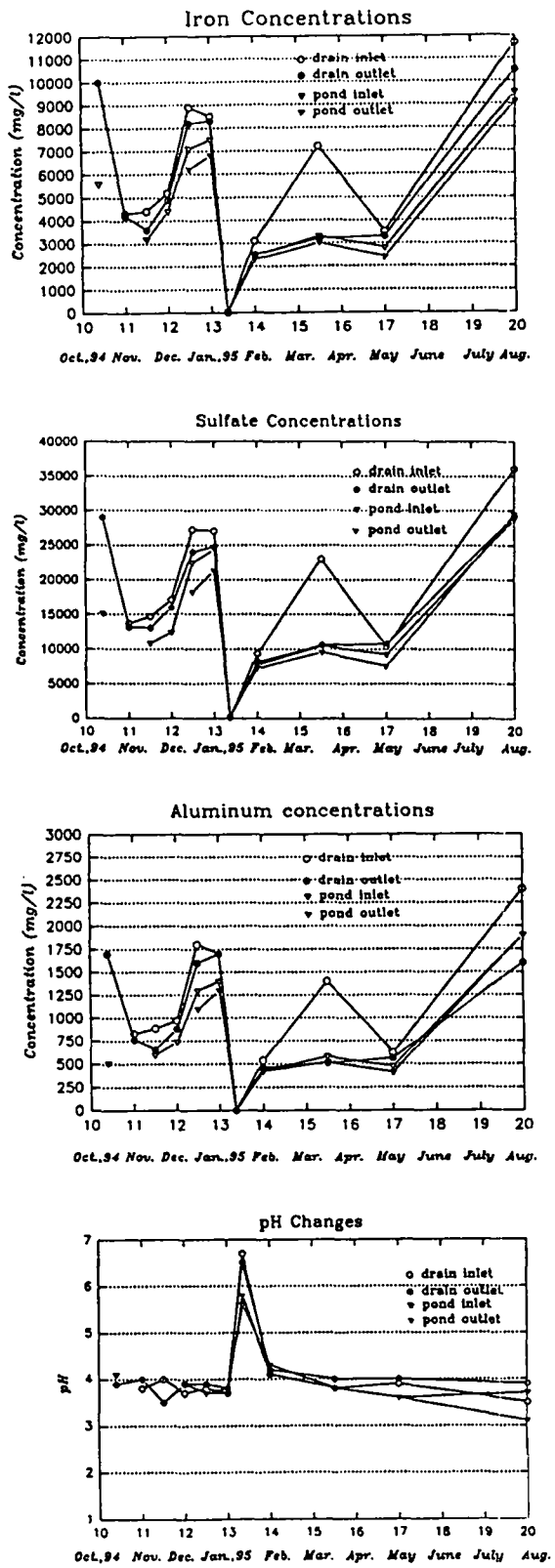

Figure 2. Iron, aluminum and sulfate removals, and pH changes

The pH is nearly constant for the influent and effluent, ranging between 3.1 and 4.3. Flow rate measured at the gabion structure ranged from 0.8 to 1.2 gpm except January 11, 1995 when a stormwater event was occurred (flow rate>100 gpm). Precipitates of iron and aluminum phosphate (yellow and white suspended solids) continued to accumulate in the settling pond (Choi and West, 1995b).

OBJECTIVE OF STUDY

A preliminary question to consider is under what conditions an apatite drain system is effective. The effectiveness of an apatite drain seems to depend on pH, flow rate, and grain size.

According to the X-ray diffraction analysis, only gypsum was detected in the precipitates collected from the water in the settling pond. It seems that remaining precipitates consist of amorphous phosphates. Therefore, it is necessary to define the composition of the precipitates resulted from reactions between AMD and apatite.

To solve these problems, the MINTEQA2 computer program was used to simulate the experimental changes in concentration of aqueous constituent in the acid mine water. MINTEQA2 is a geochemical equilibrium speciation model capable of computing equilibrium among the dissolved, adsorbed, solid, and gas phases in an environmental setting (Allison et al., 1991). The model can be used to calculate the equilibrium composition of dilute aqueous solutions in the laboratory or in natural aqueous systems. It can be used to calculate the mass distribution between the dissolved, adsorbed, and multiple solid phases under a variety of conditions including a gas phase with constant partial pressure.

The objectives of this study is to apply MINTEQA2 to the attenuation problem of AMD, to predict the outcome solution chemistry, to compare it with the experimental data, and to find out chemical processes responsible for the chemical composition of actual experiments.

GEOCHEMICAL MODELING

The apatite added to treat AMD was assumed to have homogeneous, ideal chemical composition, which can be represented by hydrapatite ($Ca_5(PO_4)_3OH$). The elements of interest in this simulation were iron, aluminum, manganese and sulfate, which react mostly with three constituents, that is, calcium, phosphate and H_2O. Francolite was not considered because it has four more elements added to the system (See Table 1) and the iteration number to reach convergence may exceed the 200 maximum allowed by MINTEQA2 which could cause error of the iteration number. The chemical characteristics for the untreated water

Table 1. Chemical analyses of phosphate pebble (+1 mm) in weight percent (from Cathcart, 1989).

$Ca_{10-X-Y}Na_XMg_Y(PO_4)_{6-Z}(CO_3)_ZF_{0.4Z}F_2$	
P_2O_5	30.7-36.1
CaO	40.8-49.0
F	3.63-4.01
SiO_2	6.33-10.4
Al_2O_3	0.57-3.77
Fe_2O_3	0.55-1.38
MgO	0.14-0.35
Na_2O	0.11-0.57
K_2O	0.11-0.57
Ti_2O_8	0.04-0.06
MnO	0.01-0.03
S	0.10-0.52
CO_2	1.83-3.80
Ratios	
CaO/P_2O_5	1.40
F/P_2O_5	0.133

Table 2. Chemical data from laboratory testing to evaluate the effectiveness of apatite for AMD control using the seepage water collected from Seep 108 located at the toe of the southwestern edge of the Southeast Gob Pile at the Friar Tuck Abandoned Mine site near Dugger, Indiana (from Choi and West, 1995a).

Water Sample	pH	Eh	Alk. (eq.	Acid. $CaCO_3$)	SO_4^{2-} (	Ca^{2+}	Mg^{2+} mg/l	K^+	Na^+	Zn^{2+})
August 29, 1991 - September 4, 1991										
Seep 108A	2.1	386	0	31200	36200	510	1300	28	50	100
Bottle A	2.4	333	0	29500	34100	620	1300	27	65	100
Bottle B	2.9	250	0	27400	33900	650	1300	27	90	100
Bottle C	3.0	237	0	25700	31300	610	1300	29	93	100
Bottle D	3.2	222	0	23900	29100	560	1500	31	96	100

Water Sample	Fe^{2+} (	Fe^{3+}	Al^{3+} mg/l	Mn_{tot}	Si	TDS)	Charge Balance	Flow rate (l/min/kg)
August 29, 1991 - September 4, 1991								
Seep 108A	4600	800	3600	250	72	47600	0.2	
Bottle A	4500	300	3400	240	69	44700	-0.1	9.52×10^{-4}
Bottle B	4500	400	3500	250	73	44800	1.5	5.35×10^{-4}
Bottle C	4400	300	3200	250	75	41800	3.1	2.39×10^{-4}
Bottle D	4300	400	3000	250	77	39500	5.4	1.21×10^{-4}

were obtained from Choi and West (1995a) and are shown in Table 2. Precipitation was allowed but no oxidation/reduction reaction was permitted. The pH was allowed to change in response to the dissolution of hydraptite and precipitation of other constituents, whereas the Eh was ignored in the program runs. The hydraptite was considered as an infinite solid for the first run to calculate the maximum amount of hydrapatite soluble in the solution with different water temperatures. Adsorption was not allowed in these runs because the data for the adsorption model such as a particular metal oxide and its surface area and sorption site concentration, could not be obtained from the experimental data. The trace elements which have a strong tendency to adsorb on ferric hydroxide were not considered in this geochemical modeling, either.

In order to simulate the pH variation with respect to the amount of apatite added and to find the reduction of metals and sulfate in the system, another run was executed with stepwise increasing apatite amount added at initial pH. Additionally, an excution was performed to trace the sequence of mineral precipitation.

RESULTS AND DISCUSSION

Speciation calculations using MINTEQA2 show that the dominant free ions and complexes for Al, Fe, and Mn include $AlSO_4^+$ and $Al(SO_4)^{2-}$, Fe^{2+}, $FeSO_4$, $FeSO_4^+$ and $Fe(SO_4)_2^-$, and Mn^{2+}, and $MnSO_4$, respectively.

When the water temperature was set at 25°C, most of the dissociation of apatite occurred at pH below 3.8. When 0.04 mol of apatite was added to the solution, the pH of equilibrium solution was 3.8. Beyond this point, the increasing rate of pH was conspicuously reduced, which may be closely related to the solubility of apatite. The maximum dissolvable amount of apatite with Seep 108A was 0.082 mol/l at pH 4.09, beyond which pH was constant and surplus of hydrapatite was not dissolved.

When the water temperature was set as 4°C, The maximum amount of apatite dissolvable in the solution was 0.046 mol/l at pH 4.62. The

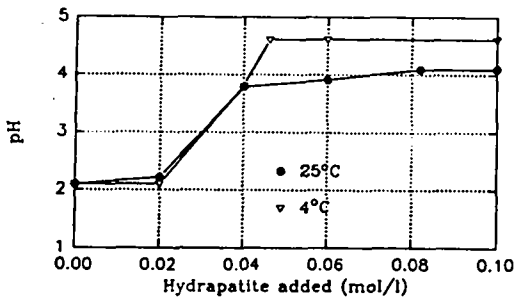

Figure 3. Simulation of MINTEQA2 on variation of pH as a function of amount of apatite added into solution and temperature

pH as a function of the added amount of apatite and temperature is shown in Figure 3.

When precipitation was allowed to occur, the possible minerals precipitated are listed in Table 3. Precipitation of manganese phosphate consumed H^+. The dissolution of hydraptite at low pH also removed H^+ from solution, which is supported by dissolved species of phosphate such as $FeH_2PO_4^+$, $H_2PO_4^-$, and $MgH_2PO_4^+$.

The solubility of apatite was closely related with the characteristics of its own chemical composition and chemistry of the solution. Especially, the solubility of hydrapatite is a function of Ca^{2+} activity in the solution (Wilson and Ellis, 1984). The Ca^{2+} is a more reactive ion component than PO_4^{3-} so that the solubility of hydraptite is controlled by the activity of Ca^{2+} ion.

Fe^{3+} and Mn^{2+} were quickly removed as soon as hydrapatite was added. Once apatite was added, the first minerals possibly precipitated were strengite and $MnHPO_4$, which are iron phosphate and manganese phosphate, respectively. Strengite ($FePO_{4(s)}$) and variscite ($AlPO_{4(s)}$) are the stable solid phase when the phosphate is precipitated in the low pH range. However, the pH of minimum variscite solubility occurs at about 1 pH unit higher than that of strengite, so that the precipitation of variscite did not occur.

The modeled concentration of dissolved manganese as a function of PO_4^{3-} does not correlate very well with the actual analysis data (See Table 2). According to the data of simulation, manganese ion was completely precipitated as a mineral of manganese phosphate. However, the actual concentration of manganese apparently did not decrease and no precipitation was observed. Thise behavior of manganese an be explained in several ways. Differences between the two results can be accounted for by the assumption that the total concentration of Mn in the experimental data would consists of Mn^{2+} only. Secondly, microcrystalline manganese phosphate might be analyzed as a dissolved phase. This kind of error can be eliminated by filtering and centrifuging the solution before analysis.

Table 3. Potential minerals precipitated at equilibrium

PHOSPHATE	
Hydrapatite	$Ca_5(PO_4)_3OH$
Strengite	$FePO_4\ 2H_2O$
Vivianate	$Fe_3(PO_4)_2\ 8H_2O$
MnHPO4	
OXIDE	
Hematite	Fe_2O_3
Ca-Nontronite	$Ca_{0.167}Fe_2(Si_{3.67}Al_{0.33})O_{10}(OH)_2$
Diaspore	AlO(OH)
SULFATE	
Gypsum	$CaSO_4\ 2H_2O$
Alunite	$K\ Al_3(SO_4)_2(OH)_6$
$AlOHSO_4$	

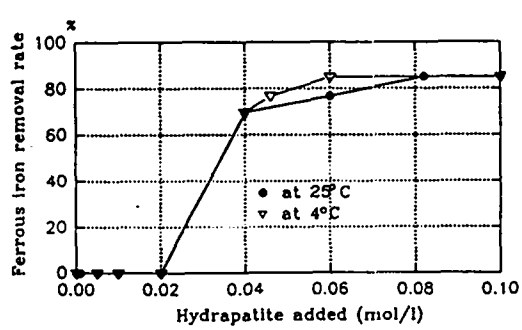

Figure 4. Ferrous iron removal rate as a function of amount of apatite added into solution.

Additionally, manganese ion can be formed in oxyhydroxides at low temperature. Therefore, it might exist in a state of flocculation in the solution and could be analyzed along with dissolved phase. Ferrous iron is a dominant cation in the acid mine drainage. Its chemical reactions have a significant effect on themobility of other solutes. Exposure to atmospheric O_2 results in oxidation of much of ferrous state to ferric state. However, the rate of conversion of ferrous to ferric iron in the aerated solutions is very slow below pH 4.0. The precipitation of ferrous iron depends on the concentration of phosphate in the solution and on the solubility of hydrapatite at low pH (See Figure 4). Therefore, the solubility of apatite is the key factor as a reagent for the treatment of AMD.

The aluminum concentration of the simulation data were clearly related to the pH control imposed on the system by the apatite solution. Nordstorm (1982) suggested that an aluminum sulfate ($(Al_2SO_4)_3$) solid may control aluminum solubility in some acid sulfate-rich waters. The minerals $AlOHSO_4$, Diaspore (AlOOH), and alunite ($KAl_3(SO_4)_2(OH)_6$) are all predicted to be supersaturated by MINTEQA2.

Lindsay (1979) mentioned that the solubility of SO_4^{2-} in soil is limited by the solubility of gypsum. The Ca^{2+}, together with high SO_4^{2-} concentrations from pyrite oxidation, yields in a saturated condition with respect to gypsum. The simulation results show that gypsum and alunite as well as $AlOHSO_4$ were precipitated out of solution.

Another run of MINTEQA2 has been performed to trace the sequence of mineral precipitation (See Table 4). On the basis of the output of this run, the sequence of mineral precipitation and dissolution could be traced step by step. In case of ferric iron, first mineral speciated was strengite During this excution. However, when pH was increased, the phosphate from strengite combined with ferrous iron and vivianite was precipitated. On the other hand ferric iron was precipitated as hematite and/or Ca-nontronite. The sequence

Table4. Variation of specific elements and mineral assemblages (0.06 mol/l hydrapatite added)

STEP	ELEMENT (% precipitated)					(not dissolved, mmol/kg)
	Fe^{3+}	Fe^{2+}	Al^{3+}	Mn^{2+}	$SO4^{2-}$	Hydrapatite
1	0.0	0.0	0.0	0.0	0.0	0.0549
2	100.0	0.0	0.0	0.0	0.0	0.0533
3	100.0	0.0	0.0	0.0	0.0	0.0533
4	100.0	0.0	94.0	0.0	0.0	0.0524
5	100.0	0.0	90.1	100.0	0.0	-
6	100.0	0.0	90.3	100.0	0.0	-
7	100.0	0.0	99.9	100.0	77.1	-
8	100.0	0.0	99.8	100.0	82.4	-
9	100.0	79.3	99.8	100.0	92.1	-
10	100.0	79.0	99.8	100.0	92.2	-
11	100.0	78.9	99.8	100.0	92.2	-
12	100.0	64.5	99.6	100.0	95.3	-
13	100.0	76.7	99.7	100.0	93.0	-

STEP	MINERAL PRECIPITATED (mmol/kg)								
	Strengite	Ca-nontronite	Hematite	Vivianite	Alunite	Diaspore	$AlOHSO_4$	$MnHPO_4$	Gypsum
1	0.0000	-	-	-	-	-	-	-	-
2	0.0150	-	-	-	-	-	-	-	-
3	0.0150	-	-	-	-	-	-	0.0048	-
4	0.0150	-	-	-	-	-	-	0.0048	-
5	0.0150	-	-	-	-	0.1317	-	0.0048	-
6	0.0150	-	-	-	-	0.1262	-	0.0048	-
7	0.0150	-	-	-	-	0.1265	-	0.0048	0.3149
8	0.0150	-	-	-	-	0.1265	0.0214	0.0048	0.3045
9	0.0150	-	-	0.0229	-	0.1185	0.0666	0.0048	0.2987
10	0.0136	0.0007	-	0.0228	-	0.0723	0.0673	0.0478	0.2974
11	0.0136	0.0007	-	0.0228	-	0.0712	0.0661	0.0478	0.2974
12	-	0.0007	0.0385	0.0186	0.0007	0.0416	0.0953	0.0478	0.2804
13	-	0.0007	0.0068	0.0221	0.0007	0.0658	0.0714	0.0478	0.2950

of aluminum precipitation started with diaspore and ended up with $AlHSO_4$ and alunite. The run of MINTEQA2 shows the precipitation sequence with time, which can be used to predict the mineral assemblage that is a product of the reaction of hydrapatite with AMD.

CONCLUSION

The apatite drain system is a new method for AMD treatment. A naturally occurred material was thought which will remove metals from AMD, produce insoluble and powdery precipitates, have buffering capacity or at the least maintain the pH after reaction, and dissolve slowly so that it lasts several years or several decades. Apatite from Florida satisfied these conditions.

Laboratory tests and the field experiment showed that this technique can be used for removal of high concentrations of iron and aluminum in AMD. In addition, this system removes both ferric and ferrous irons whereas constructed wetlands systems remove only ferric iron when it is converted from ferrous state in the wetlands. However, only gypsum was detected in the precipitates collected from the settling pond according to the X-ray diffraction analysis. Therefore, it seems that precipitates consist of amorphous phosphate.

A geochemical speciation model (MINTEQA2) was applied to simulate the precipitation of mineral phase, the solubility of apatite, pH variance with the apatite dissolution and the sequence of mineral precipitations. Hydrápatite, which is one of simple form of apatite and also has buffering capacity similar to francolite, was used for this simulation. The main conclusions are as follows:

- H^+ was neutralized primarily with phosphate mineral. These reactions were relatively rapid and resulted in a steep pH gradient over a narrow range of phosphate addition.
- The maximum amount of hydrapatite dissolvable in the solution was 0.082 mol/l at pH 4.09 and 25°C, whereas it was 0.046 mol/l at pH 4.62 and 4°C.
- Ferric iron was quickly removed as strengite at low pH and was transformed into hematite and Ca-nontronite as pH rose.
- The concentration of ferrous iron in the treated AMD was controlled by precipitation of vivianite (iron phosphate). The oxidation of ferrous to ferric iron was apparently limited by low pH.
- It is likely that concentration of Al was controlled by the solubility of $AlOHSO_4$ at low pH. Diaspore and alunite were other potential solid phases at low pH.
- Manganese might be formed by the precipitation of manganese phosphate (MnHPO4), which is the most stable Mn solid phases.
- Calcium and sulfate concentrations were controlled by the precipitation of gypsum.

The results of MINTEQA2 runs appeared to support the conclusion of the experimental works. Apatite is effective in removing iron, sulfate, and aluminum from AMD and pH was inversely proportional to flow rates until it had reached the equilibrium point (pH = 4.09 at 25°C).

However, further research on the apatite drain system is necessary to define the actual mechanism of the precipitation process and the constituents of those precipitates in order to optimize design of the apatite drain system.

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ASSESSMENT OF THE APPLICABILITY OF AN ANOXIC LIMESTONE DRAIN FOR A SURFACE MINE IN EAST CENTRAL TENNESSEE[1]

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Abstract: Anoxic limestone drains (ALDs) are a cost-effective technique for adding alkalinity to acid mine drainage. However, the applicability of an ALD is limited to a rather narrow range of mine drainage chemical conditions due to concerns about the armoring of limestone with ferric hydroxide, the plugging of flow paths with aluminum hydroxide, and the limited solubility of calcite. While the armoring and plugging potentials can be assessed with careful water quality analyses, the solubility of limestone in a particular mine water cannot, at this time, be predicted from mine water chemistry. Thus, the danger always exists that the ALD will generate insufficient alkalinity to completely neutralize the acidic water, resulting in either insufficiently treated discharge or a need for additional treatment. In order to remove uncertainty from the design of a 4,000-ton ALD, we conducted limestone incubation tests and pilot-scale ALD tests. Incubation tests were done using a modified version of the "cubitainer" procedure developed by the United States Bureau of Mines. The pilot ALD consisted of 65 tons of limestone. Hydrologic loading experiments were conducted that provided an assessment of the ALD performance under design flow conditions and flow rates four times higher than the design flow. Under design flow conditions, the pilot ALD discharged water with alkalinity concentrations similar to that predicted by the cubitainer tests (360 parts per million). With increased flow, concentrations of alkalinity in the pilot ALD effluent decreased, with the water becoming net acidic at a flow rate of 15 gallons per minute. The results of the pilot ALD were used to size a full-scale ALD and model ALD performance under a variety of flow conditions.

Additional Key Words: cubitainer, passive alkaline addition.

Introduction

Anoxic limestone Drains (ALDs) have made the passive treatment of acid mine drainage possible on hundreds of mine sites in the Appalachian coal region. The theory of ALDs is quite simple. Acidic water is directed through a buried bed of limestone gravel (Turner and McCoy, 1990). Limestone dissolution raises pH and adds bicarbonate alkalinity to the water, which promotes metals precipitation in subsequent ponds or wetlands. Anoxic conditions within the limestone bed prevent iron oxidation and limestone armoring. In some cases, it is economically feasible to construct ALDs which

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may last decades due to the low cost and limited solubility of limestone.

However, ALD technology is not without shortcomings. The use of ALDs is not suitable under certain water chemistry conditions. Ferric iron and aluminum both precipitate within an ALD, decreasing performance by armoring limestone and plugging flow paths (Hedin, et.al.). It is also possible that a properly designed and constructed ALD will not generate sufficient alkalinity to completely neutralize acidic water due to limited limestone solubility. Water chemistry which appears similar based on evaluation of standard acid mine drainage analytes may generate dissimilar alkalinity concentrations.

This paper describes the design process of a large ALD system (4,000 tons of limestone). Because of the financial commitment involved in the construction of a 4,000-ton ALD system, it was considered prudent to thoroughly evaluate the ALD concept prior to construction. This paper describes our use of cubitainer tests and a small "test" ALD to provide data.

The research was conducted at Sequatchie Valley Coal Corporation (SVC). SVC operated area surface coal mines located in east central Tennessee. One surface mine, locally referred to as Area 1, was mined from 1978 to 1982. Backfilling, regrading, revegetation, and related reclamation activities were largely completed by 1983. However, acidic seeps began to develop at topographic lows after 1983. In the period that followed, SVC collected acidic water and used standard chemical treatment and precipitation methods. Realizing that chemical treatment may not be in the best interest of SVC or the environment, SVC considered the applicability of constructed wetlands and related passive technologies (Hedin and Massey, 1995). SVC implemented an approach consisting of short term management and development of long term solutions to water quality problems.

Evaluation of groundwater quality data at Area 1 revealed good news, but also raised some questions. Positive results included the absence of aluminum and a pH of 5.5 standard units. At a pH of approximately 5.5 standard units, no ferric iron should exist (Stumm and Morgan, 1970). Therefore, armoring of limestone with iron hydroxides and flow path clogging by aluminum precipitates would not be anticipated in an ALD constructed at Area 1. However, acidity concentrations were measured above 300 ppm. ALDs have been properly sized and constructed which generate less than 200 ppm alkalinity (Hedin, et.al., 1994), which would be insufficient to completely neutralize acidity at Area 1. Therefore, as part of the evaluation of ALD applicability at Area 1, SVC desired a knowledge of the amount of alkalinity likely to be produced by an ALD. Cubitainer testing (Watzlaf and Hedin, 1993) was subsequently conducted to estimate alkalinity likely to be generation by an ALD.

Cubitainer Testing

The United States Bureau of Mines (USBM) presented an ALD alkalinity prediction method in April 1993 at the *Thirteenth Annual West Virginia Surface Mine Drainage Task Force Symposium*. The method involves placing limestone and untreated mine water in collapsible containers (cubitainers) and monitoring the alkalinity generation by periodic extraction of water. The method was developed by the USBM during studies of two ALDs with significantly different alkalinity generation characteristics. The USBM studies determined that mine water chemistry, not limestone quality, determined the alkalinity generation characteristics of an ALD. In addition, the researchers noted that the 48 hour alkalinity concentrations predicted by the cubitainer testing method were within six percent of the actual alkalinity generated by the ALDs (Watzlaf and Hedin, 1993). Prior to this research, a method for predicting ALD alkalinity generation did not exist.

In 1993, SVC engaged Skelly and Loy to assist in the evaluation of passive alternatives to chemical water treatment at Area 1. As part of the assignment, the USBM cubitainer procedure was utilized to estimate the alkalinity which may be generated by an ALD at Area 1. Specifics of the cubitainer method follow.

Collapsible, one-cubic foot cubitainers and smaller one-gallon cubitainers were used for the testing procedure. One-cubic foot containers provided adequate water to conduct a series of analytical and biological tests after the cubitainer test was

completed. The cubitainers were equipped with two-holed rubber stoppers and lids. Gas impermeable "Nalgene" tubing was then placed through each stopper hole. To form an airtight seal, silicon sealant was placed around the stopper holes and tubing. Two pieces of tubing were used. One was cut to sufficient length to extend to the bottom of the cubitainer while the other was inserted flush with the bottom of the stopper. Clamps were then attached to the tubing. An on/off clamp was attached to the longer tube (used for filling) and an adjustable clamp was attached to the shorter tube (used for expelling oxygen, nitrogen, and liquid samples). By utilizing the adjustable clamp, flow rates into and out of the cubitainers were controlled to avoid introducing oxygen into the container.

Each cubitainer was filled approximately ⅔ full (20 Kg) of 1½"x¾" limestone. The limestone was obtained locally from the Monteagle Formation. Prior to being placed into the cubitainer, the limestone was thoroughly rinsed to remove all fine particles of calcium carbonate. Rinsing was completed in a two-step process. The first rinse was completed using water from the SVC office facility well. Once the limestone was rinsed of fine particles of calcium carbonate on all surfaces, the sample was rinsed with deionized water. This second rinse was employed to completely remove residue (if present) from the well water. Limestone was then placed on a clean tarp and left to dry completely. Once dry, the limestone was introduced into the cubitainers preparing them to be filled with the water sample.

Utilizing the dewatering well field in-place at Area 1, the wells were purged with a minimum of three well volumes prior to the collection of any water samples. Background water quality parameters for pH, dissolved oxygen, and temperature were taken initially before any well purging began. Once wells were purged, readings were again taken to ensure that dissolved oxygen levels remained low, generally less than one part per million (ppm).

The cubitainer filling process then proceeded following the prescribed methods used by the USBM. In addition to using larger cubitainers, one procedural enhancement was employed. Prior to filling each cubitainer with water, each cube was filled with nitrogen. The nitrogen was pumped into each cube to remove oxygen. Through the use of an adjustable clamp, the cube was then collapsed, forcing the nitrogen and remaining oxygen out of the cube. The clamp was fully closed and the cube was refilled with nitrogen. This process was conducted three times, encouraging the replacement of oxygen with nitrogen. Once completed, the cubitainer was filled with sample water.

As another means to minimize oxygen introduction to the cubitainers, water obtained from the sample well was collected using an ISCO Model No. 3710 portable sampler equipped with a peristaltic type pump. Multiple water samples were collected from the well at a ten-foot depth. Continuous dissolved oxygen readings were monitored at this depth. Water flow rates and nitrogen release from the cubitainer were then controlled using the clamps. Each of the cubitainers was filled through the tubing which extended to the bottom of the cube. Water was introduced while forcing the nitrogen from the cubitainer. The filling procedure continued until no visible air space remained within the cubitainer.

Once each cubitainer was filled, the tubes were clamped shut. The cubitainers were then placed into a trough filled with continuously circulating water originating from the well field. The purpose of the filled trough water bath was to keep the filled cubitainers at a constant temperature (approximately equivalent to the groundwater temperature). This procedure also served to keep the cubitainers in an anoxic state. Water was pumped from the dewatering wells into the trough for the duration of the 48-hour test period.

Cubitainers remained in the water bath for a period of 48 hours. After a period of 48 hours, the one-cubic foot cubitainers were taken out of the water bath and retained for subsequent aeration and settling, analytical testing, and biological testing. The one-gallon cubitainers were kept in temperature controlled coolers for a period of one week. Samples of cubitainer water were collected after 4, 8, 24, 48, 72, 96, and 120 hours. Alkalinity and pH readings were made on each of these samples.

Cubitainer Results

Evaluation of the alkalinity results revealed that the alkalinity increased from approximately 140 mg/l to 340 mg/l in the first 48 hours of the cubitainer test performed on the larger containers. Figure 1 depicts time versus alkalinity for the first five days of the smaller scale, longer duration, one-gallon cubitainer test which confirmed results from the larger cubitainers. This figure indicates that near maximum alkalinity was achieved in the first 48 hours of the test period. Therefore, if past research conducted by the USBM is validated by this cubitainer testing, it would be expected that a properly sized and constructed ALD at Area 1 would generate alkalinity of a similar magnitude. These results indicated favorable conditions existed for utilization of an ALD at Area 1.

Pilot-Scale ALD Testing

The cubitainer test provided confidence on the applicability of an ALD solution for Area 1. As plans for the full scale ALD evolved, questions regarding ALD performance under variable flow conditions were raised. To address this concern, a small ALD was designed that would allow testing of alkalinity generated under variable flow regimes.

The 65-ton test ALD was constructed in January 1995. A rectangular pit 60 feet long, six feet wide, and four feet deep was developed to contain the gravel sized limestone. The ALD limestone was wrapped in filter fabric and plastic and buried under two feet of spoil. The plastic may inhibit water losses or gains while the spoil is intended to ensure anoxic conditions existed within the ALD. The source water was obtained from a nearby groundwater pumping system. In order to test the ALD under a variety of flow conditions, a control valve was placed in the test ALD influent line to allow for flow rate adjustment. The exit pipe was placed above the top elevation of the ALD to ensure that the limestone would remain completely under water.

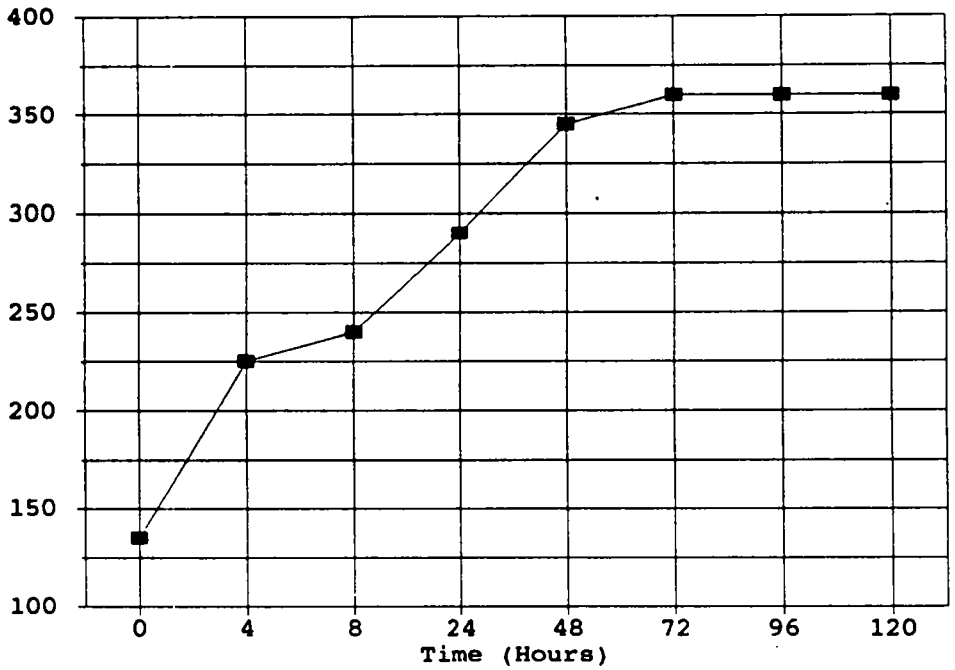

Figure 1. Cubitainer Test Contact Time Versus Alkalinity

The ALD was operated between February and May 1995. The test ALD was used to determine alkalinity generation over a range of flow conditions. In order to obtain the maximum alkalinity concentration, an ALD should contain at least 12 tons of limestone for each gallon per minute (gpm) of flow (Hedin et. al., 1994). Thus, the base flow rate used for the test ALD was approximately five gpm.

Figure 2 graphically depicts flow rate versus alkalinity concentrations for flow rates which approximate the base flow rate (three to eight gpm). These data exclude flow rate adjustment tests described later in this paper. After initial system flushing, water exiting the test ALD contained alkalinity concentrations of approximately 360 ppm. After a few months of operation, the alkalinity stabilized at approximately 320 ppm.

Figure 3 depicts flow rate versus alkalinity concentrations during flow rate adjustment tests in the ALD. As expected, alkalinity generation decreased as flow increased. The flow was increased up to 20 gpm (approximately 3 tons of limestone per gpm of flow). This flow is equivalent in tons of limestone per gpm to a flow much greater than the maximum expected flow to the ALD. Therefore, the pilot-scale test covered the full range of conditions anticipated in the proposed full scale ALD. At a flow rate of 15 gpm, alkalinity concentrations had been reduced to approximately 265 ppm and the water exiting the test ALD became net acidic.

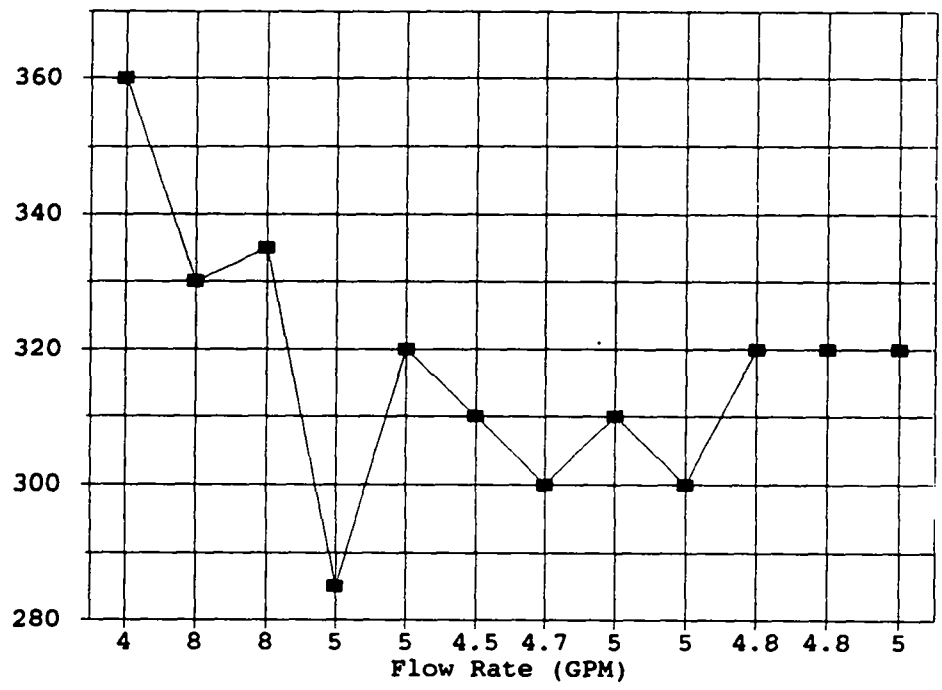

Figure 2. Pilot ALD Flow Rate Versus Alkalinity

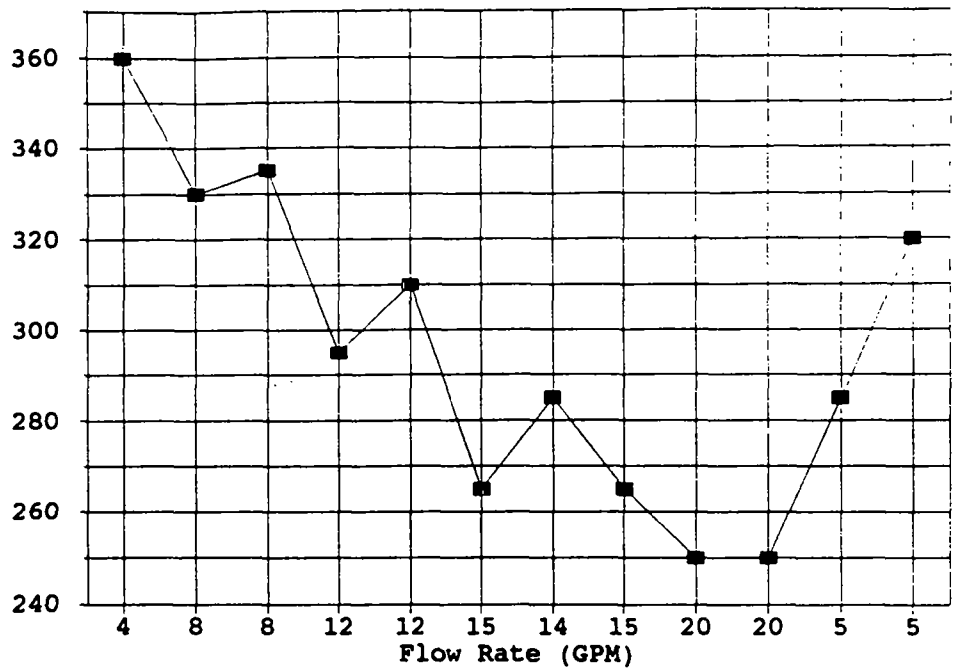

Figure 3. Pilot ALD Flow Rate Versus Alkalinity Flow Rate Adjustment Test

During maximum flow conditions expected in the full scale system, the limestone to flow ratio is anticipated to be eight. At a comparable flow rate in the test ALD (eight gpm), alkalinity concentrations exceeded 300 ppm. Using this result as a prediction tool, net alkalinity would be expected even during maximum flow conditions.

The last two months of test ALD data collection was evaluated to determine potential variation of alkalinity generation when flow remained relatively constant. The alkalinity generation was remarkably constant in the range of 310 to 320 ppm. These data are depicted by the last eight readings of Figure 2.

Conclusions

Due to the inability to determine alkalinity generation in an ALD based on water chemistry alone, other techniques must be employed. Two available methods are cubitainer testing and pilot-scale ALD testing. As documented here, both methods provided similar results under recommended ALD sizing criteria. In addition, a pilot-scale system can be used to simulate extreme flow conditions which may be anticipated. The pilot-scale test can also be used to estimate the flow rate at which the ALD effluent would become net acidic. At that point, the ALD alone would generate insufficient alkalinity to completely neutralize the acidic water.

The methods presented in this paper formed the design basis for a 4,000-ton full scale ALD which has been installed at SVC's Area 1 near the end of 1995. Initial results from the full scale ALD are remarkably similar to both the cubitainer test and pilot-scale ALD test. The first discharges from the full-scale ALD to the receiving streams have alkalinity concentrations of approximately 340 ppm. This result validates the usefulness of the techniques described in this paper.

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SESSION 7: SOILS AND OVERBURDEN

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CONSTRUCTING VEGETATION PRODUCTIVITY EQUATIONS BY EMPLOYING UNDISTURBED SOILS DATA: AN OLIVER COUNTY, NORTH DAKOTA CASE STUDY[1]

by:

Jon Bryan Burley[2], Kenneth J. Polakowski[3], and Gary Fowler[3]

Abstract. Surface mine reclamation specialists have been searching for predictive methods to assess the capability of disturbed soils to support vegetation growth. We conducted a study to develop a vegetation productivity equation for reclaiming surface mines in Oliver County, North Dakota, thereby allowing investigators to quantitatively determine the plant growth potential of a reclaimed soil. The study examined the predictive modeling potential for both agronomic crops and woody plants, including: wheat (*Triticum aestivum* L.), barley (*Hordeum vulgare* L.), oat (*Avena sativa* L.), corn (*Zea mays* L.), grass and legume mixtures, Eastern red cedar (*Juniperus virginiana* L.), Black Hills spruce (*Picea glauca* var. *densata* Bailey), Colorado spruce (*Picea pungens* Engelm.), ponderosa pine (*Pinus ponderosa* var. *scopulorum* Engelm.), green ash (*Fraxinus pennsylvanica* Marsh.), Eastern cottonwood *Populus deltoides* Bart. ex Marsh.), Siberian elm (*Ulmus pumila* L.), Siberian peashrub (*Caragana arborescens* Lam.), American plum (*Prunus americana* Marsh.), and chokecherry (*Prunus virginiana* L.). An equation was developed which is highly significant ($p<0.0001$), explaining 81.08% of the variance (coefficient of multiple determination=0.8108), with all regressors significant ($p\leq0.048$, Type II Sums of Squares). The measurement of seven soil parameters are required to predict soil vegetation productivity: percent slope, available water holding capacity, percent rock fragments, topographic position, electrical conductivity, pH, and percent organic matter. While the equation was developed from data on undisturbed soils, the equation's predictions were positively correlated (0.71424, $p\leq0.0203$) with a small data set (n=10) from reclaimed soils.

Additional Key Words: landscape planning, soil science, prime farmland reclamation, agroecology

Introduction

Reclamation research has led to the formative development of empirical prediction models to forecast the suitability of reconstructed soils (neo-sols) to support plant growth. This approach can aid in creating post-disturbance landscapes usable for agriculture, forested lands, transportation right-of-ways, naturalized vegetation associations, and urban vegetation applications. These equations may render the current time consuming and expensive reclamation assessment methods such as reference evaluation procedures to become obsolete (Doll and Wollenhaupt 1985). This paper employs Oliver County, North Dakota as a case study to investigate some of these important landscape vegetation productivity research issues.

Investigators have pursued several different methodologies to generate vegetation productivity predictive models based upon soil characteristics (Burley 1995a). Dunker, *et al.* (1992) present a recent overview and findings associated with numerous predictive efforts. One methodology primarily developed in the United States is a technique we call the "heuristic sufficiency approach" which builds pseudo empirical models to predict plant growth. The problem with this approach is that the variables in the equation may be redundant and the equation may be over specific. The major problem with the sufficiency approach is it is non-statistical. Another approach is the "reclaimed soil approach" which has been pursued quite intensively by some investigators, especially in Illinois and Kentucky. This approach is certainly empirical and

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statistical; however, to build a model with strong external validity will require the detailed assessment of plant growth across hundreds of reclaimed soils, an expensive and time consuming approach. In the long term this model building approach based upon reclaimed soils is probably the most scientifically sound in methodology; however, the data base to build such models will require substantial research expenditures and currently, no research team has built such an extensive data set. At best most researchers have built reclaimed soil models from several soil types for several crop types. In contrast, data gathered from all crop types grown on all soil types will be necessary to build strong externally valid models. Nevertheless, one federal agency, the Natural Resources Conservation Service, has been spending approximately 1 million dollars per county to construct extensive data sets from primarily un-mined landscapes, recording soil characteristics including the growth of numerous vegetation types across all of the soils in a specific county. These data sets could form the basis for building predictive models, a "soil survey approach." A review of the literature and formative ideas associated with Soil Conservation Service soil survey data modeling approach can be found in Burley (1992). The methodological foundations for this approach were described by Burley and Thomsen (1987). Currently, several soil survey based productivity equations have been generated for particular counties in the United States of America. For Clay County, Minnesota, Burley *et al.* (1989) describes an equation useful in reconstructing landscapes for agronomic crops, Burley and Thomsen (1990) describe an vegetation equation for both crops and woody plants, and Burley (1990) presents a sugar beet (*Beta vulgaris* L.) equation. Burley and Bauer (1993) have described two equations for Polk County, Florida. Their first equation is applicable to woody plants and crops in upland landscape settings and the second equations is suitable for primarily lowland woody plants. The first two county equation was reported by Burley (1995b) where a vegetation model was developed for Clay County, Minnesota, and Cass County, North Dakota. In addition to generating equations, Burley (1994) presented the relationship of the equation approach to several state surface mining reclamation laws and regulations including the states of North Dakota, Wyoming, South Dakota, Minnesota, Michigan, and Indiana. Finally, Burley and Thomsen (1990) present an application of one equation to a mining site in Clay County, Minnesota. These studies form the essential literature associated with the soil survey predictive approach employed in this study.

Study Area and Methodology

Oliver County, a predominately agricultural landscape, resides in North Dakota's coal mining region. In a description of Oliver County's physiography, Weiser (1975:116) states, "Oliver County is on the western border of the area in North Dakota where soils formed in glacial deposits, and on the eastern border of the area where soils formed in residuum weathered from bedrock." Weiser (1975:1) notes that, "ninety-two percent of Oliver County is farmed. Most farms are a combination of livestock, feed-grain and cash-grain enterprises, but there are a few large ranches and cash-grain farms." Weiser (1975:1) also states, "The mining of lignite and the generating of electricity by coal-fired steam are of increasing importance to the economy."

Weiser (1975:113) presents a complete classification of each soil type examined in the study. Primarily the soils examined in this investigation are mollisols such as the Williams series (fine-loamy, mixed, Typic Agriborolls) and Mandan series (coarse-silty, mixed, Pachic Haploborolls). A few entisols are also included such as the Cohagen series (loamy, mixed, calcareous, frigid, shallow, Typic Ustorthents) and the Trembles series (coarse-loamy, mixed, calcareous, frigid, Typic Ustifluvents). Most of the soils are neutral or slightly alkaline. A few soils are saline in character. These soils comprise the physical substrates studied by Weiser. In the Oliver County study, 102 soil descriptions and associated crop production data were utilized.

The methodological approach was identical to the procedures described by Burley and Thomsen (1987) and reviewed recently by Burley (1996), where a soil depth weighting factor was applied to soil parameters (independent variables) and principal component analysis was applied to agronomic and woody plant productivity values (dependent variables) to search for covarying vegetation types. Both the dependent variables and independent variables selected for the study of Oliver County, North Dakota were derived from Weiser (1975). Multiple regression procedures were then applied to search for regressors which can predict plant growth for a linear combination of vegetation types. Table 1 lists the soil parameters employed in this study. The soil factors are similar to the factors selected in the development of previous studies, with the exception that bulk density soil profile values were not available for this study. Table 2 describes the vegetation variables selected in the study, including silage corn. The result of these

Table 1. Main effect independent variables and units of measurement from the U.S. Soil Conservation Service (Weiser 1975 and U.S. Department of Agriculture 1951).

Abbreviation	Factor	Unit of Measurement
FR	% Rock Fragments	Proportion by weight of particles > 7.62 cm
CL	% Clay	Proportion by weight
HC	Hydraulic Conductivity	Inches/hour (1 inch = 2.54 cm)
PH	Soil Reaction	pH
EC	Electrical Conductivity	Mmhos/cm
OM	% Organic Matter	Proportion by weight
AW	Available Water Holding Capacity	Inches/inch, cm/cm
TP	Topographic Position	Scale 0 to 5 Where: 0=Low (Standing Water) 2.5=Mid-slope, 5=High (Ridge Lines)
SL	% Slope	(Rise/Run)*100

Table 2. Dependent variables and units of measurement as recorded and published by the U.S. Soil Conservation Service (Weiser 1975).

Abbreviation	Vegetation	Measured Average Yield
Evergreen Trees		
JV	*Juniperus virginiana*	feet/20 years
PD	*Picea glauca densata*	feet/20 years
PP	*Picea pungens*	feet/20 years
PS	*Pinus ponderosa scopulorum*	feet/20 years
Deciduous Trees		
FP	*Fraxinus pennsylvanica*	feet/20 years
PD	*Populus deltoides*	feet/20 years
UP	*Ulmus pumila*	feet/20 years
Deciduous Shrubs		
CA	*Caragana arborescens*	feet/20 years
PA	*Prunus americana*	feet/20 years
PV	*Prunus virginiana*	feet/20 years
Agronomic Crops		
SW	Spring Wheat	bushels/acre
BA	Barley	bushels/acre
OA	Oat	bushels/acre
SI	Silage Corn	tons/acre
GL	Grass/Legume	tons/acre

1 meter = 3.281 feet; 1 foot = 0.3048 meter
1 hectoliter = 2.837 U.S. bushels;
1 U.S. bushel = 0.363 hectoliter
1 hectare = 2.471 acres; 1 acre = 0.405 hectare
1 kilogram = 2.2046 pounds avoirdupois;
1 pound = 0.4536 kilogram
1 kilogram = 1.10×10^{-3} ton, 1 ton = 907 kilograms

procedures is the generation of equations which employ soil parameters to predict a productivity index.

The productivity index is a unitless number, indicating relative productivity. By using Burley and Thomsen's (1987) method, reported vegetation productivity scores have typically ranged in scale from five to minus ten, where a score of five is a highly productive soil and a score of minus 10 is an unproductive soil.

An equation generated by this procedure was compared to a set of 10 reclaimed soils supplied by Dr. G. Halvorson from the Land Reclamation Research Center, North Dakota State University. The origin and description of this small data set is presented in Burley (1995). The equation developed in this Oliver County study was applied to the soil properties of the reclaimed soils and compared to actual yield values by computing the Pearson product-moment correlation. used to generate a predicted productivity score

Results

Table 3 illustrates the eigenvalues (latent roots) for the Oliver County dependent crop/woody plant variables. The eigenvalue associated with the first principal component axis contains 76.721 percent of the variance in the data set and is a primary candidate for further modeling analysis. The second and third principal component eigenvalues are greater than 1.0, suggesting that the second and third axes may also merit further modeling study. All other eigenvalues are smaller than one and thus are typically not considered for further analysis. The first three

Table 3 Principal Component Analysis eigenvalues of the covariance matrix for Oliver County, North Dakota, dependent variables.

	Eigenvalue	Proportion	Cumulative
PRIN1	11.5081	0.76206	0.76721
PRIN2	1.3379	0.089194	0.85640
PRIN3	1.0183	0.067886	0.92429
PRIN4	0.5352	0.035682	0.95997
PRIN5	0.2834	0.018896	0.97886
PRIN6	0.1205	0.008031	0.98689
PRIN7	0.0911	0.006070	0.99297
PRIN8	0.0498	0.003318	0.99628
PRIN9	0.0334	0.002226	0.99851
PRIN10	0.0151	0.001009	0.99952
PRIN11	0.0065	0.000432	0.99995
PRIN12	0.0004	0.000030	0.99998
PRIN13	0.0003	0.000018	1.00000
PRIN14	0.0000	0.000002	1.00000
PRIN15	0.0000	0.000000	1.00000

eigenvalues comprise over 92% of the variance in the crop/woody plant variables.

Table 4 presents the eigenvectors for the first five principal components. The coefficients for the first eigenvector are all positive and range from 0.275 to 0.162, suggesting that the dependent variables covary together and relatively equitably. This first set can be considered an all crop/woody plant response axis.

The coefficients for the second eigenvector can be arranged in three groups: positive coefficients, negative coefficients, and coefficients near zero. This second component axis suggests the dependent variables may be divided into a crop group negatively associated with the axis, a Ponderosa Pine/Siberian Peashrub group unassociated with the axis, and a general woody plant group positively associated with the axis.

The coefficients for the third eigenvector can be divided into two broad general groups. The first group is a set containing one coefficient greater than 0.4 and the second group contains coefficients less than 0.4. The woody plant Cottonwood contains a coefficient greater than 0.4 and all other variables contain values smaller than 0.4, with most of the values ranging near the value zero. This third set could be characterized as a Cottonwood response axis.

Table 4 Principal Component Analysis eigenvectors for Oliver County, North Dakota, dependent variables. See Table 2 for a definition of each variable. The letter "Z" attached to the end of each variable indicates that the variable has been standardized to a mean of zero and a standard deviation of 1.

	PRIN1	PRIN2	PRIN3	PRIN4	PRIN5
SWZ	0.26533	-.33929	0.15995	-.02214	-.04577
BAZ	0.26556	-.33747	0.15978	-.01862	-.05497
OAZ	0.26532	-.33957	0.15889	-.02314	-.04689
SIZ	0.26099	-.31749	0.12371	0.00652	-.03857
GLZ	0.27037	-.27483	0.17292	0.03092	0.01076
PSZ	0.25322	-.01422	-.45122	0.17704	0.20248
PPZ	0.24233	0.31424	0.29091	-.38440	0.30375
PDZ	0.24233	0.31424	0.29091	-.38440	0.30375
JVZ	0.26558	0.04200	-.28443	0.18449	0.45146
FPZ	0.27499	0.07863	-.17781	0.11668	0.19601
POZ	0.16159	0.35116	0.47523	0.75314	-.02890
UAZ	0.27345	0.23994	-.11174	-.08721	-.35124
PVZ	0.27330	0.24208	-.05510	0.03729	-.44571
PAZ	0.27058	0.18289	-.17143	-.20844	-.44339
CAZ	0.26639	0.02788	-.34974	0.05873	0.06213

Since the eigenvalues associated with the remaining principal components are less then 1.0, interpretation of the remaining sets of eigenvectors for further analysis is considered unnecessary. The first three axes merit further investigation.

Table 5 illustrates the selected model representing the best equation developed from the first principal component. This equation is the all crop/woody plant model. The equation is not over specific (terms in equation = 13, C(p) = 14.09); over specific equations contain more terms than the Mallows' C(p) value (see Burley 1988). The equation explains 81.08 percent of the variation in the crop/woody plant axis.

A stepwise, maximum R-squared procedure was initiated for the second eigenvector and then for the variable Cottonwood. Statistical analysis revealed best model equations with coefficient of multiple determination values of 0.2834 associated with the second eigenvector and 0.0725 for the Cottonwood equation. Each equation explained relatively low amounts of variance in the Oliver County data set. In addition, many of these equations were over specific. For example, a model for an equation derived from the second eigenvalue axis was over specific at the six

Table 5 Stepwise Maximum R^2 Improvement--Best equation selected.

R-square = 0.81083312 C(p) = 14.09255633

	DF	Sum of Squares	Mean Square	F	Prob>F
Regression	12	942.48974442	78.54081204	31.79	0.0001
Error	89	219.88228464	2.47058747		
Total	101	1162.37202906			

Variable	Parameter Estimate	Standard Error	Type II Sum of Squares	F	Prob>F
INTERCEP	0.50452660	0.24894486	10.14757008	4.11	0.0457
SLZ	-2.42593246	0.27062970	198.52112848	80.35	0.0001
AWZ	1.01860203	0.23236161	47.47670439	19.22	0.0001
FRZ	-1.98174961	0.36135098	74.30864258	30.08	0.0001
FRZFRZ	0.24521274	0.05872842	43.07147271	17.43	0.0001
TPZTPZ	-0.59309805	0.18066683	26.62541299	10.78	0.0015
ECZECZ	-0.92280839	0.18250496	63.16463267	25.57	0.0001
HCZHCZ	-0.20010668	0.09831059	10.23582352	4.14	0.0448
HCZAWZ	-0.63992792	0.16638745	36.54443455	14.79	0.0002
SLZTPZ	0.97386757	0.22872542	44.78898062	18.13	0.0001
ECZPHZ	0.97913197	0.28591670	28.97367992	11.73	0.0009
AWZCLZ	-0.71410862	0.21886327	26.30165100	10.65	0.0016
OMZECZ	1.87147762	0.31451985	87.47287530	35.41	0.0001

Bounds on condition number: 6.227012, 546.3928
The above model is the best 12-variable model found.

variable equation level (C(p)=3.735). Even with one variable entered into the Cottonwood model, the equation was over specific (C(p)=0.979). Therefore, all equations for the second and third principal component axes were rejected as insignificant with low powers of predictability and often containing a high degree of collinearity among the independent variables, due to low C-plot scores. Only the first axis provided an equation with an acceptable level of predictability, yet not over specific.

Finally, the Pearson product-moment correlation between the predicted score and the yield results from reclaimed soils was 0.71424 ($p \leq 0.0203$). Indicating that there is a positive statistical relationship between the soil survey approach and vegetation productivity on reclaimed soils.

Discussion

The procedures reported by Burley and Thomsen (1987) and applied to Clay County, Minnesota, appear to be applicable to Oliver County, North Dakota. A highly specific equation was constructed, with a definitive overall highly significant regression ($p<0.0001$), and an R-squared value similar to past reported equations.

The multivariate analysis revealed that the woody plants and agronomic crops covary. In other words, the soil conditions suitable for spring wheat are similar to the soil requirements for Colorado spruce and for the mixture of grass and legume. In addition, silage corn, an agronomic variable not employed in past productivity equation investigation, also followed this covariance pattern. Very little theoretical work or ecological modeling has been conducted to explain and predict this covariance phenomenon. In many respects, this covariance phenomenon is contrary to results typically presented in vegetation ecology ordinations such as in Curtis (1959). If the covariance phenomenon is corroborated by future investigations, this covariance may prove to be a powerful construct in reconstructing soil profiles across a wide variety of landscape conditions where high levels of vegetation growth are

required for prime farmland, urban landscapes, transportation right-of-way, and erosion control situations. This covariance phenomenon may merit further investigation by agro-ecologists.

In addition to the covariance phenomenon, some reclamation specialists unfamiliar with these modeling techniques become concerned about the mixing of harvest values from botanical species, such as oat, with harvest values from aggregated plant associations such as grass and legume. However, there are no strict investigatory rules limiting concurrent examination of hierarchical vegetation categories as dependent variables. Instead, the multivariate analysis examines variables and presents evidence concerning their covariance. Covarying categories can be represented in the same response axis; while categories that do not covary might be represented in different response axes (recall that these axes are orthogonal and thus independent). This study and past studies suggest that hierarchical vegetation categories between herbaceous plant associations, agronomic crop species, and woody plant species covary. The vegetation productivity concordance between hierarchical types is an investigatory topic that has been relatively unexplored and may merit further inquiry.

In contrast to the covariance phenomenon, the multivariate analysis presented in this study also suggested that at least two other models may merit development; however, further investigations revealed that these models were weak in predictive ability. Presently, the sugarbeet model reported by Burley (1990) is the only equation derived from a second latent root. All other equations were derived from the first latent root (largest eigenvalue), corroborating the covariance phenomenon.

When applying regression analysis using the first eigenvector, the results of statistical analysis (Table 5) revealed Equation 1 (Figure 1). Interpreting such an equation may be difficult. The equation contains a combination of linear main effect terms, squared terms and numerous interaction terms. The equation is relatively similar to other reported equations where steep slopes and a substantial portion of rock fragments reduce vegetation productivity; while increased available water holding capacity and abundant organic matter can increase vegetation productivity. Such findings are not new, but rather the results of the analysis allow the reclamation specialist to quantitatively assess these soil parameters.

Partitioning the linear combinations from the regression equation into each regressor effect or combination of effects, such as in Figure 2 can be useful to understand the contribution each soil parameter makes toward the assessment of vegetation productivity. Figure 2 is a graph which illustrates a negative relationship between vegetation productivity and % slope. Figure 2 is obtained by plotting the

$$
\begin{aligned}
\text{Oliverplants} = \; & 0.505+[(SL-5.662)*0.971^{-1}*(-2.426)] \qquad \text{[Eq. 1]} \\
& +[(AW-0.162)*0.028^{-1}*(1.019)] \\
& +[(FR-1.661)*4.334^{-1}*(-1.982)] \\
& +[((FR-1.661)*4.334^{-1})^{2}*(0.245)] \\
& +[((TP-2.721)*0.971^{-1})^{2}*(-0.593)] \\
& +[((EC-3.128)*2.074^{-1})^{2}*(-0.923)] \\
& +[((HC-2.655)*3.569^{-1})^{2}*(-0.200)] \\
& +[((HC-2.655)*3.569^{-1})*((AW-0.162)*0.028^{-1})*(-0.640)] \\
& +[((SL-5.662)*0.971^{-1})*((TP-2.721)*0.971^{-1})*(0.974)] \\
& +[((EC-3.128)*2.074^{-1})*((PH-7.535)*0.322^{-1})*(0.979)] \\
& +[((AW-0.162)*0.028^{-1})*((CL-25.081)*13.899^{-1})*(-0.714)] \\
& +[((OM-0.707)*1.097^{-1})*((EC-3.128)*2.074^{-1})*(1.871)]
\end{aligned}
$$

Where:	Oliverplant	=	Oliver County vegetation productivity value
	SL	=	% Slope
	AW	=	Available Water Holding Capacity
	FR	=	% Rock Fragments
	TP	=	Topographic Position
	EC	=	Electrical Conductivity
	PH	=	pH
	OM	=	Organic Matter

Figure 1. Oliver County, North Dakota vegetation productivity equation.

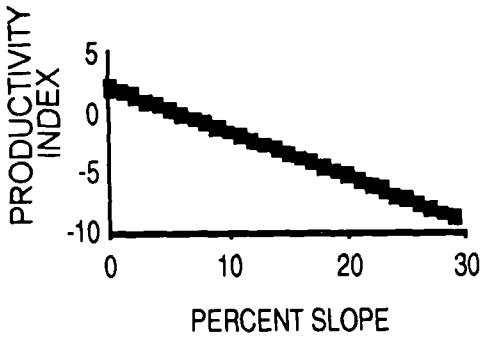

Figure 2. Main-effect relationship between percent slope and productivity index.

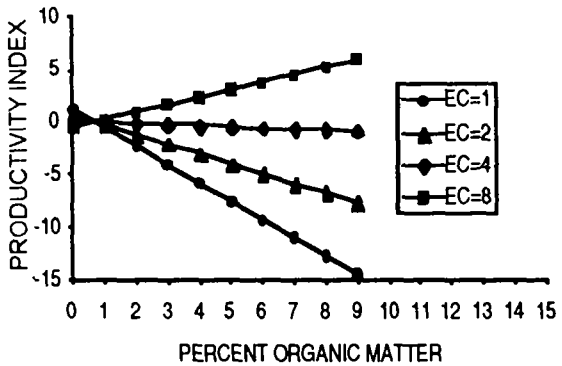

Figure 3. Productivity index determined by an electrical conductivity and percent organic matter interaction term (EC=Electrical Conductivity).

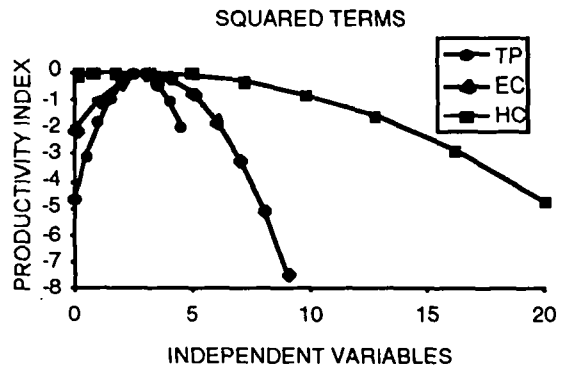

Figure 4. Parabolic relationship between squared terms and productivity index (TP=Topographic Position, EC=Electrical Conductivity, HC=Hydraulic Conductivity).

relationship between the dependent variable and a term in the regression equation containing percent slope, dropping all other regression terms, in this case Oliverplants = $[(SL-5.662)*0.971^{-1}*(-2.426)]$. In contrast to Figure 2, Figure 3 illustrates a partition of an interaction term where there is a linear relationship between percent organic matter and vegetation productivity across a variety of electrical conductivity settings. As electrical conductivity increases, the slope of the linear expression changes from a negative inverse setting to a positive proportional condition. The interaction term suggests that when electrical conductivity settings are low, low organic matter soil conditions will result in larger vegetation productivity levels; when electrical conductivity values are increased, increased organic matter will result in improved vegetation productivity levels. Notice in Figure 3 that no single electrical conductivity treatment across all percent organic matter levels will result in consistently obtaining the highest vegetation productivity level. The four lines illustrate the interaction relationship. These partition and graphing techniques allow the investigator to more fully understand and interpret the linear contributions within the equation. Burley (1988) demonstrates a comprehensive examination of a vegetation productivity equation by partitioning and graphing the results.

Not all regressors demonstrate an inverse or proportional relationship. For example according to the equation, moderate to low hydraulic conductivity rates, middle topographic positions, and moderately high electrical conductivity levels also maximize vegetation productivity, suggesting that there is an optimum parabolic (curvilinear) relationship for these regressors. Figure 4 illustrates the relationship for the squared terms from Equation 1 which contain a negative Beta coefficient.

The linear, interaction and parabolic forms of the independent variables are the basis of the vegetation productivity multiple regression equations presented for Clay County, Minnesota and now for Oliver County, North Dakota. The equation [Equation 1] provides a quantitative numerical approach to assess the vegetation productivity potential of various soil conditions.

As with some empirical equation modeling investigations, the interaction terms can be difficult to interpret. In general, some of the interaction terms can be considered correction factors, indicating a more complex relationship and allowing the independent

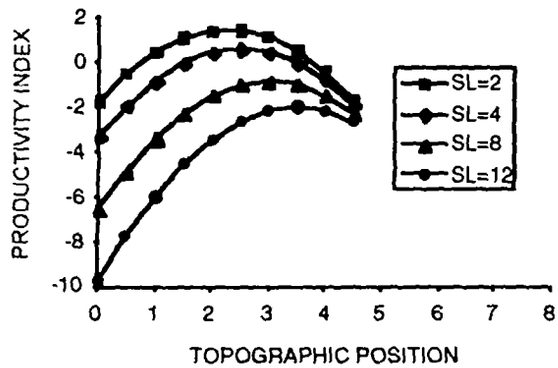

Figure 5. Productivity index determined by a combination of three terms: topographic position squared, topographic position and percent slope, and percent slope (SL=Percent Slope).

variables to more readily predict productivity. Figure 5 illustrates the depiction of a regressor that could be considered a correction of the term, allowing a better statistical fit. Note that slight slopes with middle topographic positions result in the largest vegetation productivity. Even though topographic position and percent slope are associated with interactions terms, the plot in Figure 5 demonstrates how a complex association of terms can result in a simple graphical expression of agricultural productivity where curves do not cross, suggesting a relatively intelligible interpretation of the equation.

In contrast to some counter intuitive segments of a large regression equation, some of these interaction terms found in the equation may intuitively make "common sense." For example, as organic matter increases, the water holding capacity and cation exchange capacity of some soils may increase, allowing electrical conductivity values to be greater without adversely affecting plant growth. Plants may be able to tolerate a greater level of salts/nutrients in the soil, provided water is present. In contrast, other interaction terms may be more difficult to explain, such as available water holding capacity times percent clay, where high levels for both variables are suggested as being disadvantageous for agronomic crops and woody plants. Although intuitively one might conclude that a wet clay soil may not be beneficial for crops and many woody plants, supporting investigatory evidence may not be as apparent. Consequently, some of the variables may be difficult to explain. Nevertheless, identification of these variables as significant regressors provides a pool of soil-plant relationships that may merit further study. There are still many multiple factor soil-plant investigations that have not been fully explored.

Some reclamation specialists may have reservations about the presentation of numerous interaction regressors identified in Equation 1 that may not seem intuitively meaningful. However, the methodological process originally presented by Burley and Thomsen (1987) is strictly an empirical procedure. The selection for study of any predictor for further analysis, whether a main-effect term (linear), a squared term (parabolic), a two-way interaction term, a five-way interaction term (examples in Dunker et al. 1992), a ten-way interaction term (originally hypothesized by Doll and Wollenhaupt 1985), an autoregressive term, or an exponential term, is in many respects a biased selection process, where the investigator makes an educated guess. The researcher must make a heuristic decision concerning which variables to test. Unless the investigator conducts a statistical analysis there is no reason to actually believe that a dependent variable and an independent variable are related in any mathematical manner. Thus, until a statistical examination is conducted, a main effect linear relationship is just as arbitrary as an obscure two-way interaction term. Through statistical analysis the terms are evaluated for internal validity. Acceptance or rejection of these terms may require an extensive number of investigations by numerous investigators over a substantial length of time. One reported investigation does not typically supply definitive answers, but may suggest directions for further research. Therefore, the large number of unusual interaction terms and regressors expressed in an equation should not be readily accepted or rejected.

While the internal validity of the regressors presented in Equation 1 are statistically supported, the equation presented in this paper and in others has an external validity shortcoming. The major external validity issue concerning the applicability of these models is that the equations are built from soil profiles and vegetation productivity values in a pre-mining condition. With the exception of this paper and the dissertation work prepared by Burley (1995a), no reclamation research specialist has reported the development of a soil survey model employing post-mining soil profiles and associated vegetation productivity scores.

Closing Remarks

Formative concepts concerning the development of vegetation productivity equations for assessing the plant growth potential of post-mining soil

profiles has led to the generation of inferential/empirical vegetation productivity equations. While much work remains ahead for reclamation research specialists to validate the models, develop theoretical constructs to explain the models, and expand the body of knowledge associated with soil-plant interactions, this paper illustrates that vegetation productivity models for North Dakota are possible. In reclamation planning and design, specialists can employ the equation to study and predict the vegetation productivity of various landscape configuration possibilities and to obtain an indicator of reclamation success.

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PRODUCTIVITY OF PRIME AND NONPRIME TOPSOIL RECLAIMED IN EQUIVALENT TOPOGRAPHIC POSITIONS

by

Gary A. Halvorson and M. V. Nathan

Abstract. Coal companies in North Dakota are required to separate primeland topsoil from non-primeland topsoil before mining. Following mining, the prime topsoil must be replaced in a prime location and the nonprime topsoil in a nonprime location. This separate handling of these materials is expensive and may be unnecessary. This research was conducted to (1) compare the productivity of prime and nonprime topsoil materials placed side by side in different topographic positions, and (2) to determine whether the separate handling of prime and nonprime topsoil is necessary. Plots were established at two different sites. The selected topsoil materials from Bowbells (prime soil), Williams and Zahl (nonprime soils), were transported to the reclaimed side of the pit and placed on separate plots adjacent to each other in the same topographic position. Plots were constructed on both prime and nonprime topographic positions and at the Coteau site two different topsoil depths were evaluated. In the first year of the study, dry matter yields and grain yields were lower on Zahl than on Bowbells or Williams topsoils. These differences could be accounted for by differences in initial soil moisture levels. The year 1993 was a wet year and the crop had sufficient available moisture in the profile throughout the season in the top 0 to 60 cm depth. At both sites there were no significant grain yield differences between prime and nonprime soils. No significant grain yield differences were observed between the topographic positions in the landscape. In 1994, at the Falkirk site no significant differences in yield could be determined between the three different soil series. At the Coteau site the wheat grown on the Zahl soil yielded slightly less than that growing on the Williams and Bowbells soils. As would be expected in a year when moisture was short, topographic position made a difference in yield. Depth of topsoil made no difference in yield for 1994.

Additional Key Words: mining, soil moisture, landscape

Introduction

Current federal and state regulations require separate handling of prime and nonprime topsoils. According to the present interpretation of prime farmland criteria, soils designated prime in the ustic moisture zone of North Dakota qualify because of landscape position. Most of these soils occur on nearly level or concave positions of the landscape and receive runoff from adjacent soils in a higher position which do not meet prime farmland criteria. Prime soils are therefore the product of microclimate and local surface and root zone hydrology rather than macroclimate or parent material.

In western North Dakota, availability of water is the most dominant factor controlling crop yields. Under conditions of limited rainfall, which is the general rule, the yield potential of primeland may not be significantly different from the yield potential of nonprime land. If the differences in the productivity

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capabilities of prime and nonprime soils are the results of moisture differences due to topographic location rather than to differences in the properties of soil materials, then the currently required separate removal and placement of topsoil materials is unwarranted. In addition, higher overall productivity of reclaimed land may be attained by replacing available soil materials uniformly on an area reshaped to the most effective topographic configuration.

In a greenhouse study, yields on reclaimed prime soils were initially higher than on reclaimed nonprime soils (Carter and Doll, 1983). After soil structure had been reestablished yields between the soils were not different.

Carter and Doll, (1987) reported that "*in situ*" soil properties such as bulk density, macropore space, and hydraulic conductivity are the soil parameters most severely disrupted during mining and reclamation. In continued studies, Carter (1991) found that average values of soil chemical properties, texture, and calculated percents of pore sizes were not significantly different between prime and nonprime soils located in a 25 acre site. Bulk densities at all measured depths were generally higher from prime soils than nonprime during all four years of the study.

Topography has been shown to influence crop yields on land reclaimed following mining in North Dakota (Doll et al., 1984). A method to quantify the relationship between the topographic redistribution of water and wheat yields was developed (Halvorson and Doll, 1991).

This study was undertaken to compare the productivity of prime and nonprime topsoil materials when placed side by side in different topographic positions.

Methods and Materials

One site was selected at the Coteau Properties Freedom Mine. Two plot areas were selected which would be reclaimed as rangeland with a topsoil depth of about eight inches. One of these was located on a hilltop to simulate a nonprime topographic position and the other was nearby on a toeslope position to simulate a prime topographic position. Two more plot areas were selected nearby which would be reclaimed as cropland with a topsoil depth of 15 inches. One site was on a hilltop and one was on a toeslope position to simulate nonprime and prime topographic positions, respectively. The other location was at the Falkirk Mine. Only two plot areas were selected at this location, one in a prime topographic position and the other in a nonprime topographic position which were reclaimed with a cropland soil depth of 15 inches.

Two nonprime soils used in this study were: the Zahl loam series (fine-loamy, mixed Entic Haploboroll) which is found on hilltops and shoulder positions and the Williams loam series (fine-loamy, mixed Typic Argiboroll) which is located on sideslopes and hilltops. The one prime soil used was the Bowbells loam series (fine-loamy mixed Pachic Haploboroll) which is located on footslopes and toeslopes.

Subsoil from each soil series was not segregated. Mixed subsoil was laid down at uniform depths on graded spoil material. Scrapers were used to transport the subsoil and topsoil to the plot areas. Topsoil from the Williams, Bowbells and Zahl soil series was placed on the plots at a depth of eight inches on rangeland plots and 15 inches on cropland and were replicated three times at each plot area. The plots were carefully leveled and smoothed with a bulldozer. The plots were disked and seeded to Stoa spring wheat (Triticum aestivum L.) on June 23, 1992. The plots at the Falkirk location were not completed until August, 1992 and therefore, were not planted for the first time until 1993. Soil samples from each plot were taken from each plot using recommended soil test procedures. Plots were fertilized with N and P for a 50 bushel per acre yield. Plots were seeded at the rate of one million live seeds per acre.

The plots were harvested by hand cutting three square meters from each plot, drying the bundles and then threshing the wheat in a small thresher. Yield data from three years, 1992-1994, is presented. Statistical

comparisons of the data were made within each years' data using analysis of variance and least significant difference (.05) comparisons. Least significant differences are for interaction means for all treatments at a site for a given year.

Results and Discussion

In 1992 wheat yields were very low because of the very late planting date at the Coteau site (Table 1). There were no significant differences between prime and nonprime topographic positions and no significant differences between cropland and rangeland soil depths. On the cropland soil depth wheat yields on the Zahl soil were significantly lower than on the Williams or Bowbells soils. This is mainly the result of a lower soil moisture content in the Zahl topsoil. Very little precipitation fell on these soils in the weeks prior to plot construction. The topsoil was removed from the surface five inches of the Zahl soil and the surface 15 inches and 25 inches from the Williams and Bowbells soils respectively, for plot construction. Since the surface was very dry and more moisture was available deeper in the profile, the topsoil material from the Zahl soil was the driest of the three soils. A plot of wheat grain yield versus initial soil moisture in the surface foot show the high correlation (r^2) = 0.71 between these two factors (Figure 1).

In 1993, growing season precipitation was 13.0 and 14.4 inches at the Coteau and Falkirk sites respectively, which was much above the average of 6.9 and 7.7 inches respectively. There were in fact, problems with minor flooding and water standing on the plots. In particular the prime topographic position at the Falkirk site had lower yields than the non-prime topographic position due to problems with too much water (Table 2). Wheat yields on the Bowbells and Williams soils on the prime topographic position were actually lower than on the Zahl soil. No yield differences between soils were noted on the nonprime topographic position. At the Coteau site in 1993 (Table 1) no statistically significant differences occurred between prime and nonprime topographic locations or between the three different soils tested. Wheat yields were significantly lower on the rangeland soil depth than on the cropland soil depth.

Table 1.Wheat grain yields (bu/ac) at the Coteau location, 1992-1994.

Topographic Landscape Position		
Soil Series	**Prime**	**Nonprime**
	Cropland - 1992	
Bowbells	9	10
Williams	8	9
Zahl	5	2
	Rangeland - 1992	
Bowbells	11	10
Williams	11	9
Zahl	9	8
LSD (0.5) = 3		
	Cropland - 1993	
Bowbells	52	52
Williams	46	50
Zahl	53	49
	Rangeland - 1993	
Bowbells	41	42
Williams	43	40
Zahl	44	39
LSD (0.5) = NS		
	Cropland - 1994	
Bowbells	30	31
Williams	29	26
Zahl	29	22
	Rangeland - 1994	
Bowbells	35	20
Williams	40	19
Zahl	29	18
LSD (0.5) = 6		

Table 2. Wheat grain yields (bu/ac) at the Falkirk location, 1993-1994.		
	Landscape Position	
Soil Series	Prime	Nonprime
	bu/ac	
	1993	
Bowbells	36	51
Williams	37	49
Zahl	45	51
LSD (.05) =8		
	1994	
Bowbells	34	26
Williams	39	32
Zahl	34	27
LSD (.05) = 11		

Wheat yields were somewhat lower in 1994 due to an extended period during the middle of the growing season with very little precipitation. At the Falkirk site no significant differences in yield between the three soils occurred (Table 2). Yields from the nonprime topographic position were significantly lower than yields from the prime topographic position. At the Coteau site in 1994 no differences in wheat yields occurred between cropland and rangeland soil depth (Table 1). The wheat yield on nonprime topographic positions was lower than on prime topographic positions. On the cropland soil depth in a nonprime topographic setting wheat yields were lower on the Zahl soil than on the Bowbells. On the rangeland soil depth in a prime topographic setting yields of the Williams soil were higher than the Zahl soil, but no significant differences between yields on the Zahl and the Bowbells soil occurred. Differences between the three soils did not occur on the other two plot areas at the Coteau mine.

Conclusions

Differences did occur in the treatments in the three years of the study. The Zahl soil yielded less in 1992 than the Bowbells soil, but this could be explained by the difference in initial soil moisture content. The Bowbells soil also yielded higher than the Zahl soil on the cropland soil depth in a nonprime topographic position in 1994. On the other hand in 1993 the Zahl soil yielded higher than Bowbells in a prime topographic position at the Falkirk location. Otherwise, the wheat yields from the three different soils were statistically equivalent. Overall, this is good evidence that the topsoil material itself from these three soils is of equal productivity.

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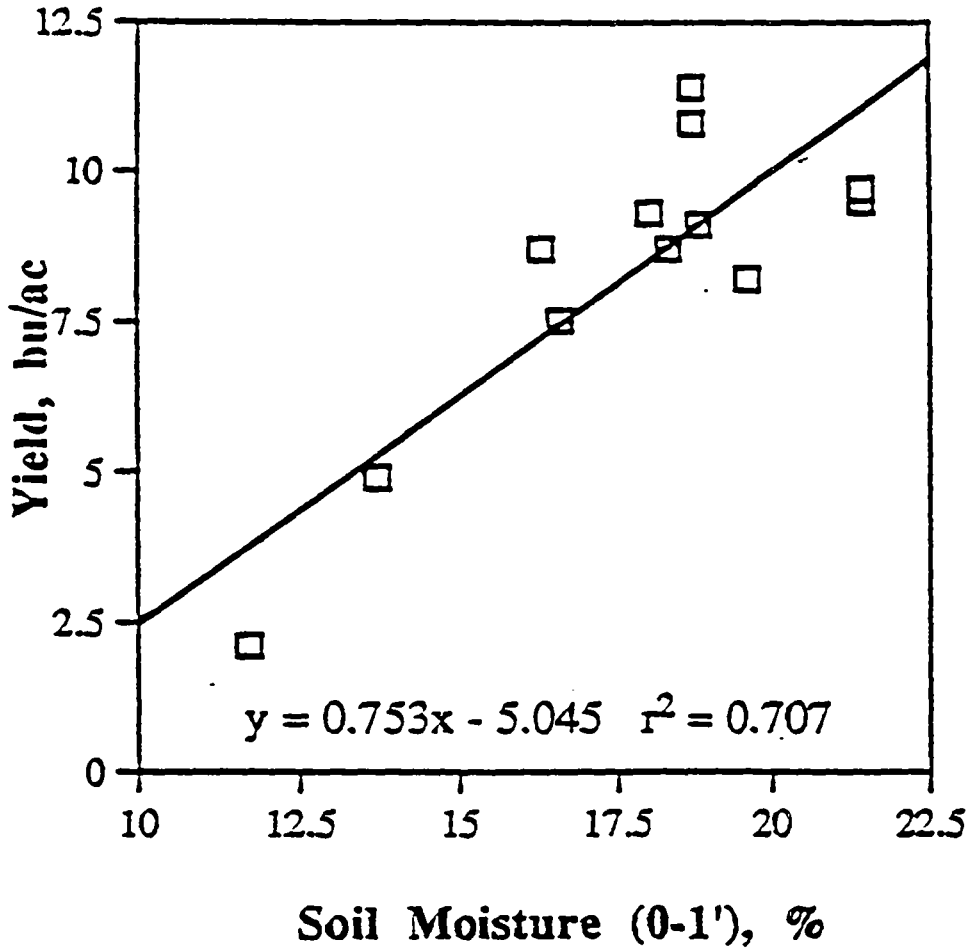

Figure. 1. The relationship between initial soil moisture at the 0-1 foot depth and grain yield in 1992 at the Coteau site.

A METHODOLOGY FOR EVALUATING THE COSTS OF SELECTIVE HANDLING OF OVERBURDEN MATERIALS[1]

by
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Abstract. Selective handling of overburden materials involves implementing mining techniques that will reduce the probability of creating acid mine drainage (AMD). These include separately handling toxic materials and placing them in locations in the backfill to limit their exposure to air and water, or blending alkaline- and acid-producing materials to create a neutral rock mass. This paper uses three selective handling situations and estimates the additional costs of selective handling above those which would normally be incurred during mining. Scenario 1 depicts a 2-m (6-ft) toxic shale layer (acid-producing) on top of the coal that must be specially handled and placed within non-toxic layers in the backfill. Scenario 2 has a 2-m toxic shale layer over the coal and also a 2-m layer of alkaline material, and these two layers are blended during backfilling. Scenario 3 details specially handling a 0.4-m (1.4-ft) toxic layer located 5 m (15 ft) up from the coal bed which requires blasting in two stages. Based on these scenarios and the overburden layers separately handled, additional costs for loading and hauling the material ranged from $663 per 46 by 43 m (150 x 142 ft) cut (or about $3,348 per ha or $1,353 per acre) for scenario 1 to $1,119 per 46 by 43 m cut (or about $5,652 per ha or $2,284 per acre) for scenario 2. These cost estimates do not include additional management costs, additional blasting costs, or training costs for equipment operators.

Additional Key Words: acid mine drainage, acid-producing materials, backfilling, land reclamation, mine hydrology, reclamation economics.

Introduction

Selective handling of overburden materials refers to several techniques that are designed to reduce the probability of encountering an acid mine drainage (AMD) problem during and after mining. These techniques are currently being used by surface mine operators to meet their Surface Mining Control and Reclamation Act (SMCRA) permit obligations and to minimize the costs of post-mining reclamation, especially with regard to AMD treatment. In 1979, the West Virginia Surface Mine Drainage Task Force (SMDTF, 1979) published "Suggested Guidelines for Method of Operation in Surface Mining of Areas with Potentially Acid-Producing Materials." This publication outlined techniques for handling surface water, ground water, and overburden during the mining process to help control the production of AMD. The task force recommended: 1) identification of acid- or alkaline-producing materials before mining, 2) mixing or blending acid and alkaline materials during backfilling when alkaline materials were in greater amounts than acid-producing materials, and 3) isolation of acid-producing material within the backfill to minimize contact with air and water when insufficient alkaline materials were available to neutralize potential acidity.

The selective handling process has two stages: analysis and implementation. The analysis stage involves identifying the quantity, type, and location of acid- and alkaline-producing materials in the overburden. The implementation stage involves developing and carrying out the selective handling plan. The plan may incorporate a number of techniques including special blasting or fragmentation techniques; blending, mixing, and isolation of overburden layers during backfilling; the use of liners or seals; alkaline addition; and total removal and disposal of acid-producing materials off site or in a refuse area.

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Analysis

It is necessary first to determine the composition of the overburden in order to select the best mining methods, the proper equipment, and the most efficient selective handling techniques. There are three primary techniques currently used to determine overburden characteristics: 1) acid-base accounting, 2) leaching and weathering procedures, and 3) modeling of hydrogeochemical systems based on data from Acid-Base Accounting (Perry 1985, Skousen et al. 1987). Each technique has advantages and disadvantages and the techniques may also be used alone or in combination. All techniques require sampling distinct rock units in the overburden and subjecting the rocks to whole rock analysis or leaching procedures. The results of the analysis should give the operator an estimate of the quantity, type, and distribution of acid-producing and acid-neutralizing materials in the overburden. Continued sampling and analysis of overburden should occur during the mining process.

Implementation

Once the composition and distribution of the overburden materials have been established, a plan for overburden removal and replacement can be developed. The hydrological characteristics of the site are important in determining this plan and may designate the areas on the site where placement of acid-producing materials should or should not occur. In general, it is best to keep toxic materials isolated from water to minimize leaching of acid into water courses. **Fragmentation** refers to the practice of blasting or ripping overburden into predetermined sizes. The operator's goal is to minimize the total surface area of acid-producing materials by leaving those strata largely intact, thereby reducing their acid-producing potential. In contrast, operators attempt to maximize the total surface area of alkaline strata by blasting or ripping them into the smallest possible fragments to enhance their neutralizing capability (Skousen and Larew 1994).

If analysis of the overburden indicates that there will be enough alkaline-producing material at the site to neutralize the acid-producing material, the mine operator may choose to **blend** these two materials. There are two ways to blend. The first is by pushing and mixing the alkaline and acid-producing materials together with equipment during overburden removal and placement. The other method, called **layering**, involves placing a layer of alkaline material on top of a layer of acid-producing material.

Isolation involves placing acid-producing material away from the highwall and above the pit floor in the backfill. Isolation is designed to minimize the contact of acid-producing material with oxygen and water. Isolation is accomplished by placing alkaline material above, below, and around the acid-producing material. In its most elaborate installation, a low permeability barrier or seal of compacted material is placed above or around the acidic material. Clay, shale, or synthetic **seals** or liners reduce air contact with acid-producing material and decrease water infiltration into the acid material.

Alkaline material in the form of limestone or other lime products can be imported when insufficient quantities of alkaline materials are found on-site to neutralize the acid-producing materials or to create a neutral rock mass. These products can be used in conjunction with the other selective handling techniques. They can be blended with or placed strategically around the acid-producing materials (Meek 1991, Skousen and Larew 1994). **Total removal** involves taking the acid-producing materials from the mining site to an area that is better suited for their disposal. Refuse sites are a common location for disposing of acid-producing materials. The area that receives the toxic materials will also need to be analyzed, sufficient alkaline material emplaced, and the entire area compacted and reclaimed.

Selective handling techniques are not mutually exclusive. These methods can be used to complement each other in some situations, while in others it may be more efficient to concentrate on a single technique with which the operator has the most confidence and familiarity.

Costs of Selective Handling

Estimating the costs of selective handling is a more complex task compared to evaluating the costs of AMD treatment systems as demonstrated in Phipps et al. (1991). With AMD treatment systems, costs can be broken down into investment costs, chemical reagent costs, annual labor and maintenance costs, and management costs (Skousen 1991, Skousen et al. 1993). These costs can be estimated separately and added together to get the total cost of the AMD treatment system. Management costs--the most difficult costs to quantify--are a small proportion of total costs with AMD treatment. With selective handling of overburden, management costs can be substantial, both in terms of developing the original toxic materials special handling plan to obtain the SMCRA permit and in day to day operations to identify, move, blend, or isolate acid-

forming materials. Selective handling practices are highly site specific. Costs will vary greatly depending on whether the acid-forming strata lie directly on top of the coal, or occur in multiple strata throughout the overburden. In the latter case, blasting may have to be conducted in multiple stages, with the overburden above each acid-forming stratum blasted separately.

Several studies have been concerned with the costs of selective handling. Kelley et al. (1982) developed a mining engineering plan to integrate selective placement of overburden into the mining cycle. They used a two bench, dragline operation as a case study and compared it to a typical shovel-truck contour mining method. Their goal was to find the most cost-effective method of selective placement for their particular mining situation. They found that truck haulback of toxic overburden was more cost effective in the mountainous Appalachian Coal Region of the eastern U.S than the use of draglines. They did not attempt to break out the incremental costs due to selective handling.

Meek (1991) evaluated the costs and effectiveness of a number of selective handling procedures employed at the Upshur Complex of Island Creek Coal Corporation in West Virginia. The techniques employed on the site include isolation and placing the toxic material above the pit floor; layering non-toxic overburden above the toxic material; covering acid-producing material with a plastic liner; and mixing phosphate and alkaline materials with toxic materials. The study found that all the selective handling techniques added significant costs to the mining operation, but that each technique reduced the total amount of acid produced on the site. Of the techniques used at the site, Meek found the two most cost-effective practices were the addition of phosphate to toxic material and the placement of a liner above toxic material. The study also compared these costs of selective handling with the costs of water treatment over specified time periods. At this site, mining costs (including special handling costs and alkaline amendments) were estimated to be $47,000 per ha ($19,000 per acre) compared with annual treatment costs of $8,400 per ha ($3,400 per acre). Therefore, if water treatment continued for more than six years, selective handling techniques resulted in lower overall reclamation costs.

Skousen and Larew (1994) studied the costs and effectiveness of transporting alkaline materials about 1.6 km (1 mile) and blending these materials with acid-producing overburden at a coal mine near Masontown, WV. The authors estimated the costs of transporting the alkaline material to be about $9,900 per ha ($4,000 per acre) and the costs of water treatment per year to be about $800 per acre. Like Meek's study, selective handling and importing alkaline materials was expensive during the mining phase of the operation, but this process eliminated the long-term treatment of AMD. If water treatment was necessary on the site for more than five years, selective handling was a more cost-effective reclamation approach.

The primary limitation of the latter two studies is their site specific focus, and the results cannot be effectively applied to other sites. The goal of this paper is to present a method for estimating the incremental costs of special handling toxic materials during the mining process for a contour surface mining operation in Appalachia. The method attempts to separate the costs of special handling into their individual components and then determine how each of the component costs vary with different mining situations and geologic characteristics.

The analysis is simplified in this paper by assuming that all SMCRA conditions have been met (i.e., any special core drilling to determine geologic layers, analysis by acid-base accounting, and other costs to obtain the necessary SMCRA and National Pollution Discharge Elimination System (NPDES) permits). The cost of gaining information needed to obtain the SMCRA permit includes any expenses necessary to develop a toxic materials special handling plan including identification and classification of overburden layers. While the three example scenarios are simplified from real world conditions, the method should be readily adaptable to other conditions.

Methods

Three scenarios are used to capture the different cost components that may be encountered when using selective handling on a contour surface mine using the haulback method. Each scenario is based on actual mine conditions found in West Virginia. All three scenarios are assumed to have a single, 1.5-m (5-ft) thick coal bed and an average ground surface slope of 25 percent. Based on this slope, the mine in each case is developed into the hillside to a distance of 39 m (129 ft) correlating to a highwall height of 20 m (64 ft), using the rule of thumb of 0.3 m (1 ft) of highwall for every 2.5 cm (1 inch) of coal (Sengupta 1993). Each stage or cut is 46 m wide (150 ft) along the contour and 39 m (129 ft) into the hillside. However, land area is calculated from 46 m in width and 43 m in length (142 ft) up the slope equaling 1,978 m^2 or 0.198 ha (21,300 sq ft or about 0.49 acres). Topsoil is stockpiled and spread later on the reclaimed area during typical contour mining, so no additional costs

are assumed for topsoil handling and placement with these scenarios except for increased hauling distance. For scenario 1, there is a 1.8-m (6-ft) acid-producing layer that lies directly on top of the 1.5-m coal bed. Scenario 2 adds a 1.5-m alkaline layer located in the middle of the overburden. Scenario 3 has only a 0.4-m (1.4-ft) acid-producing layer in the middle of the overburden.

All scenarios are compared to a typical contour mining operation using the haulback method that does not utilize special handling (Figure 1). As shown, pit (cut) 1 has been backfilled and reclaimed with the overburden from pit 3 and the topsoil from pit 4. The coal has been removed in cut 2 and backfilling may begin with overburden from cut 4 and topsoil from cut 5. It was assumed that machinery operators possess sufficient skills to identify, move, and place toxic materials adequately.

The additional equipment and labor costs of selective handling for one 46-m cut were calculated by determining the increased number of front-end loader cycles required to load overburden, toxic material, and topsoil. Each front-end loader cycle takes approximately one minute (based on field observation) and loads 10 cubic meters (13 cubic yards) of material. The total time per front-end loader cycle was 20 percent longer when handling toxic or alkaline material than when loading overburden because of the greater effort required to keep these materials separated from the rest of the overburden.

The number of truck cycles needed to haul the material was calculated based on truck and front-end loader capacities. Cycles were then converted to time, using data from Caterpillar, Inc. (1991). Truck turning and maneuvering was estimated to take 0.7 minutes at the loading site and 1.1 minutes at the dump site. Trucks hauled 30 cubic meters (39 cubic yards) of material, and loading time was assumed to be 3 minutes (three loader cycles). Travel time from the loading to the unloading site was based on an average speed of 13 km per hour (8 miles per hour). Finally, half a minute of slack time for each truck cycle was added to ensure the loader remained busy. The cycle time was estimated to be 7 minutes for the 183-m (600-ft) haul (2 pit widths back and forth or a total of 4 pit widths), 7.5 minutes for the 275-m (900-ft) haul (3 pits widths back and forth), 8 minutes for the 367-m (1200-foot) haul (4 pit widths), and 8.5 minutes for the 459-m (1500-ft) haul (5 pit widths). The costs developed in this paper should be added to the costs incurred for a typical mine operation per 46 by 43 m (150 by 142 ft) pit or per ha.

Results and Discussion

Scenario 1

While scenario 1 is the most simplistic of the three, it illustrates many of the management issues encountered with selective handling. Scenario 1 has a 1.8-m toxic (acid-producing) shale layer over the coal bed and a 4.4-m (14.5-ft) layer of oxidized material to encapsulate toxic material above the pavement and away from the highwall (Figure 2).

Figure 3 shows the mining process in six stages. The first cut has been fully reclaimed. The reclamation process proceeded as follows. After the coal was removed from cut 1, part of the oxidized material from cut 4 (no acid-producing potential) was placed on the pit floor or pavement of cut 1; then the toxic material from cut 3 was placed over the oxidized material, and then the remaining oxidized material from cut 4 was used to cover the toxic material. The overburden from cut 4 was then backfilled in cut 1, followed by a layer of topsoil from the newly opened cut 5. The coal from cut 2 is removed while overburden from cut 4 and topsoil from cut 5 are being placed in cut 1. The backfilling process in cut 2 then continues with part of the oxidized material from cut 5 being placed on the pavement of cut 2, followed by the toxic layer from cut 4, the remainder of oxidized material from cut 5, then backfilling with overburden from cut 5, and finishing with topsoil from cut 6 until cut 2 is fully reclaimed. Figure 4 shows a cross section of a fully backfilled cut with the dark area representing the toxic material placed within encapsulating layers of oxidized material.

With the mining system described above, the primary additional costs of selective handling are: 1) a longer haul of one additional cut length (46 m one-way or 92 m for round trip) for the overburden and topsoil; 2) additional costs incurred from having to shift the front-end loader from loading oxidized material in cut 4 to loading toxic material in cut 3 and then back to loading the remaining oxidized material in cut 4; and 3) delays involved when handling toxic material or when switching from one material to another. The equipment used in the operation is shown in Table 1. Total haul length for hauling the toxic material is 183 m (600 ft) or 2 pit widths back and forth, the length of the haul for overburden is 275 m (900 ft, or 3 pit widths) because it has to be moved the length of one additional cut, and the haul for topsoil is 366 m (1200 ft, or 4 pit widths). If selective handling were not being used, there would be one less cut, reducing the haul length for the overburden and topsoil by 92 m (300 ft).

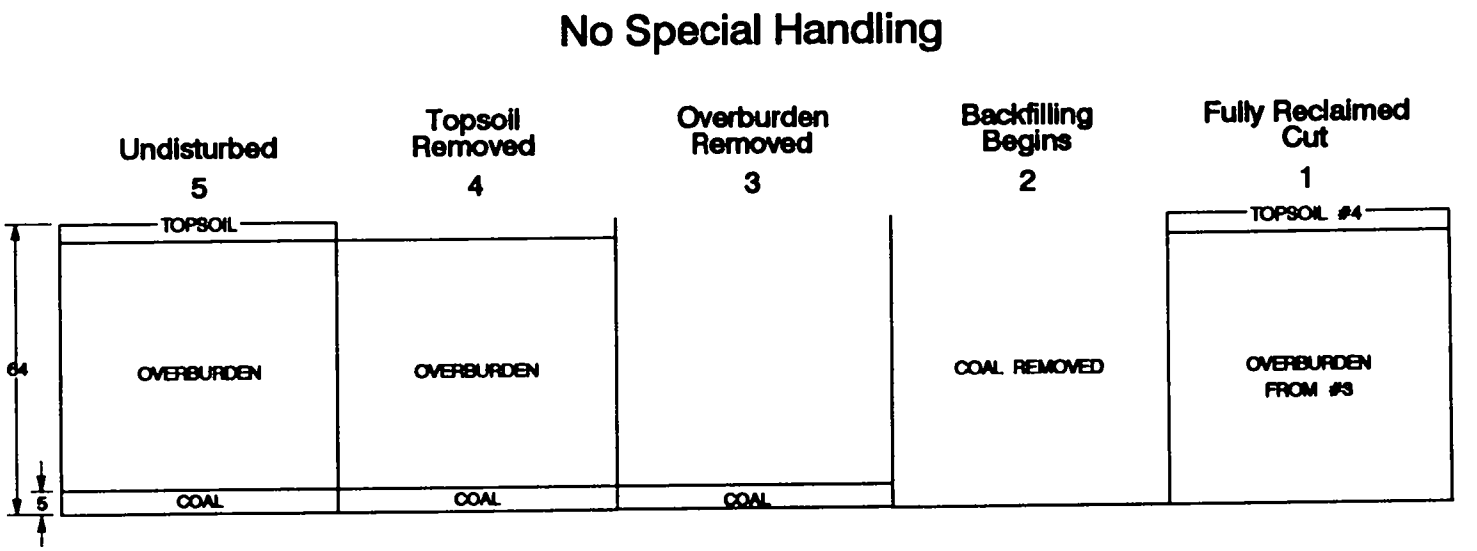

Figure 1. A simplified view of the mining process for a typical contour surface mining operation in West Virginia. Normally three pits are open at any one time to remove topsoil, overburden and coal. Cut 1 is fully reclaimed and shows the material and pit number used for backfilling.

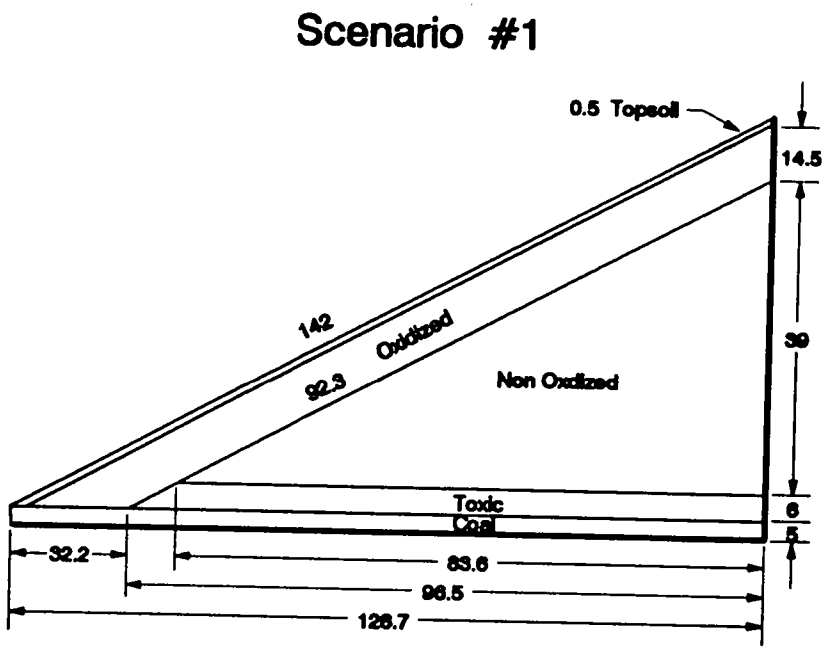

Figure 2. Undisturbed geologic cross-section for scenario 1 (all numbers in feet) with a 6-ft toxic layer overlying the coal bed.

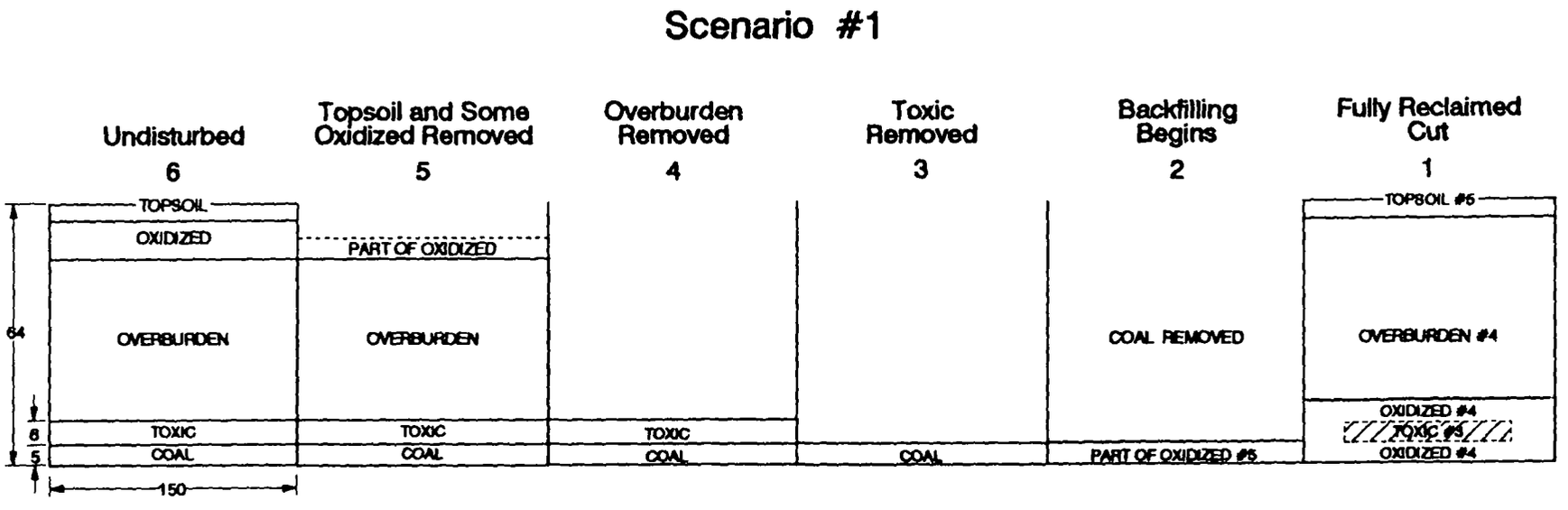

Figure 3. Simplified view of the mining process of a contour operation (scenario 1) where a 6-ft toxic layer above the coal bed is being selectively handled. Cut 1 is fully reclaimed and shows the material and pit number used for backfilling. In this case, four pits are open.

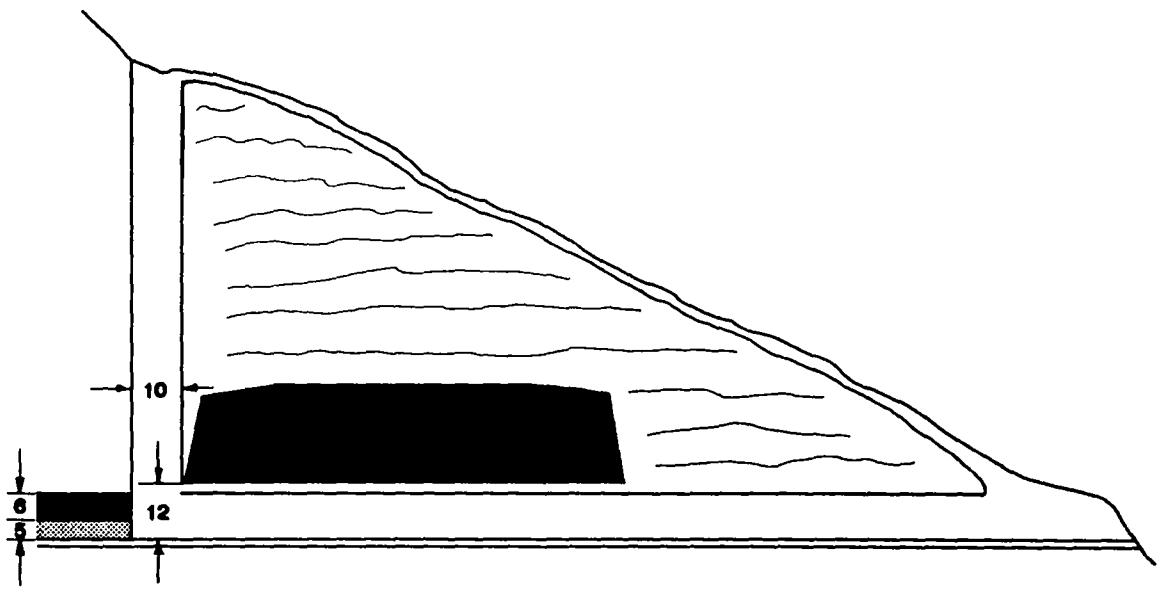

Figure 4. Cross sectional view of a fully backfilled cut for scenario 1 (numbers are in feet). The black material is the 6-ft toxic material special handled and placed on a layer of oxidized, non-toxic material.

Table 1. Types and purpose of equipment used in contour surface mining.

Type	Number	Capacity	Purpose
Cat D8[1]	1		backfilling
Cat D9	1		ripping and pushing
Cat 992	1	13 cu yds	front-end loader
Cat 773	3	50 tons	rock trucks

[1]Use of a specific brand name does not constitute endorsement of a product or its performance by the U.S. Bureau of Mines, the National Mine Land Reclamation Center, West Virginia University, or any of its affiliates.

Selective handling increased front-end loader time by an estimated 56 minutes and truck time by an estimated 278 minutes for each 46-m (150-ft) reclaimed cut. Using Caterpillar, Inc. (1991) costs (plus $39 per hour for the equipment operators), operation costs for the equipment were estimated to be $184 per hour for the loader and $104 per hour for the rock truck (under extreme terrain conditions). The total estimated increase in costs due to selective handling for both a front-end loader and rock trucks was $663 per 46 by 43 m (150-ft) cut ($3,348 per ha or $1,353 per acre). The results are shown in Tables 2 and 3.

Table 2. Estimated additional costs for a front-end loader in scenario 1 having a 1.8-m (6-ft) toxic layer above the coal bed.

	Cubic Yards	Loading Cycles	Total Time (min)	Increase For Selective Handling (min)	Cost of Increase ($184/hr)
Overburden (1 min)	21,655	1,667	1,667	0	0
Toxic (1.2 min)	3,607	278	334	56	$172
Topsoil (1 min)	394	30	30	0	0
Total	**25,656**	**1,975**	**1,975**	**56**	**$172**

Table 3. Estimated additional costs for rock trucks in scenario 1 having a 1.8-m (6-ft) toxic layer above the coal bed.

	Cubic Yards	Truck Cycles	Total Time (min)	Increase For Selective Handling (min)	Cost of Increase ($104/hr)
Overbrdn (7.5 min)	21,655	555	4,163	278	$482
Toxic (7.0 min)	3,607	92	644	0	$0
Topsoil (8.0 min)	394	10	80	5	$9
Total	**25,134**	**657**	**4,887**	**283**	**$491**

Scenario 2

Scenario 2 adds a 1.5-m (5-ft) layer of alkaline material in the middle of the overburden (Figure 5). In this scenario, selective handling involves blending the alkaline material with the toxic material as it is placed on oxidized material above the pavement. This procedure requires opening one more cut than scenario 1. The mining process is shown in seven stages in Figure 6. The first cut has been fully reclaimed. The reclamation process proceeded as follows. After the coal was removed in cut 1, part of the oxidized material from cut 5 is placed on the pavement. Then the toxic material from cut 3 and the alkaline material from cut 4 are placed on top of the oxidized layer, followed by the remaining oxidized material from cut 5. Backfilling continues with the lower overburden layer from cut 4 (OB2) and the upper overburden layer from cut 5 (OB1). Backfilling of cut 1 is finished with the topsoil from cut 6.

Additional costs for selective handling in scenario 2 (Figure 6) based on no selective handling (shown in Figure 1) occur because oxidized material, OB1, and topsoil must be hauled two additional cuts and alkaline material and OB2 must be hauled an additional cut. As in scenario 1, front-end loader time is increased by 20 percent when handling the toxic or alkaline materials. Total and incremental costs for the front-end loader and rock trucks are given in Tables 4 and 5.

Front-end loader time was estimated to increase by 101 minutes. Rock truck time would increase by 467 minutes per 46-m cut. Valuing the time as before for both a front-end loader and rock trucks, this would add $1,119 to the cost of mining for each 46 by 43 m cut, or $5,652 per ha ($2,284 per acre). The additional costs of scenario 2 are due to the two additional open cuts and the increased front-end loader time from having to selectively handle both toxic and alkaline materials.

Table 4. Estimated additional costs for a front-end loader in scenario 2 having toxic and alkaline layers that are selectively handled.

	Cubic Yards	Loading Cycles	Total Time (min)	Increase For Selective Handling (min)	Cost of Increase ($184/hr)
Oxidized (1 min)	9,750	750	750	0	0
Toxic (1.2 min)	3,607	278	333	56	$172
Alkaline (1.2 min)	2,930	225	270	45	138
OB1 (1 min)	4,118	317	317	0	0
OB2 (1 min)	4,857	374	374	0	0
Topsoil (1 min)	394	30	30	0	0
Total	**25,656**	**1,974**	**2,074**	**101**	**$310**

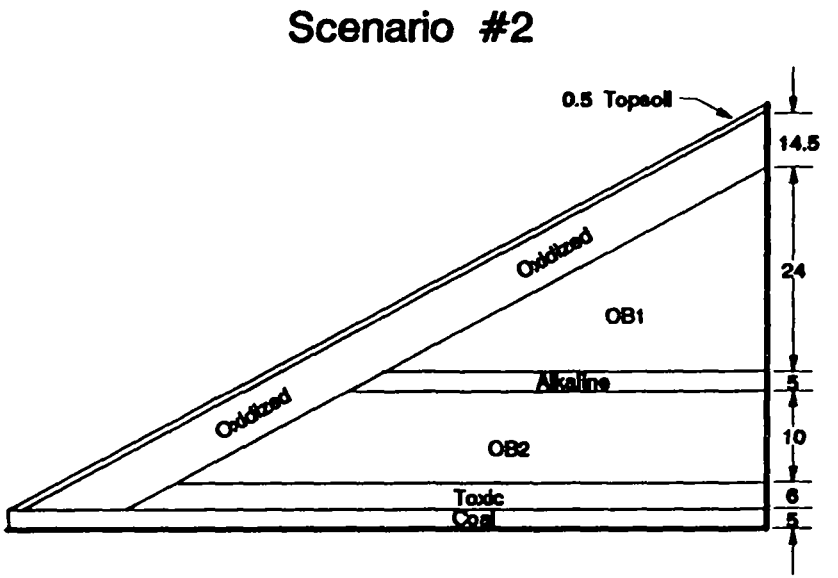

Figure 5. Undisturbed geologic cross-section for scenario 2 (all numbers in feet). This example has a 6-ft layer of toxic material overlying the coal bed and a 5-ft layer of alkaline material higher in the overburden. OB1 and OB2 are abbreviations for overburden sections 1 and 2.

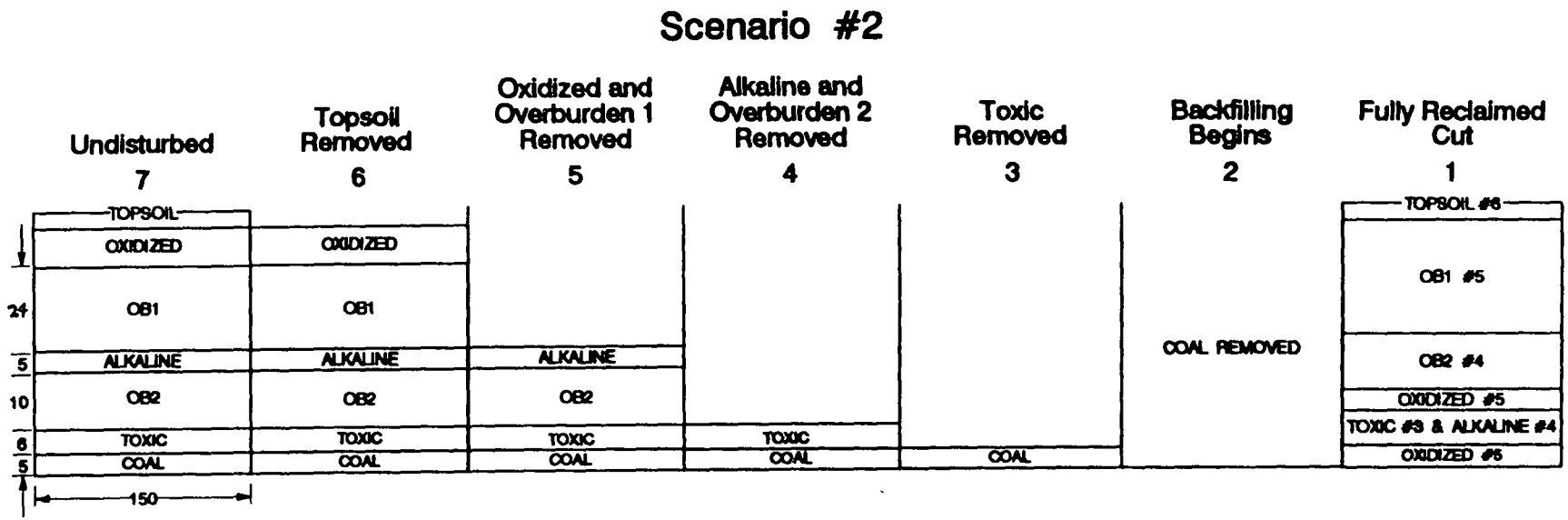

Figure 6. Simplified view of the mining process of a contour operation (scenario 2) where toxic and alkaline layers are being selectively handled and blended during backfilling. Cut 1 is fully reclaimed shows the material and pit number used for backfilling. Under scenario 2, five pits are open.

Table 5. Estimated additional costs for rock trucks in scenario 2 having toxic and alkaline layers that are selectively handled.

	Cubic Yards	Truck Cycles	Total Time (min)	Increase For Selective Handling (min)	Cost of Increase ($104/hr)
Oxidized (8.0 min)	9,750	250	2000	250	$433
Toxic (7.0 min)	3,607	92	644	0	$0
Alkaline (7.5 min)	2,930	75	563	38	66
OB1 (8.0 min)	4,118	106	848	106	184
OB2 (7.5 min)	4,857	125	937	63	109
Topsoil (8.5 min)	394	10	85	10	$17
Total	**25,656**	**658**	**4721**	**467**	**$809**

Scenario 3

The final scenario has only one stratum that requires special handling: a 0.4-m (1.4-ft) toxic layer located 4.6 m (15 ft) up from the coal bed (Figure 7). The complication in this scenario arises from the need to blast in two stages to allow special handling of the toxic layer and to avoid fracturing it into small pieces. Like scenario 1, this case also increases rock truck haul times because of the need for one additional cut, and also involves 20 percent more front-end loader time when handling toxic material.

Figure 8 shows the mining process in six stages. Cut 1 was reclaimed in the following manner. First, part of the oxidized material from cut 4 was placed on the pavement, followed by placing the toxic material from cut 3 and the remaining oxidized material from cut 4 on top. Backfilling of cut 1 continues with the lower overburden layer (OB2) from cut 3, and then the upper overburden layer (OB1) from cut 4. Finally, backfilling is concluded with the topsoil layer from cut 5.

The total and incremental costs of the front-end loader and rock trucks are given in Tables 6 and 7. The methodology is identical to that used in the earlier scenarios. Total front-end loader time per 46-m cut increased by only 6 minutes (due to the small volume of toxic material) and rock truck time increased by 221 minutes. Total cost increased by $401 per 46 by 43 m cut, or $2,025 per ha ($818 per acre).

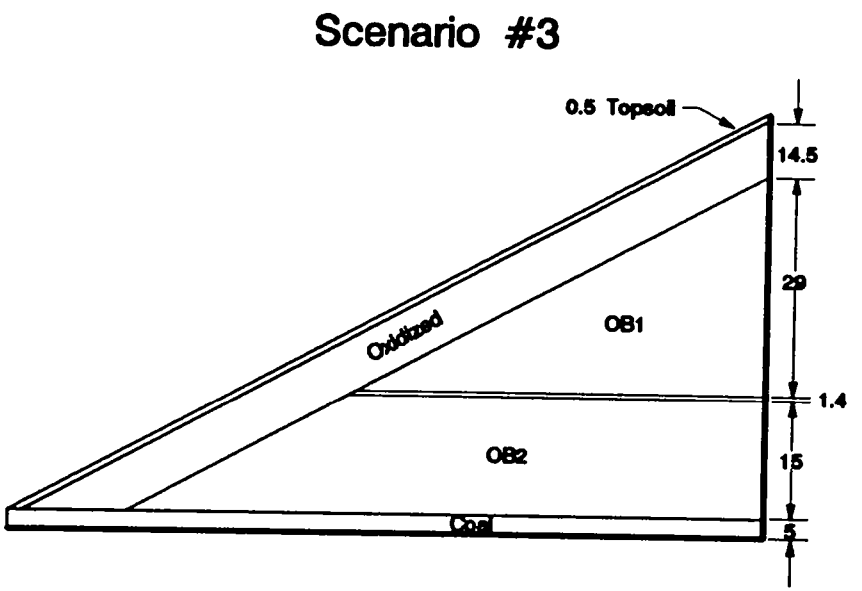

Figure 7. Undisturbed geologic cross-section for scenario 3 (all numbers in feet). This overburden is separated by a small toxic layer in the middle of the overburden. OB1 and OB2 are abbreviations for overburden sections 1 and 2.

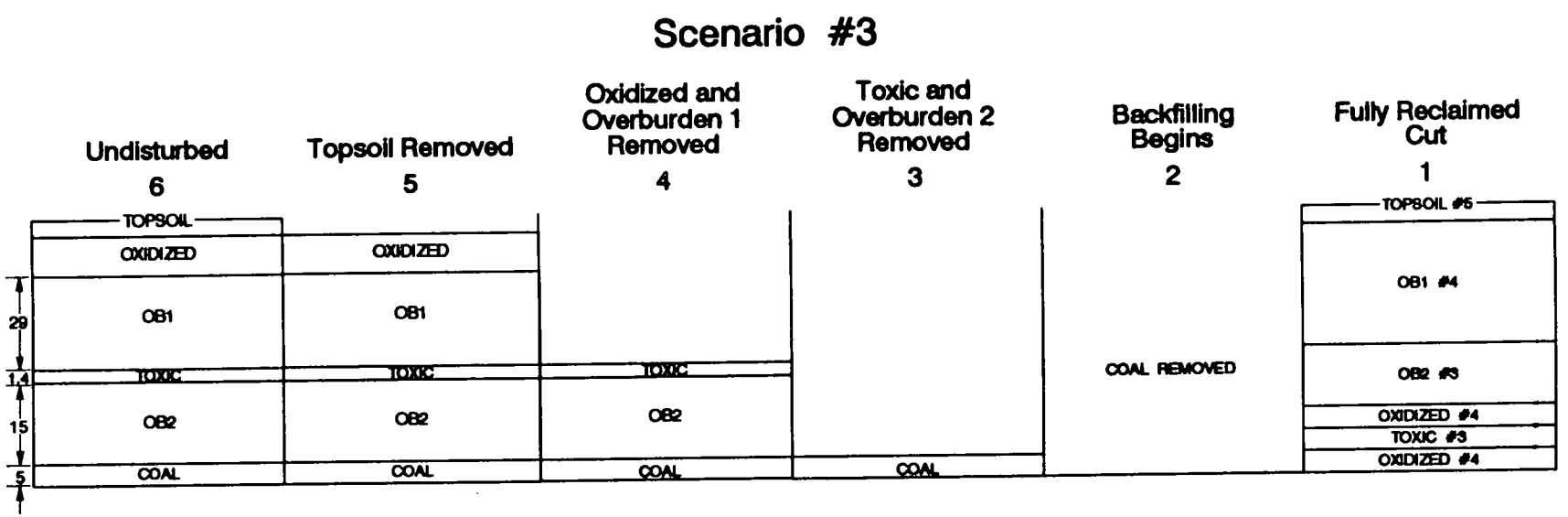

Figure 8. Simplified view of the mining process for scenario 3. This example shows a toxic layer being selectively handled and the remainder of the overburden being removed in two phases. Cut 1 is fully reclaimed and shows the material and pit number used for backfilling.

Table 6. Estimated additional costs for front-end loader in scenario 3 having a 0.4-m (1.4-ft) toxic layer.

	Cubic Yards	Loading Cycles	Total Time (min)	Increase For Selective Handling (min)	Cost of Increase ($184/hr)
Oxidized (1 min)	10,787	830	830	0	0
Toxic (1.2 min)	415	32	38	6	$18
OB1 (1 min)	6,013	463	463	0	0
OB2 (1 min)	8,047	619	619	0	0
Topsoil (1 min)	394	30	30	0	0
Total	**25,656**	**1,974**	**1,980**	**6**	**$18**

Table 7. Estimated additional costs for rock trucks in scenario 3 having a 0.4-m (1.4-ft) toxic layer.

	Cubic Yards	Truck Cycles	Total Time (min)	Increase For Selective Handling (min)	Cost of Increase ($104/hr)
Oxidized (7.5 min)	10,787	277	2,078	139	$241
Toxic (7.0 min)	415	11	77	0	$0
OB1 (7.5 min)	6,013	154	1,155	77	133
OB2 (7.0 min)	8,047	206	1,442	0	0
Topsoil (8 min)	394	10	80	5	$9
Total	**25,656**	**658**	**4,832**	**221**	**$383**

Conclusion

This paper presented three selective handling situations and estimated the additional costs associated with extra front-end loader time and rock truck time in these situations. Based on a toxic layer just above the coal and the necessity for placing that material on a layer of oxidized material, selective handling required the opening of an additional 46-m (150-ft) wide pit (compared to a non-selective handling plan) and cost an additional $663 per pit or about $3,348 per ha ($1,353 per acre) to reclaim. Other scenarios with more complex overburden handling schemes increased the costs of selective handling to $1,119 per pit or $5,652 per ha ($2,284 per acre) to reclaim. In actual mining situations, the costs of selective handling would increase if the toxic material were distributed more widely or unevenly throughout the overburden and may thereby make the cost of mining coal on the site too costly. The cost estimates do not include additional management costs, additional blasting costs, or training costs for equipment operators. In a given situation, each of these factors could add to the costs of selective handling.

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SELENITE-SELENATE SPECIATION IN MINE SOILS AND SALT SOLUTIONS: A COMPARISON OF ATOMIC ABSORPTION SPECTROSCOPY AND ION CHROMATOGRAPHY

by

Shankar Sharmasarkar, George F. Vance, and Florence Cassel-Sharmasarkar

Abstract. Coal mining activities may enhance bioavailability of selenium (Se) species in soils, which can subsequently cause toxicity and contamination problems. Information on comparative applications of spectroscopy and chromatography for selenite (SeO_3^{-2}) - selenate (SeO_4^{-2}) speciation in coal mine soils is limited. Aqueous extracts (triplicates) of five soil samples, collected from reclaimed coal mine sites in the Powder River Basin, Wyoming, and a set of SeO_3^{-2}-SeO_4^{-2} salt solutions (0.5, 1, 5, 10 and 25 mg/L, in triplicates) were speciated using atomic absorption spectroscopy with hydride generation (AAS-HG) and ion chromatography (IC). The objective of this study was to compare these two methods for Se speciation. The results indicated that AAS-HG was capable of analyzing very low Se concentration which could not be detected by IC. Presence of excessively high concentrations of SO_4^{-2} affected chromatographic Se speciation, either by shifting or overlapping Se peaks. For such cases, AAS-HG was more useful than IC. However, IC was capable of speciating aqueous SeO_3^{-2} - SeO_4^{-2} directly without any sample pretreatment, whereas AAS-HG measured SeO_3^{-2} + SeO_4^{-2}, and SeO_3^{-2} in separate runs and SeO_4^{-2} was calculated from the difference, i.e., spectroscopic speciation was an indirect method. For both Se species, AAS-HG and IC data were comparable within the limit of standard deviation, indicating the reliability of both methods for Se speciation. Thus, chromatographic and spectroscopic techniques could be applied to speciate solution Se; however, each had its own analytical limitations. For some of the speciation data there were some discrepancies, which could be due to mutual interconversions between SeO_3^{-2} and SeO_4^{-2} in the solutions, or integrational error during analysis, or some impurity in the original salts used to prepare the Se solutions. An overestimation of Se concentrations by spectroscopy was observed, which was probably due to the fact that AAS-HG estimated selenite as a component rather than a species. In aqueous soil extracts, SeO_4^{-2} was estimated to be the predominant Se species. The simultaneous use of both the spectroscopic and chromatographic techniques was instrumental in understanding details of Se speciation.

Additional Key Words: Selenium, Coal Mine Soil, Analytical, Contamination.

Introduction

The environmental importance of selenium (Se) has been attributed to its potential to cause either toxicity or deficiency to humans, animals and some plants within a very narrow concentration range (Lakin 1972). Levels as low as 0.01 mg/L are known to cause deformation and death to wildfowl (Ohlendorf 1989). The U.S. Environmental Protection Agency designated 0.01 mg/L Se as the primary drinking water standard (USEPA 1986). Selenium solubility and availability depends on the relative concentrations of

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various species present in soil solutions (Masscheleyen and others 1990), which subsequently can govern biotoxicity or deficiency. Four major Se species, selenide (Se^{-2}), elemental Se (Se^{0}), selenite (SeO_3^{-2}) and selenate (SeO_4^{-2}), can be present in soils and other geological materials (McNeal and Balistrieri 1989).

Selenium has been reported to occur naturally in association with coal mine materials of Wyoming (Sharmasarkar and Vance 1995a). In the western United States, several seleniferous coal mine environments are found where prolonged atmospheric exposure and subsequent areal oxidation of the mined materials may result in oxidized Se species, such as SeO_3^{-2} and SeO_4^{-2} in the soil solutions. Such conditions may often lead to enhanced solubility and bioavailability of these species for plant uptake and aquifer contamination. Thus, it is necessary to understand the suitable methodologies for precise determination of SeO_3^{-2} - SeO_4^{-2} in solutions. Both atomic absorption spectroscopy with hydride generation (AAS-HG) and ion chromato-graphy (IC) have been recognized as useful tools for Se analysis in soil extracts (Blaylock and James 1993); however, the efficiency of each technique may vary depending on the nature of the system under analysis. Information on comparative applications of AAS-HG and IC for SeO_3^{-2} - SeO_4^{-2} analysis in coal mine soils is not available. Therefore, a comparative study of the spectroscopic and chromatographic methods of Se assay is fundamental in the evaluation of Se solubility behavior in coal mine soils. The objective of the current study was to compare spectroscopic and chromatographic methods of SeO_3^{-2} and SeO_4^{-2} assay in aqueous soil extracts and Se salt solutions. The Se salts were used to compare the two methods in relatively homogeneous solutions, in addition to the heterogeneous mine soil extracts. The results of this study will be useful in understanding Se speciation techniques and similar approach can be applied for other contaminant assay.

Materials and Methods

Mine Soil Sampling

Five mine soils (S1, S2, S3, S4, and S5) were used for this study. The samples were collected from reclaimed coal mine sites in the Powder River Basin, Wyoming. The climate of these mine environments is temperate and semi-arid, with an average daily temperature of 5-90 °F, and an annual precipitation range of 11-18 inches. The geology of the area is comprised of Fort Union and Wasatch formations of early tertiary age, and consists of continental type sediments deposited in fluvial, lacustrine and swampy environments. The lenticular strata consist of alternating sand, silt and claystone, with occasional coal beds (Naftz and Rice 1989). Mine soil samples were air-dried, finely ground (<2 mm), and stored at room temperature in polyethylene bags until analysis.

Soil Extraction

Water extracts of the mine soils were prepared by shaking samples with distilled deionized water (solid to solution ratio = 1:2) on a reciprocating shaker (Eberbach Corporation) at 180 cpm for 24 h followed by heating (90 °C) in a waterbath for 30 min. After cooling to room temperature (24±1 °C) the suspensions were centrifuged (International Equipment Company, Model K) at 2500*xg* for 15 min followed by filtration through 0.45μ glass fiber filters. We used water extracts because aqueous Se is a potential concern to the Wyoming Department of Environmental Quality - Water Quality Division (WDEQ-WQD 1993) that has defined 0.01, 0.05 and 0.05 mg/L Se levels in groundwater as marginal levels for Class I (domestic), II (agricultural), and III (livestock) standards, respectively. Details on Se extraction procedures have been discussed by Spackman and others (1994). One sample (Id. S5) that had very low concentrations of SeO_3^{-2} - SeO_4^{-2} (<0.01 mg/L), was spiked with 15 mg/L SeO_3^{-2} - SeO_4^{-2} solution, and used as a check for this study.

Preparation of Selenite-Selenate Salt Solutions

Measured quantities of sodium salts of selenite (Na_2SeO_3) and selenate (Na_2SeO_4) were dissolved in distilled deionized water to prepare 0.5, 1, 5, 10 and 25 mg/L SeO_3^{-2}-SeO_4^{-2} solutions. The salts were purchased from Aldrich Chemical Company, Inc., Milwaukee, WI. Between analysis, the soil extracts and salt solutions were stored in polythene bottles at 4 °C.

Table 1. Analytical Parameters for SeO_3^{-2} and SeO_4^{-2} Analysis by Atomic Absorption Spectrophotometry and Ion Chromatography.

Parameters	Atomic absorption spectrophotometry	Ion Chromatography
Detection mode	Continuous absorbance	Suppressed electrical conductivity
Instrumental settings	Hollow cathode lamp (Ne gas, 15 mA current), Quartz cell, 196 nm wavelength, 0.7 nm slit width	Ion exchange separator column (AS9), Guard column (AG9), 50 µL loop
Flame	Air-acetylene (oxidizing); Flow rate: acetylene (20), air (50); Pressure: air (500 kPa), acetylene (85 kPa)	--------
Reagents	Reducing agent: 0.8 M $NaBH_4$ - 0.25 M NaOH; Sample matrix: 6 M HCl	Mobile phases: 5.4 mM Na_2CO_3 - 5.1 mM $NaHCO_3$, water, Flow rate: 2 mL/min; Regenerant solution: 0.013 M H_2SO_4, Flow rate: 5 mL/min
Carrier gas	N_2 (350 kPa)	N_2 (700 kPa), He (120 kPa)
Integration time	10 s	10 min
Boundary limits	Se detection: 0.001 mg/L, Se quantification: 0.1 mg/L	Se detection: 0.01 mg/L, Se quantification: 40 mg/L

Selenite-Selenate Speciation by AAS-HG and IC

Concentrations of SeO_3^{-2} and SeO_4^{-2} in the aqueous soil extracts and salt solutions (triplicates) were compared by both spectroscopic and chromatographic methods. Solutions having SeO_3^{-2} - SeO_4^{-2} concentrations greater than the limits of quantification were diluted before analyses. In the spectroscopic method, total SeO_3^{-2} + SeO_4^{-2} in solution was analyzed after a HCl digestion using a continuous flow hydride generator (Varian, Model VGA-76) attached to a Perkin-Elmer (Model 2280) atomic absorption spectrophotometer (AAS-HG) (Spackman and others 1994). Solution SeO_3^{-2} concentration was analyzed before HCl pretreatment. Dissolved SeO_4^{-2} was calculated from the difference between these two data. Details of Se speciation using AAS-HG have been described by Cutter (1985) and Fio and Fujii (1990). Simultaneous speciation of SeO_3^{-2} - SeO_4^{-2} was carried out using an ion chromatographic (IC) method with electrical conductivity detection (Dionex 2000i ion chromatography with AI-450 software, version 2.12). The details regarding Se analysis by these methods were described in Blaylock and James (1993). The operational conditions for SeO_3^{-2}-SeO_4^{-2} analysis by both AAS-HG and IC are described in Table 1. In the AAS-HG method, SeO_3^{-2} is reduced to H_2Se by $NaBH_4$-NaOH. The nascent hydride formed due to reduction is then replaced by an inert carrier gas (N_2) and passed through a heated quartz cell. The instrument records an absorbance reading at a Se resonance line corresponding to the wavelength of 196 nm.

In the IC method, different anionic species are retained inside an anion exchange column as the ions pass through. During the passage of the eluent (Na_2CO_3-$NaHCO_3$), ions are released, and the corresponding conductivities are recorded by the instrument relative to the retention times of different species. The conductivity response appears in the form of chromatograms, and concentrations of the ions in a sample solution are calculated from recorded peak areas of standard solutions. For both instrumental analyses, reference SeO_3^{-2}-SeO_4^{-2} solutions were used as standards. However, because of differences in

quantification limits between AAS-HG and IC, two sets of standards were used. Thus, for spectroscopy we used 0 (control), 0.005, 0.01, 0.02, 0.04, 0.08 and 0.10 mg/L SeO_3^{-2} - SeO_4^{-2} solutions; whereas for chromatography concentration levels were 0 (control), 0.05, 0.10, 0.25, 0.50, 1.0, 5.0, 10, 20 and 40 mg/L SeO_3^{-2} - SeO_4^{-2}. These standard solutions were used in triplicate. It should also be noted that in addition to the use of reference solutions, and triplicate measurements as quality controls, the standards and sample solutions were run through identical experimental and instrumental conditions.

Results and Discussion

The regressional relationship for the response of both analytical methods to Se concentrations in standard solutions is shown in Figs. 1a and 1b. For AAS-HG, the response was absorbance, while for IC, peak area was used for Se quantification. The response of both instruments was significantly linear (r 0.99) within the limit of Se quantification. Similar conditions were also described by Blaylock and James (1993). It should be noted that for IC, the slope of the SeO_3^{-2} line was greater than that for SeO_4^{-2}. Ion chromatograms showing conductivity versus time plots for SeO_3^{-2}, SO_4^{-2} and SeO_4^{-2} are described in Fig. 2. Distinct peak separation was observed for different ions when there was no mutual peak interference (Fig. 2a). In such cases AAS-HG is useful for Se speciation.

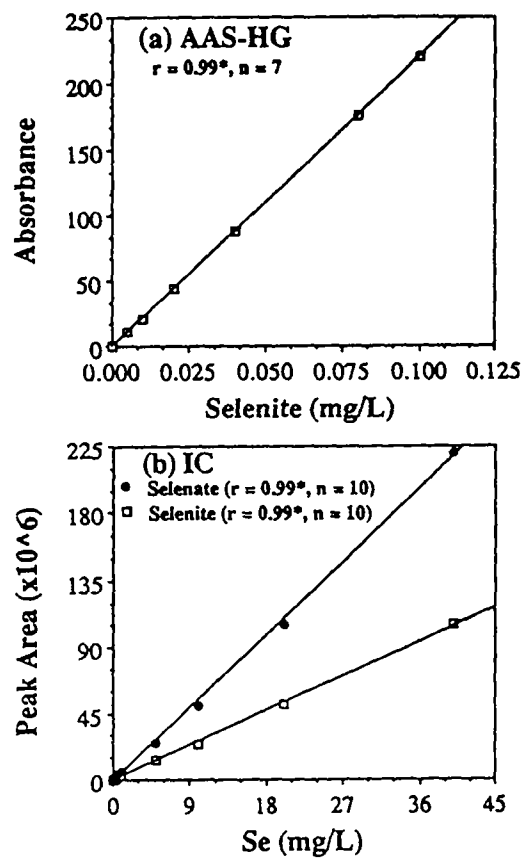

Figure 1. Se quantification by: (a) AAS-HG, (b) IC. (*r = correlation coefficient, n = number of samples, * = significant at P<0.01).

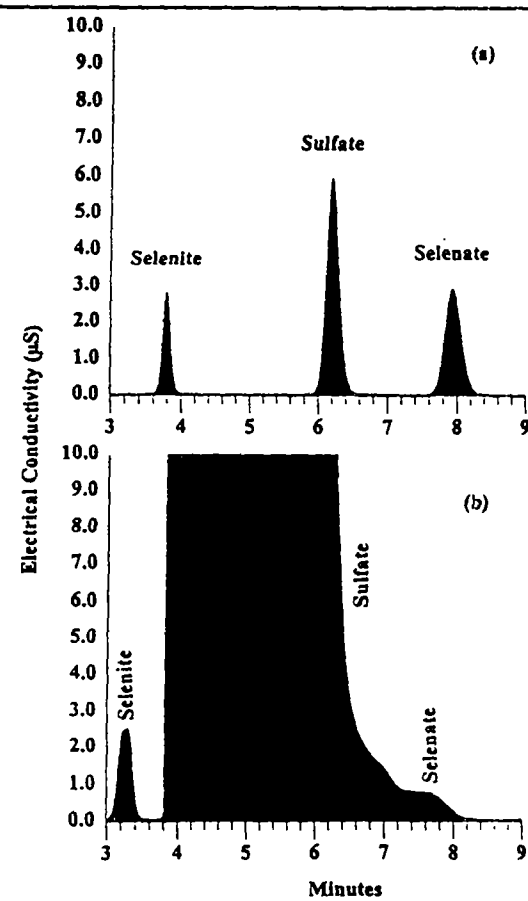

Figure 2. Ion chromatograms: (a) distinct peaks (20 mg/L of each ion), (b) shifting and overlapping of peaks (sample ID. S5+15 mg/L).

Table 2. Spectroscopic and Chromatographic Speciation of SeO_3^{-2} and SeO_4^{-2} in Mine Spoil Extracts and Salt Solutions.*

Samples	SO_4^{-2} (mg/L)	SeO_3^{-2} AAS-HG	SeO_3^{-2} IC	SeO_4^{-2} AAS-HG	SeO_4^{-2} IC
		(mg/L)			
S1	90	0.27 ± 0.02	0.25 ± 0.06	0.90 ± 0.06	0.81 ± 0.05
S2	315	0.18 ± 0.01	0.18 ± 0.08	0.13 ± 0.05	0.13 ± 0.06
S3	85	0.06 ± 0.01	0.04 ± 0.01	0.10 ± 0.01	0.13 ± 0.07
S4	203	0.06 ± 0.01	0.06 ± 0.04	3.42 ± 0.36	3.24 ± 0.60
S5	4270	0.002 ± 0.000	BDL	0.002 ± 0.000	BDL
S5+15 mg/L	4270	14.91 ± 0.23	14.24 ± 0.86	14.88 ± 0.23	ND (PO)
0.5 mg/L	0.5	0.58 ± 0.01	0.50 ± 0.07	0.42 ± 0.03	0.51 ± 0.06
1 mg/L	1	1.09 ± 0.03	1.00 ± 0.08	0.94 ± 0.17	1.00 ± 0.07
5 mg/L	5	5.30 ± 0.04	4.66 ± 0.18	5.33 ± 0.67	4.88 ± 0.38
10 mg/L	10	11.65 ± 0.10	9.35 ± 0.17	9.31 ± 1.35	9.43 ± 0.08
25 mg/L	25	26.66 ± 0.69	24.40 ± 0.53	23.99 ± 2.30	24.28 ± 0.35

* Values are mean of three replications ± SD (standard deviation). AAS-HG = Atomic Absorption Spectrophotometry with Hydride Generation, IC = Ion Chromatography. BDL = Below Detection Limit, ND = Not Detected, PO = Peak Overlap.

According to the retention time of each species, the SeO_3^{-2} peak appeared first (3.77 ± 0.40 min) followed by SO_4^{-2} (6.02 ± 0.40 min) and SeO_4^{-2} (7.72 ± 0.40 min). However, extremely high concentration of SO_4^{-2} (in this case 4270 mg/L) caused a noticable shift in the SeO_3^{-2} peak toward a lower retention time (3.28 min) and masked the SeO_4^{-2} peak, subsequently resulting in a lack of SeO_3^{-2} - SeO_4^{-2} quantification (Fig. 2b). Shifting or overlapping of different peaks is often found to be a limiting factor for Se speciation by IC.

Spectroscopic and chromatographic speciation of SeO_3^{-2} and SeO_4^{-2} in mine soil extracts and salt solutions is shown in Table 2. For both species, the AAS-HG and IC data were comparable within the limit of standard deviations, which indicated the reliability of both methods for Se speciation. For some of the salt solutions, the detected concentrations of SeO_3^{-2} and SeO_4^{-2} did not exactly correspond to the experimental concentrations used, though the sum of SeO_3^{-2} and SeO_4^{-2} detected matched more closely with the experimental concentrations of SeO_3^{-2} + SeO_4^{-2} used. The slight discrepancy could be due to mutual interconversions between SeO_3^{-2} and SeO_4^{-2} in solutions, or integrational error during analysis, or some impurity in the original salts used to prepare the solutions. Similar observations were also made by Blaylock and James (1993). For most samples including both the soils and salts, a comparison of AAS-HG and IC data also indicated there was either slight overestimation of concentrations by spectroscopy or minor underestimation by chromatography. This was probably due to the fact that AAS-HG estimated selenite as a component rather than a species, i.e., some other selenite ions (for example, biselenite) in addition to SeO_3^{-2} might have been recorded by the instrument, whereas, IC recognized only SeO_3^{-2} and SeO_4^{-2} as species, not as components. It should be noted that for the sample S5, the IC method was not useful in speciating Se at very low level (<0.01 mg/L), whereas AAS-HG was able to detect as low as 0.002 mg/L Se. Another limitation of the IC method was peak overlapping (sample: S5+15 mg/L), where SeO_4^{-2} peak was masked by a large SO_4^{-2} peak. These are important findings with respect to the environmental quality standards.

In this sample the SeO_3^{-2} peak shifted because of the high SO_4^{-2}

concentration, which, however, was corrected by adjusting the retention time for SeO_3^{-2} during the data reprocessing in the AI-450 software. Thus, AAS-HG was capable of speciating all the samples, whereas IC could speciate Se only in samples with no SO_4^{-2} interference. Again it should be noted that IC can speciate Se in the same run unlike AAS-HG which requires separate runs and some sample pre-treatments. Finally, a close review of the data suggested that SeO_4^{-2} was mainly the predominant species in the aqueous soil extracts, which conformed with the observations from a Se speciation study previously conducted by Sharmasarkar and Vance (1995b). The knowledge acquired from this study should help in obtaining a broader view regarding spectroscopic and chromatographic analysis of Se as both components and species; we recommend simultaneous use of both methods.

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COMPACTION EFFECTS ON MINELAND SOIL QUALITY[1]

Padam P. Sharma, and Fredric S. Carter[2]

Abstract. Management of water is vital to sustenance of biomass production and environmental quality of reconstructed minelands in semi-arid North Dakota. Since water resides and moves through soil pores, quantification of soil pore characteristics is a key to evaluate soil quality for reclamation success. We measured field saturated hydraulic conductivity (K) by double-ring ponded infiltration method on 26 premine and postmine profiles. The same profiles, each installed with a neutron probe access tube and a set of five tensiometers, were used to monitor post-infiltration redistribution of water by measuring volumetric water contents and matric potentials with time. An empirical equation was fitted to the soil water retention data to derive proportional distribution of various sizes of pores, maximum water capacity, median pore size, and a depth weighted pore index (Ω) to compare efficiency of each profile to transmit and retain water. The undisturbed premine soils showed high Ω's and high proportion of effective pores (ε) with radii > 5 μm. The reclaimed mineland profiles show low Ω and general deficiency of pores effective in transporting water to deeper depths. The pooled data from the premine and postmine soil profiles included in this study show that K increased linearly with Ω and ε^2. We recommend that the K - ε relationship be further evaluated by measuring infiltration and pore size distribution from various locations under different management conditions. If feasible, the easily measurable, profile scale, K - ε relationship provides a quantitative tool to evaluate soil quality of premine and reclaimed soils.

Additional Key Words: Land Reclamation, Soil Structure, Soil Hydrology

Introduction

About 800 - 1000 hectares of prairie landscapes in western North Dakota are annually disturbed for surface mining of lignite coal. Following the Surface Mine Control and Reclamation Act (U. S. Congress, 1977) guidelines, federal and state regulations on surface mining mandate that the disturbed lands be reconstructed to a level of productivity and environmental quality equal to or better than before mining. To achieve this goal, previously stripped non-sodic subsoil (original lower B and upper C) and topsoil (original A and upper B) materials are respread over reshaped spoil. The depth of respreading depends upon the physical and chemical properties of the spoil. Scrappers are generally used to respread the materials at about 15-cm increments until the desired depths of subsoil and topsoils are achieved (Halvorson et al., 1986). Vegetation is re-established and, following the SMCRA guidelines, the quality of reclaimed soils are assessed on the basis of respread soil depth and biomass yield.

In the undisturbed Mollisols of western North Dakota, the natural prairie vegetation and associated granular soil structure supports rapid movement of water through well-developed inter-aggregate voids and biologically created macropores. When these soils are mined, the existing network of soil pores is destroyed.

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During land reclamation after surface mining, the frequent and heavy vehicular traffic used in spreading and leveling operations cause compaction resulting an abundance of fine sized inter-aggregate pores. As a result, reclaimed soils lack the effective pore volume and continuity needed for adequate distribution of water and roots (Chong et al., 1986). Potter et al. (1988) compared pore volumes of constructed mine soils of 4 and 11 years after reclamation with undisturbed soils. They observed that mesopore volume (pore radii > 15 μm) of reclaimed soils and associated hydraulic conductivities, especially of the subsoil layer, were significantly lower than the undisturbed soils.

Soil quality relates to the ability of soil to function as medium for plant growth, as partitioner and regulator of water, energy, and solute fluxes in the landscape and biosphere, and as filter and buffer for inorganic and organic pollutants (Larson and Pierce, 1994). A good quality soil sustains onsite productivity, efficiently utilizes available water and nutrients, and minimizes offsite damages to water and air quality. By replacing suitable soil material to specified depths, the current reclamation mitigation package seeks to reconstruct the rootzone and achieve reference area biomass productivity. However, it ignores the role of the root zone pore space to harvest precipitation water for maximum plant growth. While the current technology tries surficial control of overland flow and effluents to minimize onsite and offsite damages, it fails to stress the role of the reconstructed root zone pore structure to achieve hydrologic balance in the watershed.

Larson and Pierce (1994) favor the use of a dynamic systems approach to assess soil quality by evaluating a management system in terms of its actual performance in the watershed rather than an output comparison with other systems. A systematic observation of changes in soil quality parameters and their behavioral indications in energy, water, sediment, and solute fluxes in the watershed should be the proper approach to monitor the success of reclamation. Since water is the key integrator of these fluxes and water resides and moves through soil pores, we propose that characterization of pore structural aspects of root zone soil quality is fundamental to the restoration and sustenance of agricultural productivity and environmental quality of restored ecosystems

To understand the effect of root zone soil characteristics on distribution of water on reclaimed lands vis-à-vis that of premine lands and pre-law abandoned minelands, we measured infiltration, water retention and seasonal variability of soil water contents on surface coal minelands of North Dakota from 1989 to 1993. The data on seasonal distribution of soil water (Sharma et al., 1993) and infiltration experiments (Sharma et al., 1994) were presented elsewhere. In this paper, we report on water retention characteristics and infer the profile scale pore structural attributes of soil quality.

Materials and Methods

Constant head, ponded infiltration was measured by double ring method (Bouwer, 1986) on premine lands, abandoned minelands, and reclaimed mineland profiles at Indian Head (T144N R89W sec. 24 and 25) and Velva (T152N R81W Sec. 27,28, 33 and 34) mines in west-central North Dakota. Prior to water application, a neutron probe access tube was installed at the center of the inner ring using a hydraulically driven core sampler of equivalent diameter to the access tube. Additionally, side walls of the access tube hole was carefully repacked by pouring loose soil material around the tube and tamping with a small metal rod. Marriotte bottles were used to supply constant head of water in the rings and infiltration measured for 2 to 3 hours. The infiltration volume vs. time data were fitted with infiltration equations to evaluate the parameters of sorptivity and field saturated hydraulic conductivity (Sharma et al., 1994).

After about 15-22 days of the first infiltration run, tensiometers were installed inside the inner ring on 26 of a total of 40 profiles used for infiltration. The tensiometers were fitted with high flow (air-entry pressure = 100 kPa) porous ceramic cups and

installed at depths of 15, 30, 46, 61, and 76 cm. Before installation, each tensiometer tubing was plastered with wet bentonite, and installation, the space around the tensiometer tubing was back filled soil and repacked to prevent preferential flow.

The tensiometers were filled with de-aired, distilled water and closed with rubber septa. The hydraulic potential was measured with a transducer readout system by inserting a hypodermic needle through the septum (Marthaler et al., 1983). A second infiltration run was conducted until the deepest tensiometer indicated a steady state reading. Water supply was then stopped, and the profile covered with mulch and heavy plastic to prevent evaporation. After about 30 days of redistribution, the mulch and the plastic were removed to allow free evaporation.

During the redistribution process, measurements of neutron probe and tensiometers were recorded at short time intervals in the beginning then daily, and weekly, at later times. The measurements were discontinued when the top two tensiometers stopped functioning due to air-entry into the ceramic cups. At the end of the experiment of about 45 to 60 days, disturbed samples taken at the approximate depth of the tensiometer locations were used to determine 1500 kPa water contents in the laboratory using a pressure membrane apparatus (Klute, 1986).

The relationship between the volumetric water contents and the corresponding matric suctions at each tensiometer depth is represented by the following two parameter closed form equation (van Genuchten, 1980):

$$\Theta = (\theta - \theta_r)/(\theta_s - \theta_r) = [(1 + (\alpha h)^n]^{(1/n-1)} \quad (1)$$

In the above equation, Θ is relative saturation; θ is water content, m^3/m^3; θ_r is residual water content, m^3/m^3, θ_s is near saturation water content, m^3/m^3; and h is matric suction in kPa. For this analysis, we used θ_r equal to the laboratory measured 1500 kPa water content and θ_s equal to 85% of the total porosity (calculated from bulk density of the samples) to approximate incomplete satiation under field conditions. The θ_s and θ_r only represent the upper and lower limits of the fitted curve, and hence, do not have a physical meaning. The curve shape parameters, α and n are derived by nonlinear curve fitting of Eq. [1] to the measured data. For freely draining undisturbed and abandoned mineland profiles, Eq. [1] fitted the data fairly well with nonlinear regression coefficients > 0.95. For tensiometers at deeper depths of reclaimed soils and at highly sodic-clay profiles, the fitting of data was erratic with nonlinear $R^2 > 0.70$.

From Eq. [1], the rate of water desorption ($d\theta/dh$), defined as the water capacity (ϕ, 1/ kPa) is calculated as (Horton and Chung, 1991):

$$\phi = (n-1)\,(\theta - \theta_r)\,[1 - \Theta^{\{n/(n-1)\}}]\,/h \quad (2)$$

The matric suction (h) at which the ϕ is maximum (ϕ_{max}) is converted to corresponding median pore size (η, μm) by dividing 150 with h. The constant of 150 comes from the capillary equation between matric suction (kPa) and capillary radius (μm) at 20° C (Gupta et al., 1989). The median pore size is assumed to represents the size of pore at which most of the water at the point of measurement will be released due to forces of gravity, root extraction, or upward flux due to evaporative demand of the surface (Hillel, 1980). From N tensiometer measurements at the various depths, a depth weighted pore index (Ω) of the profile is calculated by cumulating the product of η and ϕ_{max} for each depth of measurement:

$$\Omega = 100/z \sum_{T=1}^{N} \eta\, \phi_{max} \quad (3)$$

Here, T is the count of tensiometers and z is the depth of the deepest tensiometer, cm. The factor of 100/z is used to extrapolate the Ω value to a meter of root zone depth. The Ω index is used to assess the relative depth integrated capacity of the various profiles to desorb water. Hence it is a measure of profile scale soil pore structure.

From the soil water retention data measured at each tensiometer depth, we derive

pore size distribution using the following pore classification scheme (Table 1).

Table 1: Classification of pore sizes

Pressure head (kPa)	Pore radii (μm)	Pore class
< 3	> 50	Macropores
3 - 30	5 - 50	Mesopores
30 - 1500	0.1 - 5	Micropores
> 1500	< 0.1	Residual pores

Note: The proportion of macropores and meso-pores (volume of pores with radii > 5 μm) are collectively called effective pores. Traditionally, the volume of water held in the effective pores is defined as drainable water, and that in the micropores is called available water.

Results and Discussion

To demonstrate the effect of compaction on water retention and water capacity curves, we purposefully chose six typical premine and postmine profiles from the data set. Table 2 lists the location, major characteristics, hydraulic conductivity (K), and depth averaged median pore size (η) and weighted pore index (Ω) of the six profiles. The undisturbed (Z215I, V116) and the compacted (Z511, Z004) premine soils mainly consist of the Flaxton - Williams association (Typic haploborolls) with wind and water deposited sediments overlying glacial tills. The root zone texture ranges from sandy loam to clay loam with undisturbed profiles having a well developed aggregated structure due to dominance of natural prairie vegetation. The topsoil and subsoil layers of the reclaimed profile at Indian Head (Z503), consisting of sandy loam to loam materials, are underlain by loam to silty clay, sodic minespoil materials. The profile at Velva (V102) shows a typical pre-SMCRA characteristics with haphazard respreading of overburden materials.

Table 2: Site characteristics and pore structural indices of six representative premine and reclaimed soil profiles.

Mineland Type	Location	Dominant Characteristics	$\eta^{\dagger}$ (μm)	$\Omega^{\ddagger}$ (μm/kPa)	$K^{\S}$ (cm/h)
Premine undisturbed	Z 215I	sandy loam, natural prairie	30 (12.7)	5.42	26.33
	V116	clay loam, natural prairie	46 (8.0)	7.91	11.82
Premine compacted	Z511	vehicle traffic, hay	26 (6.9)	0.85	1.54
	Z004	vehicle traffic, grazing	24 (11.8)	0.73	0.50
Postmine reclaimed	Z503	post-SMCRA, hay	21 (14.4)	0.93	1.64
	V102	pre-SMCRA, grazing	14 (3.3)	0.08	1.82

† Median pore size, ‡ Weighted pore index, § Satiated hydraulic conductivity. The numbers in parenthesis are standard deviations.

Representative Water Retention Curves

Figure 1 shows the soil water retention characteristics with depth of the two undisturbed prairie soils from the Indian Head (Z215I) and Velva mine (V116). The sigmoid-shaped curves show the fitted water content versus pressure head relationships (Eq. [1]), and the bell shaped curves represent the derived specific water capacity versus pore size relationships (Eq. [2]) with depth. The curve shapes are more pronounced on the well aggregated clay loam profile (V116) than the sandy loam profile (Z215I). Table 2 shows that the depth weighted pore index (Ω) and hydraulic conductivity of these two profiles are significantly higher than those of premine compacted and reclaimed soils.

Examples of water retention and specific water capacity curves of compacted premine soils in the vicinity of minelands are shown in Fig. 2. These profiles were subjected to heavy vehicular traffic during mining in the 1970's. After mining was completed in the early 1980's, profile Z511 was managed with alfalfa hay while the profile Z004 was used for intensive grazing. Compared to the undisturbed profiles (Fig. 1), the retention and specific water capacity curves of the compacted profiles (Fig. 2) are flatter. Compaction has decreased the total pore volume at the cost of inter-aggregate effective pores and increased the volume of intra-aggregate residual pores. Compared to the undisturbed profiles shown in Fig. 1, the weighted pore indices (Ω) of these two compacted profiles were < 1.0.

Figure 3 shows the soil water retention characteristics and specific water capacity curves of the reclaimed profiles listed in Table 2. The Z503 profile at the Indian Head mine was reclaimed under the SMCRA guidelines in the early 1980's and restored with grass-legume hay crop. The bulged shape of the 15-cm water capacity curve indicates that the topsoil of Z503 has an improved pore structure than subsoils at deeper depths. Compared to those of the undisturbed profiles which show pronounced curve shapes throughout the depth (Fig. 1), the deeper depths of this reclaimed profile has flatter shapes indicating poor structure development. The Z503 profile has a Ω index of 0.93 which is comparable with the compacted premine profiles (Table 2).

The profile at Velva (V102) was not reclaimed as per the SMCRA guidelines, and since the vegetation reestablishment, the land is used for grazing. This overgrazed, reclaimed profile at Velva (V102) shows a severe lack of porosity with depth. Despite overnight saturation, the deeper tensiometers in this profile did not show any response during infiltration and during subsequent redistribution process. The Ω index of 0.082 calculated from the three remaining tensiometers weighted for 1.0 m of root zone depth (Eq.[3]) indicates a massive structure.

Pore Size Distribution

The depth averaged pore size distribution of the 26 different profiles for which soil water retention characteristics were measured in the field are shown by bar diagram in Fig. 4. The number at the bottom of each bar represents the average total pore volume of which the proportion of macropores, mesopores, micropores, and residual pores are shown in shades. The number at the top of each bar denotes the weighted pore index (Ω) of each profile.

In general, the six undisturbed premine profiles and the two non-sodic profiles (Z213, Z214) in the abandoned minelands show higher pore volumes. In these profiles, macropores and mesopores (effective pores) constitute about 50% of total pore space. The comparatively higher Ω index of these profiles signify a pore structure with potential to efficiently transport and store precipitation and runon water.

Compared to the well structured soils, the total pore volumes and Ω indices of the four premine compacted soils and the six reclaimed soils are smaller in magnitude. The decrease in total pore volume due to compaction is mainly due to a decrease in the proportion of macropores and mesopores. Only about 25% of the total pore volume is occupied by effective pores. Of

the four premine compacted soils shown in Fig. 4, Z511 is the farthest from the mine area with a Ω index of 2.45. The Ω indices of other premine compacted and reclaimed profiles are < 1.0. The abandoned minelands, on the other hand, show high total pore volume and a wide range of Ω values. Though not compacted, the ability of abandoned mineland profiles to transport and retain water, as signified by the Ω index, is limited by the location of sodic clay materials in the root zone. The three sodic spoil profiles (Z211, Z703, V205) have the lowest Ω and hydraulic conductivities.

Hydraulic Conductivity vs. Pore Indices

The structural signature of a soil profile is typified by its pore characteristics defined by the number and size of pores, their continuity, tortuosity and stability. The entry and rate of movement of water and its retention in soil are dependent upon the depth integrated pore characteristics of the profile. While effective porosity is a measure of the volume of large size pores, the hydraulic conductivity is a measure of the continuity and tortuosity of these pores. A relationship between the two parameters is an indicator of relative pore structure of various premine and postmine soils with respect to their ability to transport water.

We regressed the hydraulic conductivity (K) calculated from ponded infiltration experiments (Sharma et al., 1994) against the volume of effective pores (ε) and the weighted pore index (Ω) derived from water retention experiments. The K-ε relationship was derived by nonlinear curve fitting of the form $y=ax^b$, of which, the exponent 'b' was 2.02. For clarity of interpretation, we refitted the data with $K=a\varepsilon^2$. The K-Ω relationship was derived by linear regression.

Figure 5 shows that even for data from various heterogeneous premine and postmine soil profiles, K increased with ε^2 and the Ω index. Also shown in Fig. 5 inset, the best estimates for K (cm/h) for the data set included in this study are as follows:

$$K = 1.276 \times 10^{-4}\, \varepsilon^2 \qquad (r^2 = 0.86) \qquad [4]$$

$$K = 1.243 + 1.302\ \Omega \qquad (r^2 = 0.81) \qquad [5]$$

The intercept in Eq. [5] is not significantly different from 0. Of the 26 field measurements included in this regression, there were three outlier data from infiltration into coarse textured undisturbed and nonsodic abandoned mineland profiles. These profiles contained big desiccation cracks and numerous colonies of ants; in comparison to their effective porosities, these profiles had excessive infiltration rates.

Figure 5 demonstrates how a generalized profile scale K- ε or K- Ω type relationship is useful in comparing the pore structural qualities of premine and postmine profiles. For simplicity of measurement, we recommend that the K-ε relationship methodology be further evaluated with measurements on additional premine and postmine soils from different locations and under various management systems. As K can be easily measured from the steady state infiltration rate, ε can be calculated from the measurement of volumetric water content at 30 kPa (near field capacity) from undisturbed samples in the laboratory. On the other hand, development of the K-Ω relationship requires expert knowledge; it is time consuming, and the data is highly dependent upon precision and robustness of tensiometric instrumentation used under field environments.

Conclusions

The data, in general, show that the undisturbed soils have high effective porosities and associated hydraulic conductivities. The pore structural attributes of abandoned minelands show the widest range of variability depending upon the location of sodic spoil material in the profile. The post-SMCRA reclaimed profiles with subsoil and topsoil replacements show improved pore structure than when the sodic spoil materials are respread alone. However, the structural quality of these reclaimed profiles was only comparable to compacted premine soils and at a level significantly lower than that of the undisturbed premine profiles.

The data in this study and previous studies on soil structure of reclaimed soils in North Dakota and elsewhere indicate that soil structure regeneration due to natural processes of wetting-drying and freezing-thawing is extremely slow to nonexistent. With the current land reclamation technology and grass-legume hay crop as postmine land use for a decade or more seems to have improved the pore structure to a depth of about 0.3 m. The studies also point out that the deeper depths of reclaimed subsoil show a severe lack of pore space and pore continuity. The repercussion of this lack of adequate pore space in the subsoil results in a restricted root zone with lack of available water during drought and oxygen during floods. For sustained productivity of reclaimed soils, it is recommended that respreading technology, post-reclamation land management, and reclamation evaluation guidelines need to focus on increasing the effective porosity of the entire root zone.

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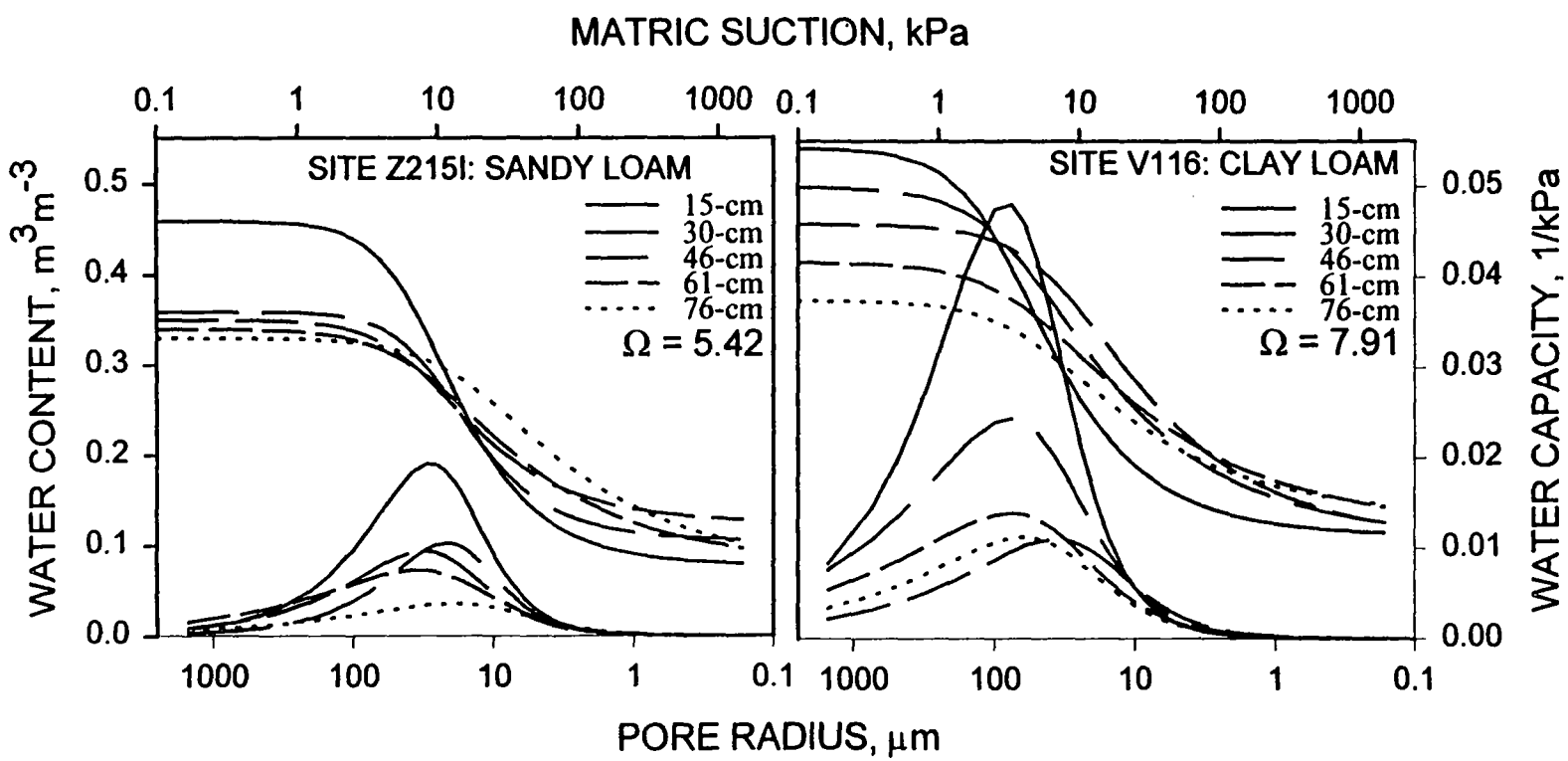

Fig.1: Fitted soil water retention and water capacity curves for two premine undisturbed profiles.

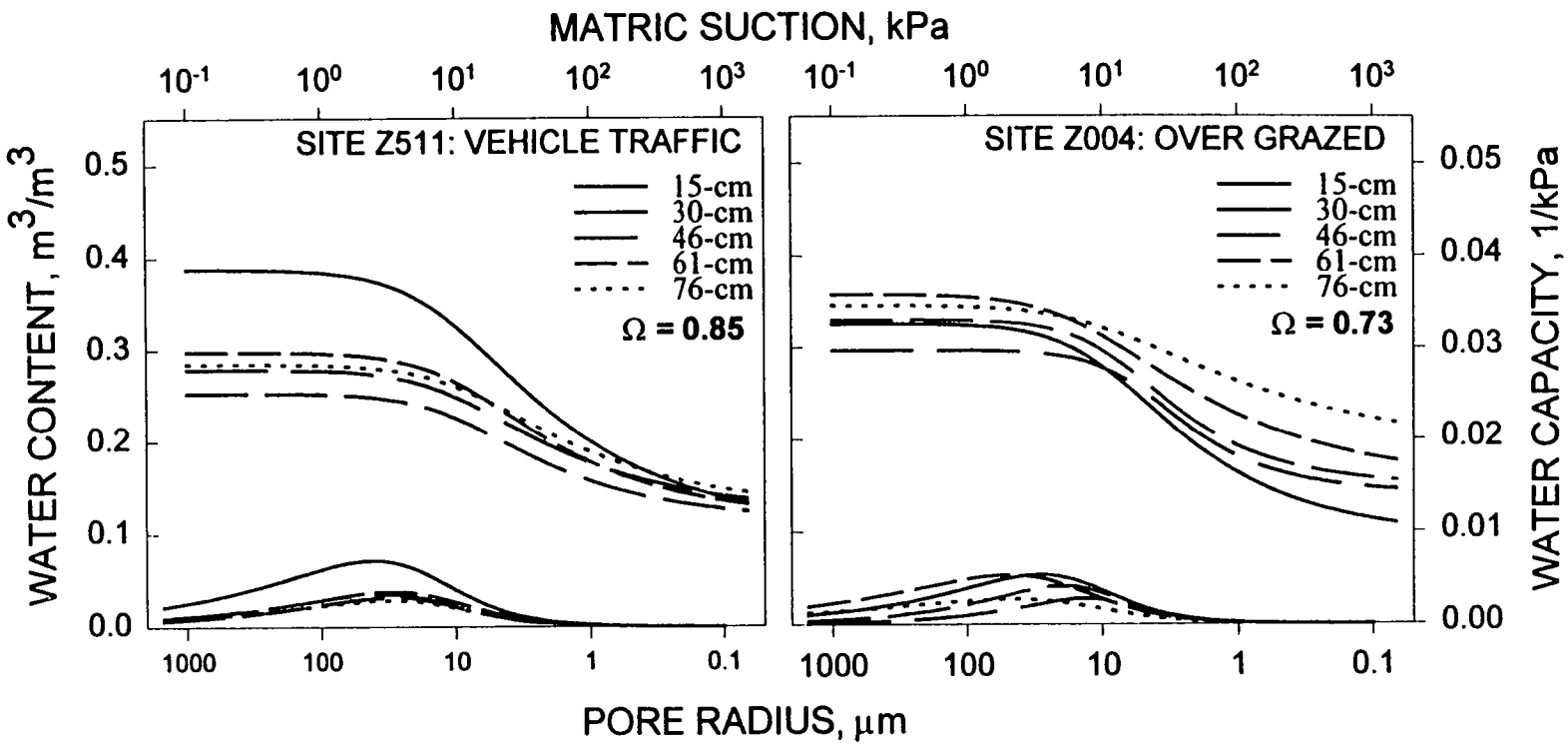

Fig. 2: Fitted soil water retention and water capacity curves for two premine compacted profiles.

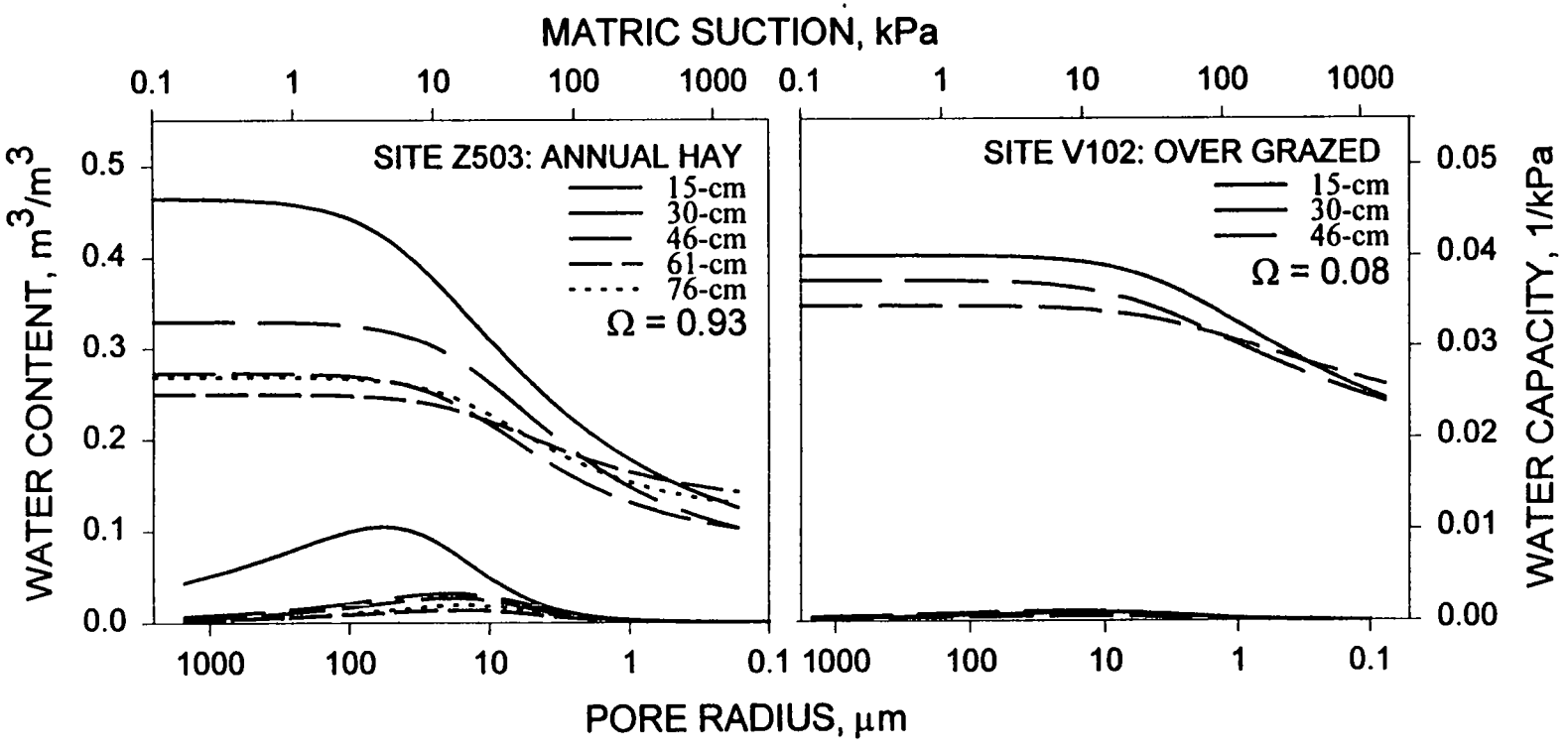

Fig. 3: Fitted soil water retention and specific water capacity curves for two reclaimed profiles.

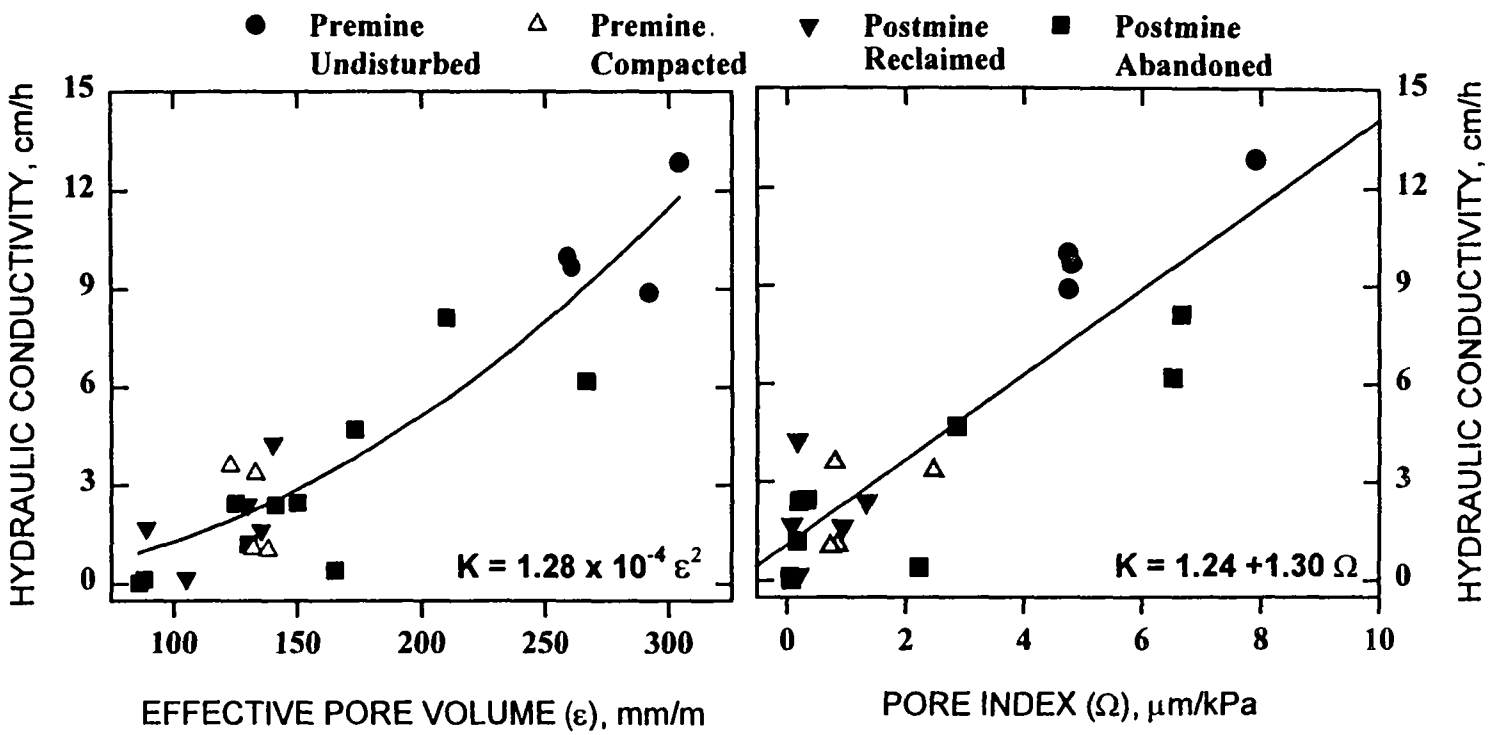

Fig. 5: Field saturated hydraulic conductivity(K) as a function of average volume of effective pores (ε) and weighted pore index (Ω) of premine and postmine soil profiles.

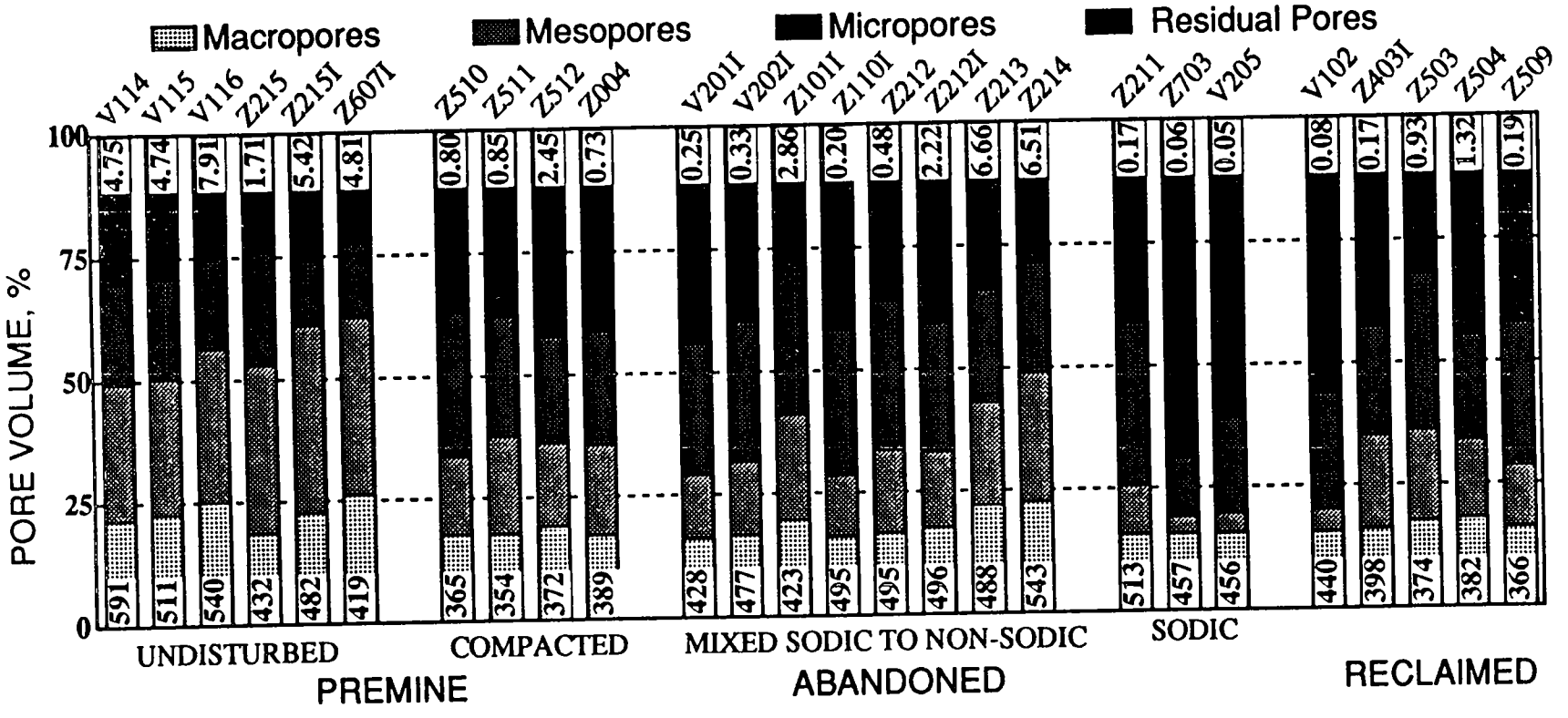

Fig. 4: Depth averaged pore size distribution of 26 pre- and post-mine soil profiles. The numbers at the bottom of each bar represent average total pore volume (mm/m), and those at the top represent the weighted pore index (Ω) of each profile.

AVAILABILITY OF P AND K IN RESTORED PRIME FARMLAND[1]

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Additional Key Words: P fixation; soil mixing; Reclamation.

Abstract. Modifications in soil nutrient availability of mine soils were studied in greenhouse and field experiments to investigate the effect of topsoil and subsoil mixing on available P and to study the response of corn to P and K fertilization. Topsoil and subsoil from an unmined Sadler silt loam (fine-silty, mixed, mesic, Glossic Fragiudalfs) were mixed in different proportions, treated with P and subjected to wetting and drying cycles in a greenhouse. Available P decreased due to wetting and drying, and more so, soil mixing, particularly with a more acidic subsoil. Two-year field studies with corn were conducted at two sites, at site 1 the soil was constructed predominantly from a Belknap silt loam (coarse-silty, mixed, acid, thermic Typic Fluvaquents) using end-dump trucks and at site 2, from a predominantly Sadler silt loam using scraper pans. Phosphorus and K were applied in a randomized complete block factorial design with four replications. In the first year, corn response to K was observed in earleaf tissue but not grain yield, at site 1. No response was observed in the second year. This was attributed to unfavorable moisture distribution. Site 2 data showed no response to P or K in both years. Compaction and moisture stress were thought as possible causes. A progressive build-up in soil test P and K was observed at both sites. About 18 kg P ha^{-1} were required for a 1 mg kg^{-1} increment in Bray I extractable P on a disturbed compared to 12 kg P ha^{-1} on an undisturbed Belknap soil. For K, 15 kg K ha^{-1} were required for a disturbed soil compared to 8.7 for an undisturbed Belknap soil. These results suggest that higher P and K fixation due to soil mixing may contribute to higher fertilizer requirements on disturbed soils compared to undisturbed ones.

Introduction

Current regulations require that prime farmland disturbed by surface mining be restored to a productivity level equal to or greater than its premined state. Frequently, efforts in meeting this condition are limited by poor soil physical conditions, low fertility, and moisture limitations, which characterize surface mined soils.

Previous studies have indicated that soil compaction (Indorante et al., 1981; Grandt, 1988), and its associated effect on root development, contribute to the observed weather sensitivity of row crops. Acidity (Dancer and Jansen,

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1987), low organic matter (OM) levels (Powell et al., 1986), microbial population, diversity, and activity (Stroo and Jencks, 1982; Visser et al., 1983) have been associated with deficiencies of essential nutrients, such as N and P, and consequently low productivity of surface mined soils.

The natural decrease in OM content with increasing soil depth and the relative difficulty of completely segregating the A horizon from underlying subsoil strata during soil removal, stock-piling, replacement, and leveling operations, results in a significant reduction in OM of the replaced topsoil (Powell et al., 1986). Beckwith (1965) and Sanchez and Uehara (1980) observed that some subsoil materials possess a higher P fixation capacity than topsoil. Field observations (Snarski et al., 1981; Barnhisel, 1988) indicate that topsoil and subsoil blending degrades topsoil physical and chemical properties. Better crop response was observed with topsoil and subsoil materials placed separately as opposed to mixing them (Jansen et al., 1985).

Although progress has been made towards establishing lime and fertilizer recommendations for forage on post mine soils (Barnhisel, 1976), similar work remains for row crops. According to Dancer and Jansen (1987), higher Bray I extractable P levels were required to attain the same level of corn production on disturbed soil sites than on undisturbed ones. In some instances, poor crop response to added P has been reported, yet low extractable P levels were observed at the end of the growing season (Dancer, 1984). Surface mining activities seem to modify soil characteristics significantly, leading to a higher P fixation and correspondingly, a higher fertilizer requirement.

The objectives of the present study were: (1) to investigate the effect of topsoil and subsoil mixing on phosphorus availability, (2) to determine the nutritional status and yield of corn following P and K treatment on restored prime farmland, and (3) to monitor changes and transformations of soil P and K following the treatments.

Materials and Methods

Greenhouse study

To study the effect of topsoil and subsoil mixing on P availability, topsoil (TS, pH_w 6.9) and subsoil (SS1, pH_w 7.1) samples of a Sadler silt loam (fine-silty, mixed, mesic, Glossic Fragiudalfs) were collected from a non mined site in Ohio County, Kentucky. Mixtures of topsoil and subsoil were prepared to make up 0, 25, 50, 75, and 100% subsoil (%SS). Three 1-kg portions of each mixture were weighed out, and mixed thoroughly with 50, 100, and 150 mg P as finely ground triple super phosphate [$Ca(H_2PO_4)_2.H_2O$]. In plastic pots, the samples were watered to field capacity (0.3 kg kg^{-1}) with deionized water and allowed to dry to wilting point (0.1 kg kg^{-1}) under greenhouse conditions. At the end of each two wetting and drying cycles, 100 g soil sub samples were collected from each mixture. The experiment was repeated using the same topsoil but with a subsoil (SS2, pH_w 4.5), predominantly a Sadler, collected from a stock-pile of a surface mine. Soil analyses for exchangeable bases based on the neutral 1M NH_4OAc (Thomas, 1982), available P by the Bray I (Bray and Kurtz, 1945) and Mehlich III (Mehlich, 1984) and P fractionation (Chang and Jackson, 1957) were performed on the sub samples collected.

Field Experiments

Field studies were conducted at two post mined sites in Hopkins and Muhlenberg Counties, western Kentucky. Soil at site 1 was reconstructed predominantly from a Belknap silt-loam (coarse-silty, mixed, mesic, Glossic Fragiudalfs) using end-dump trucks during the fall of 1987. This field was in fertilized alfalfa (Medicago sativa L.) in 1988 and 1989. Soil test data for spring 1990 indicated low P and K levels (Table 1). Phosphorus (0, 32, and 64 kg P ha^{-1}) and K (0, 75, and 150 kg K ha^{-1}) were broadcast (as in 1988), applied as factorial in a randomized complete block design with four replications. The field was planted to corn (Zea Mays L.) in June 1990 and 1991. Nitrogen (160 kg N ha^{-1}) was applied at planting and top-dressed at 80 kg N ha^{-1}.

Soil at site 2 was replaced from a Sadler silt loam using scraper pans during the spring 1988. This field was in unfertilized soybeans (Glycine max L.) in 1989. Soil test data for spring 1990 indicated very low P and low K levels (Table 1). Phosphorus (0, 37, 74, and 111 kg P ha^{-1}) and K (0, 139, and 278 kg K ha^{-1}) were applied as factorial in a randomized complete block design with four replications. The field was planted to corn in June 1990 and 1991. Nitrogen was applied at the same rates as at site 1.

Table 1. Some characteristics of the soils used in this study#.

Soil	pH_w	OM	CEC	Ca	Mg	K	Na	BP
			Greenhouse experiment					
		%	cmol(+) kg^{-1}					mg kg^{-1}
TS	6.9	3.07	15.2	7.96	0.92	0.37	0.06	9.3
SS1	7.1	1.37	10.9	7.50	1.28	0.44	0.05	5.2
SS2	4.5	0.89	10.0	2.00	1.90	0.47	0.08	3.2

Site	Text	pH_w	CEC	Ca	Mg	K	Na	BS	OM	BP	NK
			Field experiments								
			cmol(+) kg^{-1}					%		mg kg^{-1}	
1	sil	7.0	12.9	6.5	1.2	0.2	0.1	61.8	1.6	4.9	79.9
2	sil	6.9	11.2	5.6	1.8	0.2	0.1	69.3	1.6	0.8	97.3

#pH_w, OM, CEC, Ca, Mg, K, Na, and BS refer to 1:1 soil to water pH, organic matter content, cation exchange capacity, NH_4OAc extractable Ca, Mg, K, and Na, and base saturation, respectively. BP and NK are Bray I and NH_4OAc extr. P and K, respectively. Text is soil texture based on the pipette method.

Soils were analyzed for available P according to the Bray I and Mehlich III both at planting before fertilizer application and at harvest. Phosphorus fractionation was performed according to the Chang and Jackson Procedure (1957). Ear-leaf tissue sampled at silking were analyzed for five nutrients. Nitrogen and P were determined by the micro-Kjeldahl procedure as presented by Jones and Case (1990). Potassium,

Ca, and Mg were determined by the nitric-perchloric wet-ashing procedure (Jones and Case, 1990).

Results and Discussion

Greenhouse study

Topsoil and SS1 had similar 1:1 soil to water pH (pH_w) values (Table 1), with TS having higher exchangeable Ca and Bray I extractable P, but lower K and Mg than SS1. Topsoil had higher OM and CEC than SS1 while subsoil 2 (SS2) had a lower pH_w, OM, exchangeable Ca, CEC, and Bray I extractable P, but higher Mg, K, and Na than TS and SS1.

The effect of subsoil mixing on topsoil available P is presented in Fig. 1. There was a significant decrease in Bray I extractable P with increasing proportion of subsoil added to topsoil. The effect was greater in soil mixtures where a more acidic subsoil medium, SS2, was introduced. Figure 1 shows that adding 25% of SS2 to TS caused almost a 50% decrease in Bray I extractable P. Similar results were obtained with Mehlich III extractable P.

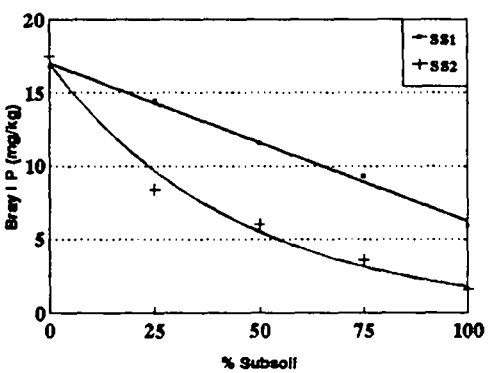

Fig. 1 Bray I P as affected by subsoil addition.

The decrease in available P (calculated as applied P minus extractable P) at a given time during the experiment was used as an estimate of the amount of P sorbed (Velayuthan, 1980). Table 2 shows that P sorption increased significantly as the proportion of subsoil in the mixture increased, particularly with SS2. These results relate closely with those of Beckwith (1965) that some subsoil materials exhibit a higher P fixation capacity.

Table 2. Sorbed P as affected by topsoil and subsoil mixing##.

%SS	Sorbed P SS1)		Sorbed P (SS2)	
	Bray	Mehl.	Bray I	Mehl.
	----------mg kg^{-1} ---------			
0	83.2	78.2	81.9	77.0
25	85.5	81.0	91.0	87.8
50	88.5	85.1	93.8	87.5
75	90.7	86.1	96.1	112.0
100	94.1	89.1	98.1	97.8
$LSD_{.05}$	2.0	3.0	2.0	2.3

Averaged across three P rates and eight cycles.

Field experiments

Soil characteristics at the beginning of field experiments are presented in Table 1. It is worth noting that soil at site 2 had much lower available P, compared to that at site 1, since unlike site 1, no previous P fertilization had taken place.

Corn response to P and K

Results for 1990 are summarized in Table 3. Phosphorus fertilization increased ear-leaf P and Ca significantly, but decreased K (possibly dilution). On the other hand, K application increased ear-leaf K, but decreased Ca and Mg contents. Many studies have reported

Table 3. Ear-leaf composition, yield, and soil test P and K as affected by P and K application, Site 1, 1990.

P rate	Ear-leaf N	Ear-leaf P	Ear-leaf K	Ear-leaf Ca	Ear-leaf Mg	Grain Yield	Bray I P
kg ha^{-1}	g kg^{-1}					Mg ha^{-1}	mg kg^{-1}
0	31.2	2.63	17.5	4.46	2.16	8.15	5.3
32	30.7	2.82	15.8	5.12	2.46	7.89	7.9
64	30.1	2.92	15.0	5.02	2.50	7.68	12.5
$LSD_{.05}$	1.2	0.14	1.2	0.51	0.41	1.37	2.9
K rate	N	P	K	Ca	Mg	Yield	NH_4OAc K
0	31.0	2.84	12.9	5.58	3.27	6.91	71.1
75	30.4	2.82	17.4	4.76	2.06	8.17	79.0
150	30.5	2.72	18.1	4.27	1.77	8.65	87.1
$LSD_{.05}$	1.2	0.14	1.2	0.51	0.41	1.37	9.7

that increased K uptake following K fertilization often decreases Ca and Mg uptake (Stout and Baker, 1981; Dibb and Thompson, 1985) and have attributed this to competition between K, and Ca and Mg during root uptake.

Results for grain yield indicated a significant response to K, but none for P. In general, yields were higher than the Phase III target value of 6.7 Mg ha^{-1} on this soil (Fehr et al., 1977). Optimum earleaf K content of about 18 g kg^{-1} corresponding to near maximum relative yield (ratio of any yield value to the maximum yield observed) was obtained at a soil test (NH_4OAc-K) value of about 86 mg kg^{-1} (Fig. 2). This is higher than the value of 71 mg kg^{-1} observed by Thom (1985) for a similar soil prior to disturbance.

Results for site 2 showed a significant response to K in ear-leaf tissue, but none for P (refer to Semalulu, 1992 for more detailed data). The lack of response to P, in spite of the very low initial soil test P level, was attributed to moisture stress. In addition, severe

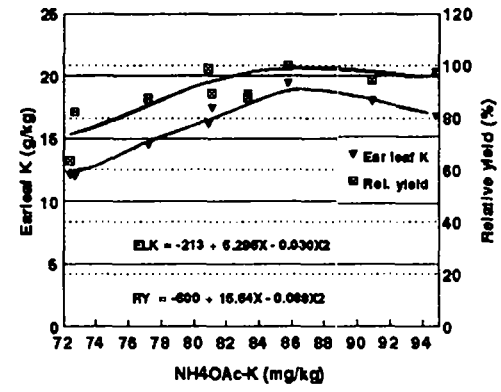

Fig. 2. Corn response to K, site 1.

compaction of soil at this site was observed during soil sampling (about 10 cm depth) the signs of which were evident during the growing season (prominent rip marks where the ripper had passed prior to seeding and water stagnation in some areas of the field during early spring). However, bulk density values were not determined.

Table 4. Changes in P fractions during the study

P rate	3/90			10/90			10/91		
	Al-P	Fe-P	Ca-P	Al-P	Fe-P	Ca-P	Al-P	Fe-P	Ca-P
kg ha^{-1}	mg kg^{-1}								
				Site 1					
0	21	79	19	14	181	33	21	157	31
32	24	57	18	21	184	37	31	178	40
64	29	59	24	32	193	41	43	203	45
$LSD_{.05}$	7	26	8	8	12	6	7	43	11
				Site 2					
0	9	82	20	5	90	27	8	81	25
37	8	90	17	7	101	25	18	111	31
74	10	83	16	12	105	25	42	129	39
111	9	90	21	15	114	33	43	152	52
$LSD_{.05}$	0.5	6	7	3	6	10	12	18	13

In 1991, climatic limitations severely affected crop performance at both sites (rainfall data presented in Semalulu, 1992). Heavy spring rains affected seed germination, in addition to possibly contributing to N loss. Also, the crop experienced water shortage at the later periods of growth, particularly during grain filling. Our results indicate a response to K at site 1 in the ear-leaf tissue, but not to P. Grain yields were lower than those in 1990, and not related to treatments. Similar trends were observed at site 2. Yields were lower than those for site 1. Compacted soil at site 2, coupled with unfavorable rainfall distribution as outlined above, might have further affected crop performance at site 2.

Phosphorus changes and transformations

There was a progressive increase in Bray I and Mehlich III extractable P due to P application, both at the end of a growing season and between years, proportional to the P rates applied. Table 4 shows that this was largely reflected in the aluminum phosphate fraction.

Figure 3 presents the relationship between the change in Bray I extractable P as a function of the total amount of P applied. Significant ($P<.01$) linear relationships were observed. For site 1,

$$Y = 0.498 + 0.057P;\ R^2 = 0.963 \quad (1)$$

and for site 2,

$$Y = -0.468 + 0.061P;\ R^2 = 0.824 \quad (2)$$

where Y is the change in Bray I extractable P (mg kg^{-1}), and P is the total amount of P (kg ha^{-1}) applied. From equation 1 and 2, the amount of P required to raise Bray I extractable P by 1 mg kg^{-1} was 17 kg ha^{-1} for site 1, and 16 kg ha^{-1} for site 2.

Previous work by Thom (1985) on an undisturbed Belknap silt loam soil in Webster County, western Kentucky, gave a value of 12 kg ha^{-1}. In studies on the P behavior of an undisturbed Sadler silt loam soil in Caldwell County, western Kentucky, Gallo (1989) reported high P immobilization by this soil. Results

from this study suggest an even higher P-fixation capacity after disturbance.

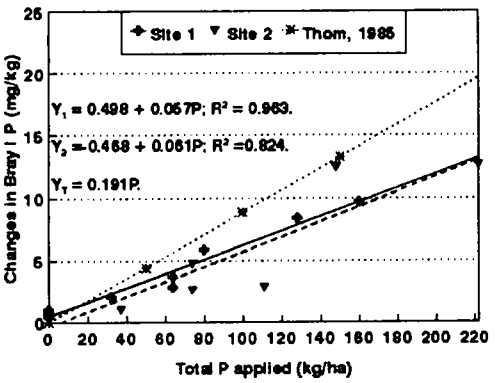

Fig. 3. Changes in Bray I P vs total P applied

Potassium changes and transformations

A progressive increase in NH_4OAc and Mehlich III extractable K was observed due to K fertilization. The change in NH_4OAc-K for site 1 was significantly related to total K applied ($P<.01$)

$$Y = 26.6 + 0.066K; \; R^2 = 0.738 \qquad (3)$$

where Y is the change in NH_4OAc extractable K, (mg kg^{-1}), and K is the total amount of K (kg ha^{-1}) applied.
From equation 3, 15 kg K ha^{-1} were required for to achieve a 1 mg kg^{-1} increment in NH_4OAc extractable K, which is higher than the value of 8.7 obtained by Thom (1985) for a similar soil prior to disturbance. However, a similar relationship for site 2 soil was not significant.

Conclusions

Soil compaction and moisture availability and distribution are major factors affecting corn growth on reclaimed soils. The optimum ear-leaf K concentration (18 g kg^{-1}) was obtained at higher NH_4OAc-K levels for a disturbed Belknap soil than for an undisturbed one (86 versus 71 mg kg^{-1}). Fertilization increased soil test P and K, and for P, this was most reflected in the aluminum phosphate fraction. Higher P and K rates were required to raise the soil test P and K levels of a disturbed Belknap soil by 1 mg kg^{-1} than for an undisturbed one (17 versus 12 kg P ha^{-1} and 15 versus 8.7 kg K ha^{-1}). Mixing of topsoil with subsoil during coal surface mining activities significantly reduces available P. The associated increase in P fixation of the resulting reconstructed soils probably accounts for the higher P fertilizer requirements as observed in the results of this study.

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SESSION 8: REVEGETATION ECOLOGY

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AMERICAN SOCIETY • SURFACE MINING and RECLAMATION •

THE POTENTIAL FOR USING WILDFLOWER SPECIES TO INCREASE NATURAL HABITAT IN CONTOUR SURFACE MINE RECLAMATION

by

John R. Heckman, Karen D. Holl, Mara Sabre, and John Cairns, Jr.

Abstract. While non-native herbaceous species are commonly used for mine reclamation, these species have low wildlife and aesthetic value and may inhibit long-term succession. The goal of this study was to determine the suitability of wildflowers for surface mine reclamation in the Appalachian mountains. A seed mixture composed of native and naturalized wildflower species was compared to the standard revegetation mixture by testing greenhouse germination rates of all species in both mine spoils and potting soil and establishing field plots at reclamation sites in southwestern Virginia. In May 1993, two 9-m^2 (97-ft^2) plots were seeded with each revegetation mixture on four slopes with different aspects. Vegetation cover and composition were recorded in all plots during the 1993 and 1994 field seasons. Wildflower species had germination rates ranging from 0-52%. In field studies, all but 2 of the 14 species of wildflowers seeded became established in study plots, while only 4 of the 8 species in the standard mixture were recorded. Cover was highly variable among plots on different aspects seeded with the same mixture. In most cases, total vegetative cover did not differ significantly between plots seeded with different mixtures. Some native and naturalized wildflower species appear to have potential for use in mine reclamation and could be included with standard revegetation mixtures in order to provide more native diversity. However, further research is necessary due to a number of factors confounding these results, including the low seeding rates used, drought conditions during the 1993 field season, and problems with regrowth of previous vegetation.

Additional Key Words: biodiversity, restoration.

Introduction

Large areas of land in the Appalachian region of United States have been disturbed by coal surface mining. According to the Office of Surface Mining, in Virginia alone nearly 50,000 ha (123,000 acres) have been disturbed by mining since passage of the Surface Mining Control

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and Reclamation Act (1977). The majority of these areas was revegetated using a few non-native grasses and legumes, including red top, Kentucky-31 tall fescue, orchard grass, clover, sweet clover, and birdsfoot trefoil. While these species provide rapid cover for erosion control and are inexpensive to seed, they provide little wildlife value and may inhibit long-term succession (Brenner et al. 1984; Burger and Torbert 1990; Hughes 1992).

In recent years, interest has increased in using wildflowers to revegetate disturbed areas such as landfills and roadsides. Wildflowers not only improve the aesthetic quality of an area, but they provide nectar and forage resources for wildlife and allow for the establishment of later successional plant species (Luken 1990). The goal of this project was to test the potential for using a number of native and naturalized wildflower and grass species for mine reclamation.

Site Description

This research was performed in Wise County, Virginia. The sites are located on the Powell River Project research area (PRP), an approximately 700-ha (1700-acre) area, about 125 ha (310 acre) of which have been mined. The PRP is a cooperative research program administered by Virginia Tech. The pre-mining soil in this area is comprised of acidic and infertile sandstones, siltsones, and shales (Daniels and Amos 1985).

The areas where test plots were located had been seeded prior to initiation of the experiment. North and south facing aspects were seeded in spring 1992; cover was predominantly birdsfoot trefoil, Kentucky 31-tall fescue, and perennial rye (*Lolium perenne*) at the initiation of the study. East and west facing slopes were seeded a few months before the start of the experiment and annual rye was just beginning to emerge.

Methods

Seed Selection and Preparation

The standard seed mixture consisted of species commonly used for mine reclamation in the region (Table 1). A list of native and naturalized wildflowers and grasses that would be tolerant of open canopy and drought conditions was compiled based on information provided by local horticulturalists and naturalists and a number of regional floras (Wofford 1993; Harvill et al. 1992; Radford et al. 1968). The preliminary wildflower species list was reduced to species for which seeds could be obtained from wildflower seed companies in the United States (Table 1).

Germination Studies

In March 1993, germination tests were conducted on seeds from different sources to determine germination rates. One hundred seeds of each species from both the standard and wildflower species mixtures were sown in two soil types in 0.093-m^2 (1-ft^2) flats in greenhouses. Soil types included a control potting soil (a 3:1:1 mixture of peat, perlite, and vermiculite), and soil collected from the top 5 cm of areas adjacent to the experiment plots. Seeds in each flat were watered daily, and the number of germinating seeds was counted every 3 to 4 days for approximately 30 days.

Site Preparation

In April 1993, two 10 x 5-m (33' x 15') plots were marked on each of four aspects, north, south, east, and west, on shallow slopes of reclaimed mine sites at the PRP. Two 9-m^2 sub-plots were marked in each of the plots with a space of 0.5 m (1.6-ft) separating the sub-plots. Permanent sub-plots of 1-m^2 (11-ft^2) were marked at the center of each 9-m^2 sub-plot for cover surveys. In

total, each mixture was seeded in eight sub-plots at each site.

The standard revegetation mixture from hydroseeding conducted the previous year began to emerge in March. In April, plots were sprayed with a glyphosate herbicide (Roundup®, Monsanto) to eradicate preexisting vegetation. Roundup® is a topically applied, systemic herbicide that kills nonwoody plants; however, it does not affect seed viability. A 0.5% solution of concentrate was mixed with water on site, and plots were sprayed using a backpack sprayer. Each block was sprayed twice, with an interval of 12 days between sprayings. All vegetation soon displayed symptoms of chlorosis and died within 14 days of spraying.

In May, each 9-m^2 sub-plot was seeded with either the standard or wildflower revegetation mixture. Seed mixtures were preweighed for each sub-plot, with seeding densities for the wildflower mixture calculated according to standard specifications from national wildflower seed companies. Most companies recommended sowing 100-120 seeds per 0.093 m^2 (1 ft^2) to provide maximum coverage of disturbed areas. Pure live seeds estimates were not used because they were not available from all companies. Grasses were seeded at a density of 270 gm/ha (0.25 lb/acre) for each species. One species, showy evening primrose, was only seeded on north and south facing aspects because of a shortage of seed. Seed amounts in the standard mixture were determined differently because the donated seeds had arrived premixed. An estimation of the number of seeds was made based on the number of seeds commonly found per gram of each species and then multiplied by the number of seeds needed to complement the number of seeds being seeded in the wildflower mixture. A total of approximately 11610 seeds were mixed with 500 g (1 lb) coarse construction sand and spread by hand broadcasting on each 9-m^2 sub-plot.

Table 1. Percent germination of 100 seeds of each species in the standard reclamation and wildflower mixture in potting soil and mine soil.

Scientific name[1]	% Germination PS[2]	MS
STANDARD MIX		
Annual rye	87	93
Birdsfoot trefoil	65	62
Clover	64	71
Foxtail millet	62	50
Kentucky-31 tall fescue	69	63
Orchard grass	85	67
Red top	64	71
Sweet clover	69	57
WILDFLOWER MIX		
Annual sunflower	4	4
Bergemont	49	0
Big bluestem	29	37
Black-eyed susan	57	28
Catchfly	78	52
Cornflower	70	28
Dame's rocket	60	0
Lance-leaved coreopsis	51	48
Little bluestem	14	33
New England Aster	55	14
Perennial lupine	37	17
Plains coreopsis	71	38
Showy evening primrose	53	11
Stiff goldenrod	57	0

[1]Common names and addresses of seed suppliers listed in Appendix 1
[2]PS = potting soil; MS = mine soil

No fertilizers nor irrigation was applied to plots.

Vegetation Measurements and Analyses

The plots were monitored biweekly between late May and early September in 1993 and 1994. On each sampling date, percent cover and species richness of planted species were recorded in the 1 m^2 at the center of sub-plot. Percent cover and composition of naturally invading species were also recorded, but these data are not presented here.

Results of species richness and cover measurements were combined for north and south aspects, as well as east and west aspects, as results were similar. Species richness values include observations from the entire field season, while cover measurements from mid-August are reported.

Soil Analyses

In May 1993 and August 1994, composite soil samples were taken from each experimental plot. All analyses were done by the Virginia Tech Soil Testing Laboratory.

Results

Germination rates for species in the standard mixture were all ≥50% and were similar in both potting and mine soils (Table 1). Germination rates for species in the wildflower mixture were mostly ≥50% in potting soil but ≤50% in mine soils. Germination rates for species in the wildflower mixture were lower in mine soils for all species except the two grasses, little bluestem and big bluestem. One species, annual sunflower, had a 4% germination rate in both potting and landfill soil.

In the field experiments, species richness was significantly higher in wildflower plots in all cases except east and west plots in 1993 (Table 2). However, this would be expected as more wildflower species were planted. The percent of seeded species that germinated was also higher in wildflower plots. This difference was statistically significant on north and south facing plots in 1993 (Table 2).

Only four of eight species in the standard reclamation mixture became established, and one species, foxtail millet, was only observed in two plots in year 1 (Table 3). Dominant species in standard mixture plots in both 1993 and 1994 were birdsfoot trefoil and clover. Twelve of fourteen species in the wildflower mixture became established (Table 3). Only stiff goldenrod and little bluestem were not recorded in experimental plots. Lance-leaved coreopsis, plains coreopsis, and black-eyed Susan were widespread in both years. Two annuals, cornflower, and catchfly, were common in year 1, but were rarely observed in the year 2. Big bluestem, New England aster, Dame's Rocket and bergemont were not observed in year 1, but became established in year 2.

Total cover of planted species for both seed mixtures was low in both years, but increased from 1993 to 1994 (Table 4). Cover was much higher in north and south plots than in east and west plots, which may be due to reoccurrence of previously established vegetation on north and south plots. In 1994, cover levels were significantly higher in north and south plots seeded with the standard reclamation mixture. However, these plots were dominated by birdsfoot trefoil, which partially regrew from plants seeded before the study.

Soil pH, nitrate, and organic matter levels were much lower on east and west facing slopes at the time of seeding, while phosphorus, potassium, and calcium levels were similar (Table 5). None of these variables changed greatly within the 15-month study period. Seed mixture did not have a significant effect on any of the variables measured.

Discussion

Results of field studies suggest that a number of native and naturalized wildflower species have potential for revegetating mined areas, despite the many obstacles to establishing vegetation on these sites, such as low pH, soil compaction, elevated surface temperatures, water stress, and low nutrients. All but two of the wildflower and native grass species became established in experimental plots. Cover levels were similar in areas seeded with the standard reclamation and wildflower mixtures.

Table 2. Mean species richness and percent of seeded species that germinated for standard and wildflower mixtures in 1993 and 1994.

Year	Aspect	Species richness[1] Standard	Wildflower	p[2]	% of species germinating Standard	Wildflower	p
1993	North/south	2.3 ± 0.5	6.0 ± 0.8	***	28 ± 7	39 ± 4	*
	East/west	2.0 ± 0.0	4.0 ± 2.0	NS	25 ± 1	31 ± 16	NS
1994	North/south	2.3 ± 1.0	6.0 ± 1.4	**	29 ± 12	43 ± 10	NS
	East/west	2.3 ± 1.0	6.0 ± 1.4	**	31 ± 13	46 ± 10	NS

[1]Mean ± standard deviation
[2]NS = p>0.05, *p<0.05, **p<0.01, ***p<0.001

Table 3. Frequency and cover measurements for standard reclamation and wildflower species seeded on mine sites.

Scientific name	Number of plots[1] 1993	1994	Cover ranking[2] 1993	1994
STANDARD MIX				
Annual rye	-	-	-	-
Birdsfoot trefoil	8	8	3	4
Clover	8	4	3	1
Foxtail millet	2	1	-	-
Kentucky-31 tall fescue	-	-	-	-
Orchard grass	-	-	-	-
Red top	-[3]	-	-	-
Sweet clover	1	4	1	2
WILDFLOWER MIX				
Annual sunflower	6	-	1	-
Bergemont	-	5	-	2
Big bluestem	-	7	-	1
Black-eyed susan	8	8	1	2
Catchfly	3	-	1	-
Cornflower	6	1	2	1
Dame's rocket	6	2	1	1
Lance-leaved coreopsis	8	7	1	2
Little bluestem	-	-	-	-
New England Aster	-	4	-	2
Perennial lupine	3	2	1	1
Plains coreopsis	8	5	1	1
Showy evening primrose[4]	4	2	-	-
Stiff goldenrod	-	-	-	-

[1]Number of plots out of a total of eight in which each species was recorded
[2]Average cover of each species in mid-August using ranking system: 0-1% = 1, 1-5% = 2, 5-10% = 3, >10% = 4
[3]Represents no appearance of the species within the experimental plot
[4]Only seeded in four plots on north and south facing slopes

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Appendix 1. Species scientific names and seed suppliers.

Common Name	Scientific Name	Seed Provider
STANDARD MIX		
Annual rye	*Secale cereale*	1
Birdsfoot trefoil	*Lotus corniculatus*	1
Clover	*Trifolium* spp.	1
Foxtail millet	*Setaria italica*	1
Kentucky 31-tall fescue	*Festuca arundinacea*	1
Orchard grass	*Dactylis glomerata*	1
Red top	*Agrostis alba*	1
Sweetclover	*Melilotus* spp.	1
WILDFLOWER MIX		
Annual sunflower	*Helianthus annuus*	2
Bergemont	*Monarda fistulosa*	3
Big bluestem	*Andropogon gerardii*	2
Black-eyed susan	*Rudbeckia hirta*	4
Catchfly	*Silene armeria*	2
Cornflower	*Centaurea cyanus*	4
Dame's rocket	*Hesperis matronalis*	4
Lance-leaved coreopsis	*Coreopsis lanceolata*	4
Little bluestem	*Schizachrium scoparius*	3
New England aster	*Aster novae-angliae*	2
Perennial lupine	*Lupinis perennis*	4
Plains coreopsis	*C. tinctoria*	4
Showy evening primrose	*Oenothera speciosa*	5
Stiff goldenrod	*Solidago rigida*	5

[1]Country Boy
1811 Gate City Hwy.
Bristol, TN 24201

[2]Applewood Seed Co.
5380 Vivian St
Arvada, CO 80002

[3]Prairie Nursery
P. O. Box 306
Westfield, WI 53946

[4]Lofts
P. O. Box 146
Bound Brook, NY 08805

[5]S&S Seeds
P. O. Box 1275
Carpenteria, CA 93013

PREDICTING GRASS STAND ADEQUACY FROM EARLY STAND PARAMETERS[1]

by

R. E. Ries[2]

Abstract. Early evaluation of grass stand establishment is important for making timely decisions on reseeding or proper management that will enhance the new grass stand. A field study with 5 species, 11 seeding dates per year, and 3 replications was conducted to determine which dates resulted in the best stands when direct seeded into wheat stubble using the same seeding techniques at each date. Seedings were made during the seeding years of 1987, 1988, and 1989. Stands were monitored for 5 years, 2 years after each seeding year. The number of grass seedlings m^{-2} was measured 45 days after emergence. This early stand parameter was correlated with the number of grass plants m^{-2}, number of grass stems m^{-2}, and grams of grass dry matter m^{-2} measured 2 years after the seeding year. The highest correlation coefficients ($P<0.01$) were between the number of grass seedlings m^{-2} 45 days after emergence and the number of grass stems m^{-2} 2 years after seeding for smooth bromegrass, western wheatgrass, crested wheatgrass, and sideoats grama. Correlation coefficients were 0.69, 0.59, 0.71, and 0.41, respectively. The highest correlation coefficient (0.71, $P<0.01$) for blue grama was between the number of grass seedlings m^{-2} 45 days after emergence and the number of grass plants m^{-2} 2 years after seeding. Predictions of expected grass stems m^{-2} at 2 years after seeding and actual measured stems m^{-2} at 2 years were compared with a standard number of stems need for a successful stand. A decision on stand success made from the seedlings m^{-2} at 45 days after emergence was correct 100, 85, and 76% of the time for smooth bromegrass, western wheatgrass, and crested wheatgrass, respectively. For blue grama, prediction of stand success from the number of seedlings present 45 days after emergence was correct 79% of the time. Two year success estimates were not determined for sideoats grama because the relationship of seedlings m^{-2} and stems m^{-2} was only significant 1 out of 3 seeding years. Based on these results, a reasonable estimate of stand adequacy at 2 years after seeding may be predicted by the number of grass seedlings counted at 45 days after emergence.

Additional Key Words: seedlings m^{-2}, stems m^{-2}, plants m^{-2}, grams m^{-2}, regression equation, parameter correlations, smooth bromegrass, sideoats grama, western wheatgrass, blue grama, crested wheatgrass.

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Introduction

Monitoring changes occurring in vegetative communities has been recognized as the key to natural resource management (Bonham 1989). Changes occurring in grass stands are an indication that management changes may be needed to initiate and maintain productive stands. The relationship of early stand characteristics with productive and established stand characteristics is important to early determination of grass seeding success or failure. Early detection of seeding failures or marginal grass stands can result in timely decisions to reseed or apply proper management that will enhance the young grass stand. Vallentine (1979) indicated that monitoring new grass seedings should begin as soon as reliable data can be measured. Data evaluation indicating when monitoring of new grass stands should begin is scarce. Little information is available in the literature concerning the relationship of early grass stand parameters with the parameters that characterize an established and productive grass stand. Cook and Stubbendieck (1986) recommended that stand establishment measurements of perennial plants seeded in the semi-arid range should start near the end of the second growing season. According to their experience, only tentative data on new stand characteristics could be obtained during the first growing season. Ries and Svejcar (1991) reported data that suggest new stands of crested wheatgrass [*Agropyron desertorum* (Fisch. ex Link) Schult.] and blue grama [*Bouteloua gracilis* (H.B.K.) Lag. ex Steud.] could be considered established 21 days after emergence under the specific environmental conditions of their study. The purpose of this current paper is to characterize the relationships between grass stand parameters found during early and late stand development.

Study Area and Methods

This study was conducted at the USDA-ARS Northern Great Plains Research Laboratory, Mandan, North Dakota, to evaluate 11 seeding dates per year over 3 seeding years. The study was conducted on a Parshall fine sandy loam soil (coarse-loamy, mixed *Pachic Haploborolls*) on nearly level terrain. Monoculture grass seedings were made directly into spring wheat stubble with a double disk cone seeder with depth bands and packer wheels. Seedings were made August 15, September 4 and 24, October 9, November 1, April 1 and 21, May 9 and 26, June 10, and July 1 during the seeding years of 1987 (1986-87), 1988 (1987-88), and 1989 (1988-89). All species were seeded at 9 kg ha^{-1} pure live seeds at a depth of about 13 mm. Species studied included 3 cool-season grasses, smooth bromegrass [*Bromus inermis* Leyss., 'Lincoln'], western wheatgrass [*Agropyron smithii* Rydb., 'Rodan'; syn. = *Pascopyron smithii* (Rydb.) Löve], and crested wheatgrass, [*Agropyron desertorum* (Fisch. ex Link) Schult.) 'Nordan']. Two warm-season grasses were also evaluated; sideoats grama [*Bouteloua curtipendula* (Michx.) Torr., 'Pierre'], and blue grama [*Bouteloua gracilis* (H.B.K.) Lag. ex Steud., native collection].

Plots (3 m wide and 7.6 m long) were arranged in a randomized complete block design, split/split plot in time with 3 replications. Whole plots were years, with the first split species and the final split seeding date. Grass stands were sampled for seedling density (plants m^{-2}) 45 days after emergence was observed by counting the seedlings present on 5 randomly located 30.5 by 30.5 cm plots for each species and seeding date. The stands were sampled again during the growing season at near peak biomass accumulation (between July 1-15) 1 and 2 years after the seeding year. Measurements of grass plant density (plants m^{-2}), stem density (stems m^{-2}), and dry matter

(grams m^{-2}) were taken by counting the number of grass plants and grass stems, and then clipping and oven drying (60°C for 48 h) grass plants sampled from 5 randomly located 30.5 by 30.5 cm plots for each species and seeding date.

Environmental conditions during the study were monitored by a weather station located on the study area following National Weather Bureau standards. Weather conditions measured included air temperature at 2 m above the soil, soil temperature at 16 mm-soil depth, precipitation, and free water evaporation from a Class A pan.

The grass species studied did not respond in the same manner to study conditions; therefore, correlation/regression analyses were conducted by species. Each seeding year included 11 seeding dates and 3 replications for a total of 33 observation. Since weather conditions and seedling response at each seeding date within each seeding year were different, an average function was developed by including data from all 3 years for a total of 99 observations. The number of grass seedlings m^{-2} counted at 45 days after emergence was correlated with the number of grass plants m^{-2}, number of grass stems m^{-2}, and grams of grass dry matter m^{-2} measured in each stand at near peak biomass accumulation 1 and 2 years after the seeding year.

Regression equations for the parameters with the highest correlation coefficients that were significant for all 3 seeding years were determined. The number of grass stems m^{-2} expected 2 years after the seeding year were predicted for smooth bromegrass, western wheatgrass, and crested wheatgrass by the regression equation developed for the relationship of the number of grass stems m^{-2} at 2 years (Y) with the number of grass seedlings m^{-2} at 45 days after emergence (X). The number of grass plants m^{-2} expected 2 years after seeding for blue grama were predicted by the regression equation for the relationship of the number of grass plants m^{-2} at 2 years (Y) with the number of grass seedlings at 45 days after emergence (X). A standard of 11 plants m^{-2} (Great Plains Council 1966) at 45 days after emergence was accepted as a measure of a successful grass seeding. The same regression equations were used to predict the expected number of stems m^{-2} or number of plants m^{-2} (blue grama) after 2 years that would be expected from a successful stand with 11 plants m^{-2} at 45 days after emergence. The predicted and actual measured stems m^{-2} or plants m^{-2} during the growing season 2 years after seeding were compared with the expected stems m^{-2} or plants m^{-2} for a successful stand. When either the actual or predicted values was less than the successful stand density, an error in estimating stand success from 45 days after emergence data occurred. Thirty-three stands minus the number of errors equalled the number of successful stands predicted from 45 days after emergence data. The number of successful stands predicted divided by 33 equalled the percent success.

The relationships of all combinations of average grass plants m^{-2} with grass stems m^{-2} with grams of grass dry matter m^{-2} measured 2 years after seeding were also quantified by correlation analysis. Relationships of all combinations of average grass plants m^{-2} with grass stems m^{-2} with grams of grass dry matter m^{-2} measured at 1 year and 2 years after seeding were also established by correlation analysis.

Results and Discussion

Weather during this 5 year study was variable among months as is common for the semi-arid area of the Northern Great Plains. Air temperatures for August through October were lower than long-term averages for the study area, while air temperatures for April through July were above the long-term

average (Table 1). Mean soil temperatures at the 16 mm-soil depth were warmest during July with an average of 25.6 C (Table 1).

Soil temperatures for August and June were 23.2 and 23.7 C, respectively. During the months of August and May precipitation was highest at 60 and 61 mm, respectively. August precipitation was 10 mm higher than the long-term average, while the May precipitation was about equal to the long-term average. The greatest amount of free water evaporation during the study was observed during July and was only 6 mm more than the long-term average. April and June free water evaporation were also above the long-term average during this study (Table 2).

Table 1. Range, average (1986-91), and long-term average mean monthly air and soil temperatures (°C).

Month	Mean Air Temp. @ 2 m			Mean Soil Temp. @ 16 mm		
	Range	Avg.	LTA[1]	Range	Avg.	LTA[2]
Aug	18 to 22	19.7	20.4	20 to 27	23.2	--
Sep	11 to 16	13.9	14.0	12 to 18	16.1	--
Oct	5 to 7	6.5	7.8	7 to 9	7.6	--
Nov	-4 to 2	-1.1	-1.8	1 to 2	0.5	--
Apr	6 to 10	7.7	5.3	9 to 12	10.3	--
May	13 to 16	14.1	12.5	16 to 19	17.5	--
Jun	18 to 25	20.1	19.4	21 to 30	23.7	--
Jul	21 to 24	22.2	21.6	24 to 29	25.6	--

[1] LTA = long-term average--78 years (1914-1991).
[2] LTA not available.

Table 2. Range, average (1986-91), and long-term average monthly precipitation and free water evaporation (mm).

Month	Precipitation			Free Water Evaporation		
	Range	Avg.	LTA[1]	Range	Avg.	LTA[2]
Aug	38 to 99	60	50	115 to 231	182	203
Sep	1 to 108	38	40	80 to 154	126	130
Oct	0 to 6	4	21	--	--	--
Nov	0 to 71	18	11	--	--	--
Apr	0 to 58	22	39	112 to 174	139	106
May	20 to 88	61	57	131 to 185	158	173
Jun	4 to 112	52	89	186 to 282	213	195
Jul	21 to 95	44	57	164 to 287	238	232

[1] LTA = long-term average--78 years (1914-1991).
[2] LTA = long-term average--28 years (1964-1991).

The largest significant correlation coefficients ($P<0.01$) were for the number of seedlings m^{-2} 45 days after emergence and the number of stems m^{-2} 2 years after the seeding year for smooth bromegrass, western wheatgrass, crested wheatgrass, and sideoats grama. Coefficients were 0.69, 0.59, 0.71, and 0.41, respectively (Bold numbers in Table 3). Number of seedlings m^{-2} at 45 days after emergence was most highly correlated with the number of plants m^{-2} 2 years after seeding for blue grama ($r = 0.71$, $P<0.01$, Table 3). Correlation coefficients for these relationships for individual study years are shown in Table 4.

Table 3. Correlation coefficients (r) for seedlings m^{-2} at 45 days after emergence with grass stand parameters at 2 years after seeding for all 3 seeding years (n=99).

	Parameters		
Species	plants m^{-2}	stems m^{-2}	grams m^{-2}
Smooth bromegrass	0.54**	**0.69****	0.56**
Western wheatgrass	0.52**	**0.59****	0.47**
Crested wheatgrass	0.69**	**0.71****	0.41**
Sideoats grama	0.39**	**0.41****	0.20*
Blue grama	**0.71****	0.64**	0.36**

**P<0.01 -- Bold type indicates largest r value.
*P<0.05

Table 4. Correlation coefficients (r) for seedlings m^{-2} at 45 days after emergence with stems m^{-2} or plants m^{-2} (Bogr) at 2 years after seeding.

Species	1987	1988	1989	1987-89 Avg.
Smooth bromegrass	0.63**	0.69**	0.44*	0.69**
Western wheatgrass	0.75**	0.60**	0.40*	0.59**
Crested wheatgrass	0.75**	0.78**	0.59**	0.71**
Sideoats grama	0.30^{ns}	0.15^{ns}	0.54**	0.41**
Blue grama	0.62**	0.80**	0.68**	0.71**
# of observations	n=33	n=33	n=33	n=99

**P<0.01
*P<0.05

Most of the correlation coefficients were significant during each year except for sideoats grama where coefficients were nonsignificant during the 1987 and 1988 seeding years. Because of the insignificance of this relationship during some years, the regression equation was not used to predict stand parameters 2 years after seeding (Table 5). The equations used to predict future stand parameters for smooth bromegrass, western wheatgrass, crested wheatgrass, and blue grama are presented in Table 5.

Table 5. Regression equations quantifying the relationship of stems m^{-2} or plants m^{-2} at 2 years after seeding with seedlings m^{-2} present at 45 days after emergence (n=99)[1,2,3].

smooth bromegrass	
stems m^{-2} @ 2 yrs. = 270.43 + 4.11 (seedlings m^{-2} @ 45 days)	r^2 = 0.48**
western wheatgrass	
stems m^{-2} @ 2 yrs. = 374.49 + 7.27 (seedlings m^{-2} @ 45 days)	r^2 = 0.35**
crested wheatgrass	
stems m^{-2} @ 2 yrs. = 544.53 + 6.80 (seedlings m^{-2} @ 45 days)	r^2 = 0.50**
blue grama	
plants m^{-2} @ 2 yrs. = 5.10 + 0.18 (seedlings m^{-2} @ 45 days)	r^2 = 0.50**

[1] P>T = 0.0001 for intercept.
[2] P>T = 0.0001 for regression coefficient.
[3] P>F = 0.0001 for overall equation.
**P<0.01

These equations were used to predict the expected number of stems m^{-2} or plants m^{-2} (blue grama) 2 years after seeding from seedlings m^{-2} data measured 45 days after emergence. Correct predictions of stand success based on the number of stems m^{-2} were made 100, 85, and 76% of the time using the regression equations for smooth bromegrass, western wheatgrass, and crested wheatgrass, respectively. Correct predictions of stand success were made 79% of the time based on the number of plants m^{-2} expected for blue grama. Based on these results, a reasonable estimate of stand adequacy at 2 years after seeding may be predicted by the number of seedlings present 45 days after emergence.

There was also a highly significant relationship between the number of stems m^{-2} and grams dry matter m^{-2} 2 years after seeding (Table 6). For all species, a measure of stems m^{-2} strongly reflected grams dry matter m^{-2}. This suggests that a good estimate of grams of grass dry matter m^{-2} for a 2 yr old stand could be obtained from the counted number of stems m^{-2} rather than the clipping, oven drying, and weighing usually required when stand production is of interest.

Table 6. Correlation coefficients (r) for stems m^{-2} with grams dry matter m^{-2} at 2 years after seeding (n=99).

Species	grams m^{-2}
smooth bromegrass	0.74**
western wheatgrass	0.92**
crested wheatgrass	0.78**
sideoats grama	0.74**
blue grama	0.78**

**P<0.01

The parameters of plants m^{-2}, stems m^{-2}, and grams dry matter m^{-2} measured 1 year after seeding were significantly correlated with the same parameter measured 2 years after seeding. The highest correlation coefficients were 0.77 and 0.73 for the number of stems m^{-2} (year 1 with year 2) for smooth bromegrass and western wheatgrass, respectively (Bold numbers in Table 7).

Table 7. Correlation coefficients (r) between grass stand parameters measured 1 year and 2 years after seeding (n=99).

	Species				
	Smooth bromegrass	Western wheatgrass	Crested wheatgrass	Sideoats grama	Blue grama
	plants m^{-2}				
Year 1	37	32	28	15	13
Year 2	25	49	29	13	11
r	0.59**	0.42**	**0.89****	**0.88****	**0.88****
	stems m^{-2}				
Year 1	268	277	488	286	343
Year 2	476	665	1040	356	610
r	**0.77****	**0.73****	0.80**	0.56**	0.87**
	grams m^{-2}				
Year 1	130	56	122	43	17
Year 2	299	204	351	118	74
r	0.63**	0.71**	0.47**	0.70**	0.78**

**P<0.01 -- Bold type indicates largest r value.

Correlation coefficients of 0.89, 0.88, and 0.88 were obtained for the number of plants m^{-2} (year 1 with year 2) for crested wheatgrass, sideoats grama, and blue grama, respectively (Bold numbers in Table 7). The number of stems m^{-2} was better related between year 1 and year 2 for the two rhizomatous species, while the number of plants m^{-2} was better related between year 1 and year 2 for the bunch-grass type species. This occurs because of the difficulty in identifying individual plants for rhizomatous species while counting the number of stems of rhizomatous species is readily repeatable. Similar information was reported by Cook and Stubbendieck (1986) and Bonham (1989). The significant correlations between year 1 and year 2 stand parameters support the concept that valid evaluation of grass stand success can be made 1 year after seeding when this period includes a winter dormant period (McWilliams 1955 and Cook and Stubbendieck 1986).

Summary

A good evaluation of stand parameters best used in evaluating grass stand success have been found from this study. Seedlings m^{-2} present 45 days after emergence were a reliable measure that correlated well with stems m^{-2} 2 years after seeding for smooth bromegrass, western wheatgrass, crested wheatgrass, and sideoats grama. Seedlings m^{-2} 45 days after emergence were better correlated with plants m^{-2}

than stems m^{-2} after 2 years for blue grama. A significant relationship between stems m^{-2} and grams dry matter m^{-2} 2 years after seeding was found. Data also showed that stand parameters 1 year after seeding were highly correlated with the same parameters 2 years after seeding, further suggesting that stand success can be evaluated 1 year after seeding when this year includes a dormant period. Stems m^{-2} 1 year and 2 years after seeding were more highly correlated for rhizomatous species; while, plants m^{-2} 1 year and 2 years after seeding were more highly correlated for bunchgrass species.

Results from this study are encouraging that good estimates of stand success may be made as early as 45 days after emergence. Estimates made in our study were quite successful and can be used along with good observations and judgment when evaluating stands as early as 45 days after grass seedling emergence. The r^2 for our study range from 0.35 to 0.50 for the relationship of seedlings m^{-2} 45 days after emergence with stems or plants m^{-2} 2 years after seeding. Except for sideoats grama, the relationship was significant in each seeding year. The goal now is to improve these r^2 numbers by adding secondary stand parameters that may further define the establishment level of the grass seedlings at 45 days after emergence. Addition of specific weather conditions may also increase these initial r^2 numbers. Both secondary stand parameters or weather conditions may improve the relationship of seedlings m^{-2} 45 days after emergence with the success of the stands 1 year or 2 years after seeding.

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VEGETATION DIVERSITY WITHIN NATIVE AND RECLAIMED COAL MINE SITES: ENVIRONMENTAL FACTORS AND SEASONAL VARIABILITY[1]

by

Brenda K. Schladweiler and George F. Vance[2]

Abstract Within arid and semiarid environments of western North America, vegetation composition, i.e., quantity and quality of plant species, can be highly variable on native lands and reclaimed coal mine sites. Therefore, we examined the vegetation composition within 23 native and 79 reclaimed plots at two active coal mines in the Powder River Basin, Wyoming. Cool season, warm season and annual grasses, annual and perenial forbs, and half and full shrubs were inventoried and sampled once in 1991 and twice during 1992 and 1993. Seasonal variation was determined based on site data and plant sampling during early (May) and late (July and August) growth stages. General conclusions drawn from vegetation cover results, i.e., species composition, diversity, and trends, suggest the following relations. Based on age of reclaimed areas, species composition changed from early successional plants to longer-lived individuals. Diversity of the seeded areas reflected actual seed mixes with some invasion based on reclamation method such as direct haul topsoil versus stockpiled topsoil and post-management techniques (e.g. livestock grazing). On native areas, lifeform category cover percentages, e.g., annual grasses and forbs, varied with climate factors such as precipitation and temperature. Such factors had a strong influence on the presence of certain plant species. Due to the extreme variability in weather conditions over the three-year project, we can only conclude that much of the vegetation variability was due to weather conditions.

Additional Key Words: Species Composition, Diversity, Trends, Lifeforms, Grasses, Forbs, Shrubs, Weather, Native, Reclaimed

Introduction

The Powder River Basin (PRB) coal region currently contains twenty-one active or proposed coal mines on a north-south trend within a 700 square mile area surrounding the towns of Gillette and Wright, Wyoming. Based on funding from the Abandoned Coal Mine Lands Research Program (ACMLRP), an intensive three-year study was conducted from 1991 to 1993 to determine the relationship between soil Se levels and plant uptake levels at native and reclaimed areas at two active coal mines within the PRB.

The main objective of this three-year study was to identify possible plant and soil Se relationships. A secondary role of that project was to determine differences in Se uptake between species growing on either native or reclaimed areas.

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This paper presents results of the 1991 through 1993 vegetation sampling programs. Results presented are preliminary as detailed statistical analysis of the specific vegetation encountered during the sampling has not been completed.

Methodology

Fieldwork was conducted on two active mines south of Gillette, Wyoming, within the PRB coal region of northeastern Wyoming. These two mines were ARCO's Black Thunder Mine (BTM), approximately 55 miles southeast of Gillette, and Coal Creek Mine (CCM), approximately 30 miles southeast of Gillette. Vegetation sampling was conducted in 1991 through 1993. Vegetation was sampled once in 1991 and twice in 1992 and 1993; the dual sampling in 1992/1993 was conducted to determine seasonal differences, i.e., May to late July/August.

Twenty-three native area (13 from BTM and 10 from CCM) sample locations were sampled during all three years. Reclaimed area sampling locations were: 79 (71 from BTM and 8 from CCM) in 1991 and 1993, and 52 (44 from BTM and 8 from CCM) in 1992. Within native areas, sample locations were randomly chosen to represent all vegetation types present on the active mine permit areas.

Sample locations were placed among reclaimed areas ranging from 2 to 10 years since revegetation. Sample locations were marked with steel metal fence posts, which were located 7.5 m south of the actual sample location, to prevent any deleterious effects on the vegetation from grazing animals.

Vegetation sampling was conducted at each of the 79 sample locations in 1991 and 1993; questionable vegetation analysis reduced the number of original 1991 sample locations. According to the original purpose of the study, a five to ten gram vegetation sample for selenium content was collected within a 3.5 m radius from the center of the site. Sampled plants included the dominant four species, based on a visual determination of relative site cover, and a composite grass sample designed to simulate herbivore grazing. General plant cover was visually estimated using a system that grouped plants into the following gross cover percent categories: <1, 1-10, 11-25, 26-50, 51-75, 76-100. It is this vegetation cover from which species composition and diversity were derived and are summarized in this paper.

Results

Based on a review of the compilation of the total number of species encountered during sampling by mine, type, and sampling period (Table 1), some conclusions can be drawn. One observation is the extremes in diversity between native and reclaimed areas. The number of species encountered on reclaimed areas at the Coal Creek Mine were approximately 50 to 70% of the number encountered during sampling of native areas. For the Black Thunder Mine, only 50 to 80% of the total number of species observed on native areas were determined in surveys of reclaimed areas. This is primarily due to the reduced number of species within a seed mix, but is also likely a product of topsoil stripping and placement, i.e., direct haul versus stockpiled material. The direct haul material is not stored and would contain a much greater source of viable seeds from recently stripped areas. Perennial forb reduction was noticeable at both Black Thunder and Coal Creek Mines.

In addition to the change in the number of species encountered during sampling, lifeforms generally changed as well. Prior to disturbance, the major lifeform categories at the Black Thunder Mine in terms of number of species encountered were cool season grasses, annual forbs, and perennial forbs. Annual forbs were conspicuously limited at the Coal Creek Mine, which may indicate less disturbance either from oilfield activity, grazing

Table 1. Number of different species encountered by lifeform category for each sampling period. Note: 1991 Black Thunder data based on selenium sampling, not cover estimates, which were done in 1992 and 1993.

Sampling Date	Mine	Type	CSG	WSG	AG	AF	PF	HS	FS	SUC	TOTAL
1991	BTM	N	4	0	0	0	2	1	1	0	8
		R	11	5	5	16	11	3	5	1	57
	CCM	N	19	1	3	10	33	4	3	1	74
		R	16	4	3	14	11	2	2	1	53
Early	BTM	N	15	2	1	17	37	4	2	1	79
1992		R	14	4	3	16	14	3	4	1	59
	CCM	N	14	1	2	2	35	5	3	1	63
		R	16	2	1	7	8	2	1	0	39
Late	BTM	N	14	4	2	11	33	3	2	1	70
1992		R	13	6	4	13	11	3	4	1	55
	CCM	N	15	1	3	3	39	5	3	1	70
		R	14	2	2	7	9	2	2	0	38
Early	BTM	N	14	2	3	13	38	3	1	1	75
1993		R	12	4	3	17	17	3	4	1	61
	CCM	N	16	1	3	7	44	5	4	1	81
		R	16	3	2	9	15	3	3	1	52
Late	BTM	N	19	5	3	19	42	4	1	1	94
1993		R	11	5	3	12	11	3	4	1	50
	CCM	N	11	1	1	3	20	4	3	1	44
		R	10	1	1	4	9	2	2	0	29

MINE: BTM = Black Thunder Mine; CCM = Coal Creek Mine
TYPE: N = Native; R = Reclaimed
LIFEFORM: CSG = Cool Season Grass; WSG = Warm Season Grass; AG = Annual Grass; AF = Annual Forb; PF = Perennial Forb; HS = Half Shrub; FS = Full Shrub; SUC = Succulent; TOTAL = Total Species

pressure, or roads. Cool season grasses were also a significant part of the overall numbers of species at the Coal Creek Mine. Curiously, the number of warm season grasses went up on the reclaimed areas versus native areas at the Coal Creek Mine; however, cover percentages were not similar to native areas.

Weather patterns played a significant role in the distribution of lifeform category numbers during the sampling years. During 1991, late spring rains and cooler temperatures favored cool season grass and annual forb growth. During 1992, temperature

Table 2. Examples of vegetation diversity and percent cover on old (reclaimed >5 years from time of sampling) and young (reclaimed <5 years from time of sampling) areas. Note: 1991 Black Thunder data based on selenium sampling, not cover estimates, which were done in 1992 and 1993.

Reclamation	Vegetation Species	1991	Early 1992	Late 1992	Early 1993	Late 1993
		% Vegetation Cover				
Old	Agrdas	0-1	10-25	1-10	1-10	10-25
	Agrsmi	10-25	1-10	10-25	10-25	10-25
	Agrtra	0-1			1-10	
	Stivir	0-1	1-10	1-10	1-10	1-10
Young	Agrcri		0-1	0-1	0-1	0-1
	Agrdas	1-10	1-10	1-10	1-10	1-10
	Agrsmi	1-10	1-10	1-10	1-10	1-10
	Agrtra	1-10	1-10	10-25	10-25	1-10
	Oryhym	0-1	0-1	0-1	1-10	0-1
	Sticom		0-1		1-10	
	Bougra	0-1	1-10	0-1	0-1	
	Medsat	1-10	1-10	0-1	10-25	1-10
	Ratcol	0-1	0-1	0-1	1-10	0-1
	Cerlan	0-1	1-10	0-1	0-1	0-1
	Atrcan	0-1	0-1	0-1	1-10	0-1
	Opupol	0-1	0-1	0-1	0-1	0-1

and moisture averages for June and July were reversed, which resulted in reduced growth of cool season grasses and a greater abundance of warm season grasses. During 1993, higher amounts of moisture were noted in the earlier part of the growing season, which resulted in increased cool season grass and perennial forb growth for that year.

Species diversity within reclaimed areas generally reflect the seed mix used in revegation practices specific to the different mines and to age of reclamation. This was especially apparent in older reclaimed areas, i.e., greater than 5 years old at the time of sampling, which generally reflect less diverse seed mixes or a lack of innovative reclamation practices such as dual seeding of cool season grasses separate from shrubs and warm season grasses (Table 2). For example, at the Black Thunder Mine, the number of species encountered in one sampling period ranged from approximately 10 in older seeded areas to approximately 30 in newer seeded areas.

Table 3. Comparison of Big Sagebrush Grasslands at Black Thunder and Coal Creek Mines. Note: no cover estimates were done in 1991 at the Black Thunder Mine.

Mine	Vegetation Species	1991	Early 1992	Late 1992	Early 1993	Late 1993
		% Vegetation Cover				
Black	Agrsmi		0-1	1-10	1-10	0-1
Thunder	Carste		1-10		0-1	0-1
NATIVE	Koemac		0-1	0-1	1-10	1-10
	Stivir		0-1	1-10	1-10	1-10
	Bougra		0-1	1-10	1-10	1-10
	Gaucoc		1-10	0-1	1-10	0-1
	Vicame		0-1	0-1	1-10	0-1
	Arttri		1-10	1-10	10-25	1-10
Coal	Agrdas	1-10	1-10	1-10	1-10	0-1
Creek	Agrsmi		1-10	0-1	0-1	0-1
NATIVE	Carfil		0-1	0-1	0-1	
	Koemac	1-10	1-10	1-10	1-10	0-1
	Stivir	1-10	1-10	1-10	1-10	1-10
	Bougra			0-1	0-1	0-1
	Oxylam	1-10	1-10	0-1	1-10	0-1
	Artfri		1-10	1-10	1-10	0-1
	Arttri	1-10	1-10	1-10	1-10	1-10

The two mines sampled in this study varied greatly in their overall plant species composition. Soils at the Black Thunder Mine tend to be sandier; historically high grazing pressure has resulted in an increase in pricklypear cactus and warm season grasses. Soils at the Coal Creek Mine are heavier, i.e., clayier; grazing pressure at Coal Creek has favored increased sagebrush cover. Table 3 is a comparison of the plots, one at each mine, that were located in the same native vegetation type, i.e., Big Sagebrush Grassland.

Plant cover percentages varied both between years and seasons. Cool season grass cover declined from the early to late sampling periods while warm season grass cover increased. Overall, annual forb and annual grass cover declined also over that same time interval, although this was species specific, especially with forbs. Perennial forb cover also generally declined over a season due to reduced leaf area.

Yearly variations were primarily due to differences in amount and timing of precipitation. Differences in later growth season precipitation such as found in 1992 was due not only to regional weather patterns but to thunderstorm placement as well. This could create large differences between the two mines, even though they are only approximately 20 miles apart. Weather within the Powder River Basin during 1991 was unusually cool and wet during April and May and extended to the end of June. During 1992, April and May were above normal for temperature and provided little or no precipitation while June and July were below normal for temperature and above average for precipitation. In many respects, the spring and summer sampling periods were practically reversed during 1992 despite the calendar months.

Overall plant species observed for each mine, type, and sampling period are found in Table 4. Some of the differences described above are evident in this table. For example, the following observations are based on the number of species encountered (not cover percentages). More cool season grasses and warm season grasses were observed in Coal Creek reclaimed than Coal Creek native sites. Annual forbs were limited in Coal Creek native, while annual grasses were more numerous in Coal Creek native versus reclaimed sites. The diversity of perennial forbs and half shrubs is higher on the Coal Creek native sites. Full shrubs were also more dominant on Coal Creek native lands.

At the Black Thunder Mine the diversity in cool season grasses and annual forbs between native and reclaimed areas was mixed; however, diversity of warm season grasses increased on Black Thunder reclaimed sites. Annual grasses were slightly more predominant on Black Thunder reclaimed sites. The number of perennial forbs encountered increased on native sites with time. With the exception of broom snakeweed, half shrubs were generally equal between Black Thunder native and reclaimed sites; however, full shrubs increased on Black Thunder reclaimed sites.

Discussion

The previously described results indicate the importance of seasonality on collecting vegetation data within a given year and overall weather conditions between years. It is extremely important to note weather patterns, especially temperature and monthly precipitation amounts and distribution. This is true, not only in sampling for selenium content of vegetation, but also for overall species diversity and composition.

In addition to the selenium information derived from this project, some useful trend information was documented. Trend information is highly useful to mining personnel and regulators who ultimately must decide on bond release criteria. Longterm vegetation composition data, for three or more consecutive years (as was done in this project) or for three years spread over a ten year bonding period, is extremely important in indicating overall vegetation patterns. Successes and failures can be documented and management strategies adjusted accordingly.

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Table 4. Plant species observed on native and reclaimed areas at Coal Creek and Black Thunder Mines during 1991 to 1993. Note: 1991 Black Thunder data based on selenium sampling, not cover estimates, which were done in 1992 and 1993.

			Coal Creek Native					Coal Creek Reclaimed					Black Thunder Native					Black Thunder Reclaimed				
Code	Scientific Name	Common Name	1991	Early 1992	Late 1992	Early 1993	Late 1993	1991	Early 1992	Late 1992	Early 1993	Late 1993	1991	Early 1992	Late 1992	Early 1993	Late 1993	1991	Early 1992	Late 1992	Early 1993	Late 1993
Cool Season Grasses																						
Agrcri	Agropyron cristatum	Crested Wheatgrass					X	X	X	X	X	X		X	X	X	X	X	X	X	X	X
Agrdas	Agropyron dasystachyum	Thickspike Wheatgrass	X	X	X	X	X	X	X	X	X	X		X	X	X	X	X	X	X	X	X
Agrine	Agropyron inerme	Beardless Wheatgrass	X						X	X	X								X	X		X
Agrint	Agropyron intermedium	Intermediate Wheatgrass						X		X	X											
Agrsmi	Agropyron smithii	Western Wheatgrass	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Agrspi	Agropyron spicatum	Bluebunch Wheatgrass	X	X	X	X	X		X	X	X	X						X	X	X	X	X
Agrtra	Agropyron trachchycaulum	Slender Wheatgrass	X					X	X	X	X	X						X	X	X	X	X
Agrtri	Agropyron trichophorum	Pubescent Wheatgrass																X	X	X		
Broine	Bromus inermis	Smooth Brome						X	X	X	X	X						X	X	X	X	X
Carfil	Carex filifolia	Threadleaf Sedge	X	X	X	X	X							X	X	X	X					
Carste	Carex stenophylla	Elk Sedge	X	X	X	X								X	X	X	X					
Disspi	Distichlis spicata	Inland Saltgrass	X	X	X	X	X									X	X					
Elycin	Elymus cinearus	Basin Wildrye	X						X	X	X											
Horjub	Hordeum jubatum	Foxtail Barley	X					X										X	X			
Koemac	Koeleria macrantha	Prairie Junegrass	X	X	X	X	X	X	X	X	X	X		X	X	X	X		X		X	
Oryhym	Oryzopsis hymenoides	Indian Ricegrass	X	X	X	X	X	X	X	X	X	X		X	X		X	X	X	X	X	X
Poaamp	Poa ampla	Big Bluegrass	X	X	X	X	X	X	X	X	X			X			X					
Poacan	Poa canbyi	Canby Bluegrass	X	X	X	X								X	X	X	X				X	X
Poacus	Poa cusickii	Cusick Bluegrass						X	X		X			X								
Poafen	Poa fendleriana	Fendler Bluegrass												X		X	X					
Poajun	Poa juncifolia	Alkali Bluegrass	X	X	X	X																
Poapra	Poa pratensis	Kentucky Bluegrass	X		X	X		X						X			X					
Poasan	Poa sandbergii	Sandberg Bluegrass	X	X	X	X								X	X	X	X		X		X	
Poasp	Poa sp.	Bluegrass				X		X				X	X	X	X	X	X			X		
Schpan	Schedonnardus paniculatus	Tumblegrass													X	X	X					
Sticom	Stipa comata	Needle and Thread	X	X	X	X	X	X	X	X	X		X	X	X	X	X	X	X	X	X	
Stivir	Stipa viridula	Green Needlegrass	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	

Table 4. Continued

Code	Scientific Name	Common Name	Coal Creek Native					Coal Creek Reclaimed					Black Thunder Native					Black Thunder Reclaimed				
			1991	Early 1992	Late 1992	Early 1993	Late 1993	1991	Early 1992	Late 1992	Early 1993	Late 1993	1991	Early 1992	Late 1992	Early 1993	Late 1993	1991	Early 1992	Late 1992	Early 1993	
Warm Season Grasses																						
Andsco	Andropogon scoparius	Little Bluestem						X			X				X			X	X	X	X	X
Boucur	Bouteloua curtipendula	Sideoats Grama						X	X	X	X	X						X		X		X
Bougra	Bouteloua gracilis	Blue Grama	X	X	X	X	X	X	X	X	X			X	X	X	X	X	X	X	X	X
Callon	Calamovilfa longifolia	Prairie Sandreed						X									X	X	X	X	X	X
Spoair	Sporobolus airoides	Alkali Sacaton													X	X	X			X		
Spocry	Sporobolus cryptandrus	Sand Dropseed												X			X	X	X	X	X	X
Annual Grasses																						
Brojap	Bromus japonicus	Japanese Brome	X	X	X	X	X	X		X	X				X	X	X	X	X	X	X	X
Brotec	Bromus tectorum	Cheatgrass Brome	X	X	X	X		X				X		X	X	X	X	X	X	X	X	X
Fescot	Festuca occidentalis	Sixweeks Fescue	X		X	X										X	X	X				
Triaes	Triticum aestivum	Cultivated Wheat						X	X	X	X							X	X		X	
Annual Forbs																						
Alydes	Alyssum desertorum	Desert Alyssum	X					X			X						X	X				
Alyaly	Alyssum alyssoides	Pale Alyssum						X		X	X				X	X	X				X	X
Alyspp	Alyssum spp.	Alyssum						X	X										X			
Cammic	Camelina microcarpa	Littleseed Falseflax	X			X	X	X		X	X	X		X	X	X	X	X	X	X	X	X
Chealb	Chenopodium album	Lambsquarters						X	X	X	X			X			X		X	X	X	X
Chelep	Chenopodium leptophyllum	Chenopodium												X			X		X			
Chensp.	Chenopodium sp.	Chenopodium						X							X			X		X	X	
Cirsium	Cirsium sp.	Thistle					X							X	X							
Cirvul	Cirsium vulgare	Bull Thistle	X			X								X	X	X	X					
Despin	Descurainia pinnata	Pinnate Tansymustard						X	X	X	X	X		X		X	X	X	X	X	X	
Dessop	Descurainia sophia	Tansymustard	X																X		X	X
Gnapal	Gnapthalium palustre	Cudweed												X		X	X					

Table 4. Continued

Code	Scientific Name	Common Name	Coal Creek Native					Coal Creek Reclaimed					Black Thunder Native					Black Thunder Reclaimed				
			1991	Early 1992	Late 1992	Early 1993	Late 1993	1991	Early 1992	Late 1992	Early 1993	Late 1993	1991	Early 1992	Late 1992	Early 1993	Late 1993	1991	Early 1992	Late 1992	Early 1993	
Kocsco	Kochia scoparia	Kochia						X	X	X	X			X	X	X	X	X	X	X	X	X
Lacser	Lactuce serriola	Prickly Lettuce										X		X	X		X					
Lepden	Lepidium densiflorum	Prairie Peppergrass			X	X								X		X	X	X	X	X	X	X
Lapred	Lappula redowskii	Stickseed									X			X	X	X	X	X	X	X	X	
Luppus	Lupinus pusillis	Lupine												X	X	X	X	X			X	X
Meloff	Melilotus officinalis	Yellow Sweetclover	X	X	X	X	X	X	X	X	X	X						X	X	X	X	X
Micgra	Microsteris gracilis	Microsteris												X		X	X					
Phalin	Phacelia linearis	Phacelia												X		X	X					
Plapat	Plantago patagonica	Pursh's Plantain	X	X	X	X								X	X	X	X	X	X	X	X	
Polavi	Polygonum aviculare	Prostrate Knotweed						X	X									X				
Salaus	Salsola australis	Russian Thistle						X	X	X	X				X		X	X	X	X	X	X
Sisalt	Sisymbrium altissimum	Tumbling Hedgemustard						X							X				X	X	X	X
Sisloe	Sisymbrium loeselii	Hedgemustard																X	X		X	X
Solros	Solanum rostratum	Nightshade						X														
Thlarv	Thlaspi arvense	Field Pennycress	X			X		X														
Perennial Forbs																						
Achmil	Achillea millefolium	Common Yarrow	X	X	X	X	X	X			X			X	X	X	X	X	X			
Alltex	Allium textile	Prairie Onion	X	X	X	X					X			X		X	X				X	X
Antpar	Antennaria parvifolia	Little Pussytoes			X	X										X						
Arehoo	Arenaria hookeri	Hooker Sandwort	X	X	X	X	X		X					X	X	X	X				X	
Arnica	Arnica sp.	Arnica												X	X	X	X					
Artlud	Artemisia ludoviciana	Louisiana Sagewort				X								X	X	X	X		X			
Astbis	Astragalus bisulcatus	Two-grooved Milkvetch	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X		X	X	X	X
Astcic	Astragalus cicer	Cicer Milkvetch	X					X	X	X	X	X				X		X	X	X	X	X
Astfal	Astragalus falcatus	Aster	X		X	X																

Table 4. Continued

Code	Scientific Name	Common Name	Coal Creek Native					Coal Creek Reclaimed					Black Thunder Native					Black Thunder Reclaimed				
			1991	Early 1992	Late 1992	Early 1993	Late 1993	1991	Early 1992	Late 1992	Early 1993	Late 1993	1991	Early 1992	Late 1992	Early 1993	Late 1993	1991	Early 1992	Late 1992	Early 1993	
Astmol	Astragalus mollis	Milkvetch	X	X	X	X																
Astpur	Astragalus purshii	Wooly Milkvetch	X			X	X															
Astspa	Astragalus spatulatus	Spoonleaf Milkvetch	X	X	X	X	X							X	X	X	X					
Astspp	Astragalus sp.	Milkvetch	X											X	X		X			X		
Cirarv	Cirsium arvense	Canadian Thistle			X	X								X		X	X					
Comumb	Comandra umbellata	Bastard Toadflax		X	X	X								X	X	X	X					
Crepis	Crepis sp.	Hawksbeard												X	X	X	X					
Crycel	Cryptantha celosioides	Cryptantha	X	X		X																
Cryspp	Cryptantha sp.	Cryptantha					X							X	X	X	X					X
Dalcan	Dalea candida	Prairie Clover													X	X	X					
Equlae	Equisetum laevigatum	Horsetail												X	X	X	X					
Erigeron	Erigeron sp.	Fleabane												X			X			X		
Eripum	Erigeron pumilus	Daisy Fleabane	X	X	X	X								X	X							
Erioch	Erigeron ochroleucus	Fleabane		X		X	X				X			X	X	X	X					
Gaucoc	Gaura coccinea	Scarlet Gaura	X	X	X	X	X							X	X	X	X	X				
Glylep	Glycyrrhiza lepidota	Wild Licorice	X	X	X	X	X															
Grisqu	Grindelia squarrosa	Curlycup Gumweed	X	X	X	X	X															
Hapnut	Haplopappus nuttalianus	Nuttal goldenweed	X	X	X	X	X															
Hetvil	Heterotheca villosa	Wooly Goldenaster												X	X	X	X					
Ipocon	Ipomopsis congesta	Ipomopsis		X	X	X																
Ivaaxi	Iva axillaris	Iva	X		X	X																
Lesare	Lesquerella arenosa	Sand Bladderpod	X	X	X	X																
Lewred	Lewisia rediviva	Bitterroot		X	X	X																
Lomfoe	Lomatium foeniculaceum	Biscuitroot				X								X		X	X					
Lomori	Lomatium orientale	Lomatium	X	X	X	X					X			X	X	X	X					

Table 4. Continued

Code	Scientific Name	Common Name	Coal Creek Native					Coal Creek Reclaimed					Black Thunder Native					Black Thunder Reclaimed				
			1991	Early 1992	Late 1992	Early 1993	Late 1993	1991	Early 1992	Late 1992	Early 1993	Late 1993	1991	Early 1992	Late 1992	Early 1993	Late 1993	1991	Early 1992	Late 1992	Early 1993	
Lupinus	Lupinus sp.	Lupine												X	X	X	X					
Lygjun	Lygodesmia juncea	Skeletonweed												X	X	X	X					
Macgri	Macaeranthera grindelioides	Machaeranthera												X		X	X					
Medsat	Medicago sativa	Alfalfa				X	X	X	X	X	X	X						X	X	X	X	X
Musdiv	Musineon divaricatum	Biscuitroot	X	X	X	X										X	X	X				
Oencae	Oenothera caespitosa	Primrose	X	X	X	X															X	
Oenosp.	Oenothera sp.	Primrose												X		X	X					
Onovic	Onobrychis viciifolia	Sanfoin						X	X	X	X	X						X	X	X	X	X
Oxylam	Oxytropis lambertii	Lambert Locoweed		X	X	X	X										X					
Oxytropis	Oxytropis sp.	Locoweed												X		X	X					
Penalb	Penstemon albus	White Beardtongue		X	X	X									X	X	X		X		X	
Peneri	Penstemon eriantherus	Beardtongue		X	X	X					X			X	X	X	X					
Petsp	Petalostemon sp.	Petalostemon												X	X		X					
Phlhoo	Phlox hoodii	Hood's Phlox	X	X	X	X	X							X	X	X	X					
Picopp	Picrodeniopsis oppositifolia	Picrodeniopsis	X	X	X	X	X	X	X	X	X	X		X	X	X	X					
Psoarg	Psoralea argophylla	Silverleaf Scurfpea				X								X	X	X	X					
Psoralea	Psoralea sp.	Scurfpea														X	X		X			
Psoesc	Psoralea esculenta	Indian Beardroot	X	X	X																	
Ratcol	Ratibida columnifera	Prairie Coneflower						X		X	X	X						X	X	X	X	X
Sencan	Senecio canus	Senecio	X	X	X	X																
Solmis	Solidago missouriensis	Prairie Goldenrod	X	X	X	X																
Sphcoc	Sphaeralcea coccinea	Scarlet Globemallow	X	X	X	X	X						X	X	X	X	X	X	X		X	
Taroff	Taraxacum officinale	Dandelion	X	X	X	X	X	X		X	X	X		X	X	X	X	X	X	X	X	X
Therho	Thermopsis rhombifolia	Yellow Pea	X	X	X	X								X	X	X	X				X	X
Tradub	Tragapogon dubius	Yellow Salsify	X	X	X	X	X	X				X		X	X		X	X	X	X	X	X

Table 4. Continued

Code	Scientific Name	Common Name	Coal Creek Native					Coal Creek Reclaimed					Black Thunder Native					Black Thunder Reclaimed				
			1991	Early 1992	Late 1992	Early 1993	Late 1993	1991	Early 1992	Late 1992	Early 1993	Late 1993	1991	Early 1992	Late 1992	Early 1993	Late 1993	1991	Early 1992	Late 1992	Early 1993	
Trirep	Trifolium repens	Clover						X	X	X	X											
Vicame	Vicia americana	American Vetch	X	X	X	X	X	X	X	X	X	X		X	X	X	X		X	X	X	
Half Shrub																						
Artfri	Artemisia frigida	Fringed Sagewort	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Atrgar	Atriplex gardneri	Gardner's Salatbush	X	X	X	X	X				X			X	X	X	X	X	X	X	X	X
Cerlan	Ceratoides lanata	Winterfat												X			X	X	X	X	X	X
Gutsar	Gutierrezia sarothrae	Broom Snakeweed	X	X	X	X	X	X		X	X	X		X	X	X	X					
Xylgla	Xylorrhiza glabriuscula		X	X	X	X	X															
Full Shrub																						
Arttri	Artemisia tridentata	Big Sagebrush	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X		X	X	X	X
Artcan	Artemisia cana	Silver Sagebrush	X	X		X	X											X	X	X	X	X
Atrcan	Atriplex canescens	Fourwing Saltbush	X		X	X	X	X	X	X	X							X	X	X	X	X
Chrnau	Chrysothamnus nauseosus	Rubber Rabbitbrush									X							X	X	X	X	X
Roswoo	Rosa woodsii	Wood's Rose		X	X	X												X				
Symocc	Symphoricarpos occidentalis	Western Snowberry												X	X							
Succulent																						
Opupol	Opuntia polyacantha	Plains Pricklypear	X	X	X	X	X	X	X		X			X	X	X	X	X	X	X	X	X

NATURAL AND PLANTED FLORA OF THE LOG MOUNTAIN SURFACE-MINED DEMONSTRATION AREA, BELL COUNTY, KENTUCKY[1]

by

Ralph L. Thompson, Gary L. Wade, and R. Allen Straw[2]

Abstract. A descriptive study of the naturally invading and planted flora was conducted during 1984-1985 on a 14- and 21-year-old contour surface mine, the 14.2 ha Log Mountain Demonstration Area (LMDA), in Bell County, Kentucky. Six habitats are designated from areas created from coal mining; the 1963 bench, 1970 bench, bench highwalls, mine outslopes, mine seeps, and coal haul-telephone microwave tower road. Twenty-four of 25 woody and herbaceous species (11 indigenous, 13 non-indigenous) have persisted from plantings by personnel of the Northeastern Forest Experiment Station, USDA Forest Service. We recommend 11 native and exotic woody and herbaceous species for planting on coal surface-mined areas. An annotated list of vascular plants comprises 360 taxa (286 indigenous, 74 non-indigenous) in 224 genera from 82 families. Taxa consist of 1 Lycopodiophyta, 1 Equisetophyta, 8 Polypodiophyta, 7 Pinophyta, and 343 Magnoliophyta. The most species-rich families are the Asteraceae (64), Poaceae (39), Fabaceae (20), Cyperaceae (16), Rosaceae (13), and Lamiaceae (11). A total of 155 Bell County distribution records were documented. Three threatened Kentucky species (*Gentiana decora, Liparis loeselii, Silene ovata*) were present in refugial habitats created by surface mining. The high species richness has resulted from native and naturalized invading species from the environs, native and exotic planted species, and species from the remnant seed bank. Forest vegetation is a complex mosaic of natural and semi-natural plant communities on the unplanted and planted areas of LMDA .

Additional Key Words: biodiversity, plant invasion, plant succession, reclamation, refugium.

Introduction

Coal mine operators have been required to reclaim and revegetate surface-mined spoils even before the passage of Public Law 95-87, the Surface-Mining Control and Reclamation Act of 1977 (SMCRA). From 1963-1971, the Northeastern Forest Experiment Station, USDA Forest Service, Berea, Kentucky, cooperated with several private coal companies in the planting of native and exotic reclamation species on five surface-mined sites in eastern Kentucky. Four of these surface-mined areas were selected as pre-SMCRA demonstration or research sites by the Federal-State Interagency Research Coordinating Committee (FSIRCC). Reclamation goals were to prevent erosion and siltation, foster soil development and forest succession, provide suitable wildlife habitat, determine potential species for forest products, and contribute to landscape aesthetics.

We inventoried the entire flora of the Log Mountain Demonstration Area (LMDA), a 14- and 21-year-old pre-SMCRA contour surface-mined site in Bell County, Kentucky, during 1984-1985. The LMDA constitutes 14.2 ha; a 9.1 ha area mined in 1963 and an additional 5.1 ha area mined in 1970. The original 9.1 ha area was selected by the FSIRCC in 1965 for reclamation plantings toward forestry postmining land use.

Our study objectives at LMDA were to: 1) report the success of planted native and exotic species after 14 and 21 years; 2) describe the establishment of native and exotic volunteer species; 3) describe the plant communities and plant succession in the different surface-mined habitats; and, 4) compile an annotated list of vascular plants.

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Complete floristic inventories, vegetation development studies, and success of natural and planted species observations have also been made on an 18-year-old area-type mine in Laurel County (Thompson et al. 1984), and on a 12-year-old contour surface-mined area in Rockcastle County (Thompson and Wade 1991). Assessments of experimental plantings at the Laurel County site and at a 20-year-old mined site in Bell County have revealed the potential of those sites for forestry, wildlife habitat, soil building, and site stabilization (Wade et al. 1985, 1986).

Methods

We collected representative vascular plant specimens approximately every two weeks during the growing season from March 1984 through October 1985. Voucher specimens were processed according to standard herbarium procedures and deposited in the Berea College Herbarium (BEREA). Manuals used for plant identification were Gleason and Cronquist (1963) and Strausbaugh and Core (1978). Plant classification and nomenclature follow Gleason and Cronquist (1991).

The LMDA is divided into two contour-mined sites; "Site A" which was mined in 1963, and "Site B" which was mined in 1970. Reclamation woody plantations within these two sites are designated as "Areas 1-10." The physical environments which have developed on LMDA are designated as "habitats." In some cases, these "habitats" may cross the boundaries of both site A and B, and different area designations may overlap, e.g., mine highwalls, outslopes, and seeps.

Table 1 shows the pH of 30 soil samples from the surface layer (0-17 cm) in nine plantation areas in July 1985 at LMDA. Soil samples were analyzed from seven plantation areas on the 1963 site A bench, the mixed pine plantation above the 1963 highwall, and the 1970 site B bench (Dan Childress, NRCS Scientist, unpublished data 1985).

Table 2 reports the complete inventory of the entire 1963 site A bench in August 1985 for total seedlings, saplings, and trees by diameter at breast height (dbh). This census includes planted Areas 1-9. Percentage composition or relative density was calculated from the tree population results according to Brower and Zar (1977).

Table 3, the annotated list of vascular plants, is arranged alphabetically by division, family, and species. All non-indigenous taxa, naturalized or introduced, are preceded by an asterisk (*). Those non-indigenous taxa planted as reclamation species are preceded by a (+). Native reclamation species planted on LMDA are coded by a circle (o). Bell County distribution records are indicated with a dagger (†). These county records are based on herbarium specimens on deposit at (BEREA), the University of Tennessee Herbarium (TENN), and the University of Kentucky Herbarium (UK), and on literature citations (Braun 1943, Ettman 1976, Cranfill 1980, Hinkle 1975, Beal and Thieret 1986, Pounds et al. 1987). Vernacular or common names follow Strausbaugh and Core (1978), Gleason and Crinquist (1991), and Swink and Wilhelm (1994).

From field reconnaissance, we identified six different habitats on LMDA based on the effects of 1963 and 1970 mining procedures, reclamation processes, topographic terrain features, and present plant communities. In Table 3, these six habitats are designated as: 1=1963 mine bench, 2=1970 mine bench, 3=mine highwalls, 4=mine outslopes, 5=mine seeps, and 6=coal haul-telephone microwave tower road.

Relative abundance for each taxon was determined by field observations from each habitat. Each taxon was compared with numbers of individuals or colonies from other taxa of similar growth habit and duration. In Table 3, the abundance scale values are: Rare (R)=1-5 individuals or isolated colonies, Infrequent (I)=6-25 individuals or colonies, Occasional (O)=26-100 individuals or colonies; Frequent (F)=100s of individuals or colonies; and Abundant (A)=1000s of individuals or colonies.

The evaluation of native and planted species and plant successional trends are based upon our personal observation, tree sampling data, and unpublished planting documentation from the USDA Forest Service Northeastern Forest Experiment Station, Berea, Kentucky.

Log Mountain Demonstration Area

The Environmental Site

The LMDA, is a WNW-trending, 14.2-ha contour surface-mined area, which is located 18 km west of Middlesboro off Kentucky Highway 74 in southwestern Bell County. The mine lies within the Log Mountains at Maiden Ridge at latitude 36° 37' 05" N and longitude 85° 51' 01" W. Elevation of the study site ranges from 850 m on the 1963 lower outslope to 908 m at the ridge crest bordered by the coal haul-telephone microwave tower road (Fig. 1).

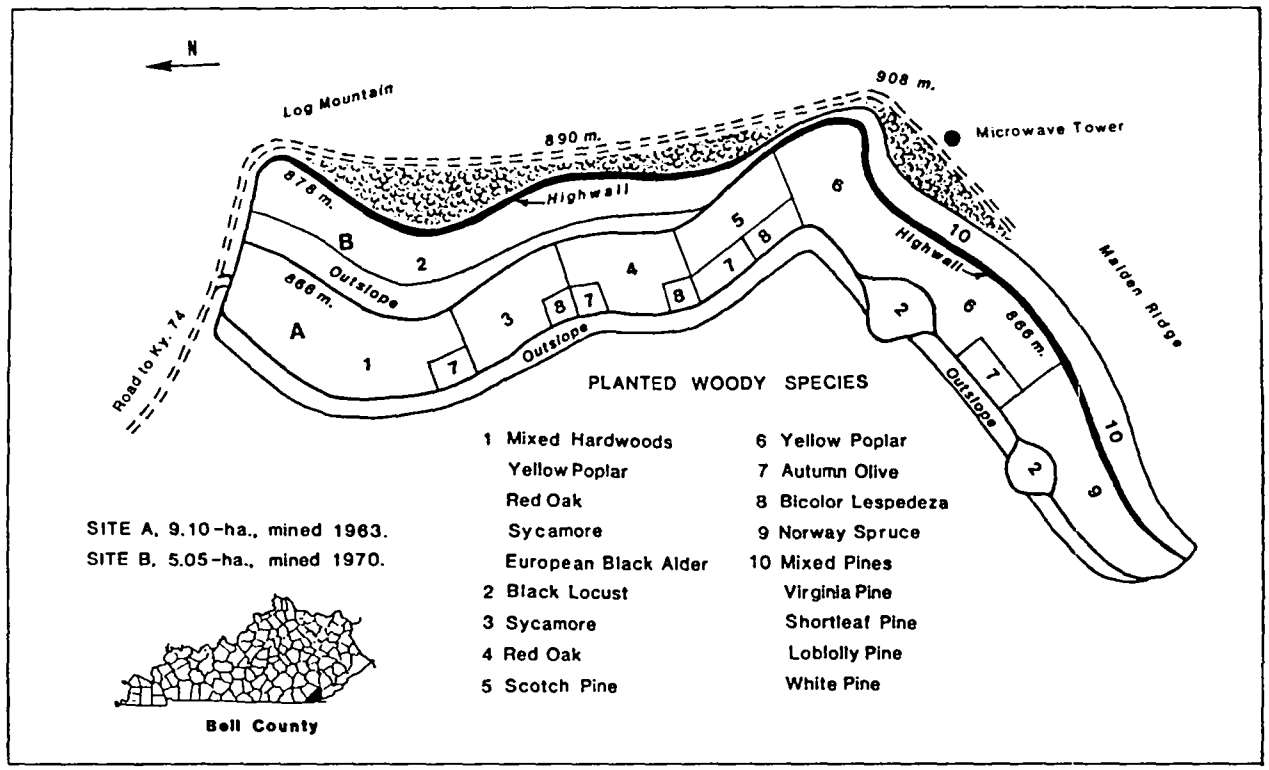

Figure 1. Log Mountain Demonstration Area, Bell County, Kentucky

Physiography and Vegetation. Fenneman (1938) described the Log Mountains as a small mountain range near the middle of the broad syncline between the Cumberland Mountains and Pine Mountains of the Appalachian Plateaus Province. The upper slope and ridgetop forest of the Log Mountains are described as the All Deciduous Mixed Mesophytic Community Type of the Mixed Mesophytic Forest Region by Braun (1950). The Cumberland Mountains have been described as the richest center of the Mixed Mesophytic Forest where several important canopy trees characteristically share dominance (Braun 1950).

Geology. Bedrock of Log Mountain and Maiden Ridge belongs to the Bryson Formation of the Lower to Middle Pennsylvanian Series of the Pennsylvanian System. The lithology of the 112 m thick Bryson Formation consists of siltstones, claystones, sandstones, and four coal beds. The Red Springs coal bed, the major coal seam of 1.0-2.0 m, forms the basement layer of the formation. A thin, unnamed coal rider in sandstone-filled channels lies above the Red Springs coal bed (Rice and Maughan 1978).

Soils. The surface-mined soils in the Log Mountain region have been classified as part of the Fairpoint soils series, a loamy-skeletal, mixed, nonacid, mesic Typic Udorthents. These soils are nonacid (pH 6.6-7.3) very deep (>60 inches), well-drained, and moderately permeable. They are found in coal-mined mountain ridges and slopes within the Cumberland Mountains (Childress 1992).

Climate. The nearest United States Weather Bureau station is located at Middlesboro at 358 m elevation. During the 1961-1990 period, the mean annual precipitation was 130 cm including a mean of 37 cm snowfall. Mean annual temperature was 13.3°C with the mean winter minimum -4°C and the mean summer maximum 23°C. The mean growing season (based on 0°C) was 170 days (University of Kentucky Agricultural Weather Center 1995).

LMDA Mined Sites

Surface-mining was done at two different time periods at LMDA. In 1963, Site A , an area of 9.1 ha was created by the contour mining of the Red Springs coal bed. A 16 m WNW vertical highwall was formed by the final mining cuts. The nearly level bench extends from the highwall to a steep 35-50 percent outslope at 866 m elevation. This outslope was produced from excess overburden spoils pushed over unmined forest soils.

In 1970, Site B, a tract of 5.1 ha, was formed by the contour mining of a thin, unnamed rider coal seam above the Red Springs coal bed. Part of the 1963 highwall was included in this mining. A second vertical W-trending 4 m highwall and a 2-6 percent sloping bench were created. Elevation varied from 878 m on the bench to 882 m at the top of the highwall. Additional mine spoils were displaced over the 1963 highwall area to form a steep 20-35 percent outslope down to the 1963 mine bench.

A continuous highwall was formed from the 1970 mining and 1963 mining of Log Mountain at the junction of the forested ridge top at Maiden Ridge. A compacted gravel-stone road initially used as a coal haul road presently serves as an access road from the telephone microwave tower to Kentucky Highway 74. This access road forms the eastern boundary of the LMDA and lies on the ridgetop of Log Mountain and Maiden Ridge (Fig. 1).

A 4.7 ha stand of mixed mesophytic hardwood forest lies above the highwall habitat between the access road. This forest has been disturbed from the mining processes by soil scraping, coal haul road construction, and past timbering practices. Wet seep pools have formed on the 1963 site A bench from the highwall in two planted areas, and on the 1970 site B bench at the highwall base.

From LMDA, we have designated six habitats from the 1963 and 1970 mining procedures, topographic features, reclamation processes, and existing plant communities: the 1963 mine bench, 1971 mine bench, mine highwalls, mine outslopes, mine seeps, and coal haul-telephone tower road.

LMDA Revegetation Efforts

Twenty-five species were planted during 1964, 1965, 1971, and 1984 on the LMDA. These species consisted of 11 native trees, four non-indigenous trees, two non-indigenous shrubs, and eight non-indigenous perennial herbs. All 25 species have been recommended for revegetation of surface-mined lands in the eastern United States (Vogel 1981).

In the spring of 1964, the 1963 site A mined area was seeded with Korean lespedeza (*Lespedeza stipulacea*) and tall fescue (*Festuca elatior*) for initial ground cover. In 1965, 10 demonstration areas were established and 13 woody species were planted to promote postmining forestry land use (Fig. 1). Four nitrogen-fixing tree and shrub species were planted on the bench; black locust (*Robinia pseudoacacia*), European black alder (*Alnus glutinosa*), bicolor lespedeza (*Lespedeza bicolor*), and autumn olive (*Elaeagnus umbellata*).

Nine native and non-indigenous tree species of potential forestry value were also planted in mixed hardwood, mixed pine, and individual hardwood and conifer plantations on the 1963 mined area. Yellow poplar (*Liriodendron tulipifera*), American sycamore (*Platanus occidentalis*), northern red oak (*Quercus rubra*), were planted in mixed hardwood blocks and on individual areas on the bench.

Table 1. LMDA soil pH from the surface layer (0-17 cm) at nine areas (Fig. 1).

Area	Mean pH Value	pH Range
1963 Bench Site A		
Area 1: Mixed hardwoods	5.3	4.9-6.6
Area 4: Northern red oak	6.4	6.1-6.6
Area 5: Scotch pine	6.1	6.0-6.1
Area 6: Yellow poplar	7.3	6.1-7.7*
Area 7: Autumn olive	7.4	7.2-7.6*
Area 8: Bicolor lespedeza	7.1	6.9-7.2*
Area 9: Norway spruce	4.9	4.8-5.1
1963 Highwall		
Area 10: Mixed Pine	4.3	4.0-4.9
1970 Bench Site B		
Area 2: Black locust	5.7	5.6-5.9

Code: Three samples from each area except six samples from Area 2; (*) carbonates present.

The non-native Scotch pine (*Pinus sylvestris*) and Norway spruce (*Picea abies*) were planted in individual bench areas. A mixed block of Virginia pine (*Pinus virginiana*), shortleaf pine (*P. echinata*), loblolly pine (*P. taeda*), and eastern white pine (*P. strobus*) was established above the highwall rim adjacent to the disturbed mixed mesophytic hardwood forest. In 1965, two perennial non-indigenous herbs, crown-vetch (*Coronilla varia*) and creeping foxtail (*Alopecurus arundinaceus*) were also seeded in two areas on the bench.

The 1970 site B was hydroseeded in 1971 on the bench and outslope with a grass-legume herbaceous mixture of tall fescue and nitrogen-fixing

species; sericea lespedeza (*Lespedeza cuneata*), red clover (*Trifolium pratense*), yellow sweet clover (*Melilotus officinalis*), white sweet clover (*M. alba*), and black locust. In 1984, seven tree species were interplanted in 48 plots with 24 seedlings each on the 1970 site B bench after herbiciding the herbaceous ground cover. These planted species were white ash (*Fraxinus americana*), northern red oak, saw-tooth oak (*Quercus acutissima*), black walnut (*Juglans nigra*), sugar maple (*Acer saccharum*), yellow poplar, and eastern white pine.

LMDA Minesoils

Minesoils are representative of the Fairpoint soil series derived from siltstone, sandstone, claystone, and coal intermixed with original forest soils. From the soil analysis of the 30 samples at LMDA, the mean pH value of the 1963 bench, 1970 bench, and 1963 highwall was 6.1 with a range from 4.0 to 7.7 pH. The mean pH value on the site A bench was 6.4 and varied from 4.9 to 7.7 with some free carbonates present. On the site B bench, the mean pH value was 5.7. The mean pH of the 1963 highwall mixed pine plantation was 4.3 and a range from 4.0-4.9 (Table 1). In general, the LMDA minesoils are less acid than unmined contiguous areas because of the basic nature of the carbonates contained in some of the Fairpoint soils (Dan Childress, NRCS Scientist, unpublished data 1995).

Results

LMDA Habitats

In 14 and 21 years after coal mining, a complex vegetational mosaic of natural and semi-natural plant communities is developing in the seven habitats from the mining effects and reclamation processes. Plant communities are progressing through various seral stages of secondary succession toward a young mixed hardwood forest throughout the planted and unplanted mine habitats.

1963 Site A Bench. Environments range from seasonally hydric sites near the highwall and open grassy xeric areas near the outslope to mixed mesic forested areas. A tree population analysis of 7,320 individuals reveals the development of a young mesophytic hardwood-dominated forest. Different degrees of canopy closure occur from volunteer native and exotic woody and herbaceous species from the contiguous forest environs, the nine woody plantations, and the existing spoil seed bank (Table 2). The highest species richness, 223 taxa, is found on this habitat (Table 3).

The most significant native trees colonizing the 1963 bench are white ash, yellow poplar, sugar maple, black locust, red maple (*Acer rubrum*), and Virginia pine (Table 2). Important nitrogen-fixing woody species spreading from the 1963 bench plantings are black locust, autumn olive, and European black alder. Shade-intolerant, mid-successional trees, sourwood (*Oxydendrum arboreum*), wild black cherry (*Prunus serotina*), eastern redbud (*Cercis canadensis*), white sassafras (*Sassafras albidum*), red elm (*Ulmus rubra*), blackgum (*Nyssa sylvatica*), and eastern red cedar (*Juniperus virginiana*), are volunteering on the bench (Table 2).

Shrubs present are American elderberry (*Sambucus canadensis*), wild hydrangea (*Hydrangea arborescens*), winged sumac (*Rhus copallina*), smooth sumac (*R. glabra*), and bicolor lespedeza. Characteristic woody vines on the bench are virgin's bower (*Clematis virginiana*), poison ivy (*Toxicodendron radicans*), moonseed (*Menispermum canadense*), Japanese honeysuckle (*Lonicera japonica*), glaucous greenbrier (*Smilax glauca*), common greenbrier (*S. rotundifolia*), summer grape (*Vitis aestivalis*), and Virginia creeper (*Parthenocissus quinquefolia*).

The numerically most important herbs are members of the Asteraceae, Poaceae, Fabaceae, Cyperaceae, and Rosaceae. Representative mesic woodland herbs include eulalia (*Microstegium vimineum*), pale jewelweed (*Impatiens pallida*), white snakeroot (*Eupatorium rugosum*), joe-pye weed (*E. fistulosum*), late boneset (*E. serotinum*), crownbeard (*Verbesina alternifolia*), sweet cicely (*Osmorhiza claytonii*), honewort (*Cryptotaenia canadensis*), black raspberry (*Rubus occidentalis*), wood nettle (*Laportea canadensis*), climbing buckwheat (*Polygonum scandens*), pokeweed (*Phytolacca americana*), Indian hemp (*Apocynum cannabinum*), cinnamon vine (*Dioscorea batatas*), deer-tongue panicum (*Panicum clandestinum*), cleavers (*Galium aparine*), spring beauty (*Claytonia virginica*), and violets (*Viola* spp.).

In drier exposed bench areas, herbs include tall fescue, broomsedge (*Andropogon virginicus*), hairy panic grass (*Panicum lanuginosum*), Kentucky bluegrass (*Poa pratensis*), Canadian goldenrod (*Solidago canadensis*), old-field goldenrod (*S. nemoralis*), rough goldenrod (*S. rugosa*), hairy white aster (*Aster pilosus*), ox-eye daisy (*Chrysanthemum leucanthemum*), wild carrot (*Daucus carota*), Allegheny blackberry (*Rubus allegheniensis*), northern dewberry (*R. flagellaris*), common cinquefoil (*Potentilla simplex*), wild strawberry (*Fragaria virginiana*), Japanese knotweed (*Polygonum cuspidatum*), sericea lespedeza, and Korean lespedeza.

Table 2. Tree population analysis on the1963 bench, Log Mountain Demonstration Area, Kentucky.

Species	Seedlings <1.0 m tall	Saplings <0.5	Decimenter Size Classes (dbh) 0.5-1.0	1.0-2.0	2.0-3.0	3.0-4.0	Species Total	Percentage Composition
Fraxinus americana	286	1118	151	27	3	1	1586	21.67
o*Liriodendron tulipifera*	311	301	180	214	55	6	1067	14.58
Acer saccharum	453	528	57	2	0	0	1040	14.21
o*Robinia pseudoacacia*	203	325	130	89	51	14	812	11.09
Acer ruburm	317	377	54	4	1	0	753	10.29
+*Alnus glutinosa*	165	229	53	35	19	7	508	6.94
o*Pinus virginiana*	95	171	33	7	0	0	306	4.18
o*Platanus occidentalis*	52	125	37	21	1	0	236	3.22
+*Pinus sylvestris*	0	12	63	92	9	0	176	2.40
o*Quercus rubra*	31	86	31	9	1	0	158	2.16
Oxydendrum arboreum	58	97	2	0	0	0	157	2.14
+*Picea abies*	0	13	45	90	8	0	156	2.13
Prunus serotina	38	43	12	5	4	3	105	1.43
Salix nigra	24	45	5	3	0	0	77	1.05
Sassafras albidum	28	25	5	3	0	0	61	0.83
Cornus florida	2	30	4	0	0	0	36	0.49
Tilia americana	2	7	3	5	0	0	17	0.23
Nyssa sylvatica	7	5	1	3	0	0	16	0.22
Juniperus virginiana	1	11	2	0	0	0	14	0.19
Cercis canadensis	6	5	0	0	0	0	11	0.15
Ulmus rubra	1	5	0	0	0	0	6	0.08
Carya cordiformis	1	3	1	0	0	0	5	0.07
Liquidambar styraciflua	1	1	2	1	0	0	5	0.07
Quercus velutina	3	1	0	0	0	0	4	0.06
Aesculus flava	2	2	0	0	0	0	4	0.06
Betula lenta	1	1	0	0	0	0	2	0.03
Magnolia acuminata	1	1	0	0	0	0	2	0.03
Total Species: 27	2089	3567	871	610	152	31	7320	100.00

Code: (*) Percentage Composition=Relative Density (Brower and Zar 1977).
(+) Planted nonindigenous trees on bench.
(o) Planted native trees on bench.

The nine woody demonstration areas are discussed regarding the success of the planted species and the influence of native and exotic species. Three Kentucky endangered species are present on the 1963 bench habitat that were not encountered in the unmined environs.

Area 1: Mixed Hardwoods. The four tree species planted in blocks are yellow poplar, American sycamore, northern red oak, and European black alder. Yellow poplar is volunteering and reproducing on the bench while exhibiting excellent growth. American sycamore has persisted with only fair to good success and is not volunteering on the site. Northern red oak has rather poor growth and appears stunted. European black alder, an actinomycete nitrogen-fixing tree, is spreading vegetatively and by seed although some older trees are senescent. Crown-vetch plantings have spread but are suppressed by the lush, dense growth of mesic woodland herbs. Two endangered species for Kentucky, ovate-leafed catchfly (*Silene ovata*) and Appalachian showy gentian (*Gentiana decora*), are growing in the mixed hardwoods plantation at the base of the 1970 outslope.

Area 2: Black Locust. The stand has good growth in height, 6-8 m, and is present in all size classes from 1-5 dm dbh on the bench (Table 2). Planted and volunteer trees have reproduced vegetatively and from seed. Black locust is especially important where bench soils have slumped onto the 1963 outslope.

Area 3: American sycamore. Sycamore ranges from 4-7 m in height and 1-2 dm dbh, but only had fair growth success. Area 3 appears to have droughty, shallow soil with a good stand of tall fescue, sericea lespedeza, Korean lespedeza, and white sweet clover.

Area 4: Northern red oak. Most red oak has rather poor growth with most trees less than 3 m tall and only a few trees 1-2 dm dbh.

Area 5: Scotch pine. This introduced pine has exhibited good growth with 9-11 m height and several trees in the 2-3 dm dbh size class. Although ovulate cones are produced, Scotch pine is not reproducing from seeds in cones (Table 2). Tall fescue dominates under trees, and several woodland herbs are present near the outslope. A wet seep depression near the highwall contains several wetland species. Creeping foxtail has locally become established and expanded its area from the Scotch pine plantation into the yellow poplar plantation (Thompson and Thieret 1986a).

Area 6: Yellow poplar. Some trees of yellow poplar have excellent growth with some 8-14 m tall and 3-4 dm dbh. It is also an important invading species. Several mesic woodland herbs are present along the outslope as well as wetland herbs from the highwall seep habitats, the yellow poplar plantation, and the Scotch pine plantation.

Area 7: Autumn olive. This actinomycete nitrogen-fixing species has migrated from its plantation and spread into adjacent Areas 3, 6, and 9. Autumn olive characteristically has some dieback, but is readily reproducing vegetatively and from seed.

Area 8: Bicolor lespedeza. Nitrogen-fixing bicolor lespedeza has migrated from subplots 3, 4, and 5 and spread into Areas 1 and 6. As it has spread, considerable root resprouting has occurred from old decadent stems.

Area 9: Norway spruce. This introduced spruce has growth comparable to the Scotch pine. Some taller trees are 12-13 m high and several are in the 2-3 dm dbh class. Although Norway spruce is producing ovulate cones, it is not reproducing from seeds.

1970 Site B Bench. The 1970 bench is rather open, dry, and grassy with numerous dense black locust and blackberry thickets near the outslope and highwall. A good ground cover has developed from the hydroseeded tall fescue, sericea lespedeza, Korean lespedeza, red clover, and black locust. The annual white and yellow sweet clovers have thinned out, but are still persisting. Black locust developed dense seedling thickets within six to eight years and has currently formed a partially closed canopy. The seven tree species interplanted in the 1984 plots have not had enough time to influence the bench vegetation. In one year, white ash, yellow poplar, and sugar maple are growing well; but, northern red oak, black walnut, and eastern white pine are barely persisting. Saw-tooth oak did not survive in any of the one-year-old tree plots.

Plant communities are composed mostly of volunteering, mid-successional, native woody plants and invading native and exotic herbaceous species. Successional trees invading the 1970 bench include red maple, white ash, yellow poplar, sugar maple, wild black cherry, and sourwood. Allegheny blackberry is the pre-eminent perennial forming dense thickets with glaucous greenbrier, northern dewberry, black raspberry, and multiflora rose (*Rosa multiflora*).

Several important herbs include tall fescue, broomsedge, oat poverty grass (*Danthonia compressa*), orchard grass (*Dactylis glomerata*), sericea lespedeza, cinquefoil (*Potentilla simplex*), wild strawberry (*Fragaria viginiana*), and sticktights. Hairy white aster, Canadian goldenrod, and slender goldenrod are the most important members of the Asteraceae. The site B bench has the second highest species richness with 160 taxa (Table 3).

Highwalls. Characteristic shade-intolerant woody invaders on the open highwall rim crest and in talus areas include Virginia pine, black locust, red maple, blackgum, sourwood, sassafras, and some shortleaf pine. Seral shrubs in open, grassy areas are poison ivy, winged sumac, smooth sumac, glaucous greenbrier, lowbush blueberry (*Vaccinium pallidum*), and St. Andrews-cross (*Hypericum stragulum*). Perennial herbs on the steep highwall slope and upper rim include oat poverty grass, broomsedge, hairy panicum, Allegheny blackberry, old-field goldenrod, tickseed sunflower (*Coreopsis major*), and Small's ragwort (*Senecio anonymus*). Steep highwall talus and crevices are colonized by patches of the naturalized common dandelion (*Taraxacum officinale*), coltsfoot (*Tussilago farfara*), and Japanese knotweed. Ninety-eight taxa were recorded from the highwalls (Table 3).

Area 10: Mixed pines. The only plantation that was not established on the 1963 site A bench was a block plantation of Virginia pine, shortleaf pine, loblolly pine, and eastern white pine. This mixed pine area was planted above the site A highwall adjacent to the disturbed mixed mesophytic hardwood forest. The forest soils are mostly unmined. The mean pH value is 4.3 and ranges from

Table 3. Annotated Vascular Plant List, Log Mountain Demonstration Area, Bell County, Kentucky.

Family	Species	Common Name	Habitats	1	2	3	4	5	6
POLYPODIOPHYTA (True Ferns)									
Aspleniaceae (Spleenwort Family)									
	Asplenium platyneuron (L.) Oakes.	Ebony Spleenwort.		R	R	-	-	-	-
	Polystichum acrostichoides (Michx.) Schott.	Christmas Fern.		I	-	R	-	-	-
	Thelypteris hexagonoptera (Michx.) Weath.	Broad Beech Fern.		R	-	-	-	-	-
	T. noveboracensis (L.) Nieuwl.	New York Fern.		R	I	-	-	-	-
Ophioglossaceae (Adder's Tongue Family)									
	Botrychium dissectum Spreng.	Cut-leaved Grape Fern.		R	-	-	-	-	-
	B. virginianum (L.) Sw.	Rattlesnake Fern.		-	-	-	R	-	-
Osmundaceae (Royal Fern Family)									
	Osmunda cinnamomea L.	Cinnamon Fern.		-	-	-	-	R	-
Polypodiaceae (Polypody Family)									
	Polypodium virginianum L.	Common Polypody.		-	-	-	R	-	-
EQUISETOPHYTA (Horsetails)									
Equisetaceae (Horsetail Family)									
	†*Equisetum arvense* L.	Common Horsetail.		-	-	-	-	I	-
LYCOPODIOPHYTA (Lycopods)									
Lycopodiaceae (Clubmoss Family)									
	†*Lycopodium digitatum* Dillen.	Southern Ground-cedar.		I	-	I	-	-	-
PINOPHYTA (Conifers)									
Cupressaceae (Cypress Family)									
	†*Juniperus virginiana* L.	Eastern Redcedar.		I	R	R	-	-	-
Pinaceae (Pine Family)									
	†+*Picea abies* (L.) Karst.	Norway Spruce.		O	-	-	-	-	-
	†o*Pinus echinata* P. Mill.	Shortleaf Pine.		-	I	I	-	-	-
	†o*P. strobus* L.	Eastern White Pine.		-	R	I	-	-	-
	†+**P. sylvestris* L.	Scotch Pine.		O	-	-	-	-	-
	†o*P. taeda* L.	Loblolly Pine.		-	-	R	-	-	-
	o*P. virginiana* L.	Virginia Pine		F	O	I	O	-	-
MAGNOLIOPHYTA (Flowering Plants)									
Aceraceae (Maple Family)									
	Acer negundo L.	Boxelder.		R	-	-	-	-	-
	A. rubrum L.	Red Maple.		A	A	A	O	-	-
	o*A. saccharum* Marsh.	Sugar Maple.		A	F	F	O	-	-
Anacardiaceae (Sumac Family)									
	†*Rhus copallina* L.	Winged Sumac.		R	O	I	I	-	-
	†*R. glabra* L.	Smooth Sumac.		-	I	I	-	-	-
	Toxicodendron radicans (L.) Kuntze.	Poison Ivy.		I	I	I	I	-	I
Apiaceae (Parsley Family)									
	Cryptotaenia canadensis (L.) DC.	Honewort.		I	-	-	-	-	-
	**Daucus carota* L.	Wild Carrot.		O	O	O	-	-	I
	Osmorhiza claytonii (Michx.) Clarke.	Hairy Sweet Cicely.		I	-	-	-	-	-
	†*Sanicula canadensis* L.	Black Snakeroot.		R	-	-	-	-	-
	†*Thaspium barbinode* (Michx.) Nutt.	Hairy Meadow Parsnip.		I	-	-	-	-	-
	Zizia aptera (Gray) Fern.	Heart-leaved Meadow Parsnip.		-	R	R	-	-	-
Apocynaceae (Dogbane Family)									
	Apocynum cannabinum L.	Indian Hemp.		O	O	I	-	R	-
Araceae (Arum Family)									
	Arisaema triphyllum (L.) Schott.	Jack-in-the-Pulpit.		R	-	-	-	-	-
Aristolochiaceae (Birthwort Family)									
	Asarum canadense L.	Wild Ginger.		R	-	R	-	-	-
Asclepiadaceae (Milkweed Family)									
	†*Asclepias exaltata* L.	Poke Milkweed.		-	R	-	-	-	R
	A. quadrifolia Jacq.	Whorled Milkweed.		R	-	-	-	-	-

†*A. syriaca* L. Common Milkweed.	I	I	I	-	-	-
Asteraceae (Aster Family)						
†**Achillea millefolium* L. Yarrow.	I	R	-	-	-	I
Ambrosia artemisiifolia L. Common Ragweed.	O	O	-	-	-	F
A. trifida L. Giant Ragweed.	O	O	-	-	-	O
Antennaria plantaginifolia (L.) Rich. Pussy-toes.	-	-	I	-	-	-
†**Arctium minus* Schk. Common Burdock.	-	-	-	-	-	R
†**Artemisia vulgaris* L. Mugwort.	-	-	-	-	-	R
Aster cordifolius L. Heart-leaved Aster.	R	-	-	-	-	-
A. divaricatus L. White Wood Aster.	R	-	-	-	-	-
A. dumosus L. Bushy Aster.	-	R	R	-	-	-
A. lowrieanus Porter Smooth Heart-leaved Aster.	-	-	R	-	-	-
A. lateriflorus (L.) Britt. Calico Aster.	-	-	I	-	-	I
†*A. phlogifolius* Muhl. Phlox-leafed Aster.	-	-	R	-	-	-
†*A. pilosus* Willd. White Heath Aster.	O	O	O	I	-	I
A. undulatus L. Wavy-leaved Aster.	-	O	I	-	-	-
Bidens polylepis S. F. Blake. Bur Marigold.	O	O	O	-	I	-
Cacalia atriplicifolia L. Pale Indian Plantain.	-	-	-	-	R	-
**Chrysanthemum leucanthemum* L. Ox-eye Daisy.	O	O	I	I	-	-
**Cichorium intybus* L. Chicory.	-	-	-	-	-	R
†**Cirsium arvense* (L.) Scop. Field Thistle.	-	O	-	-	-	O
†*C. discolor* (Muhl.) Spreng. Pasture Thistle.	-	I	-	-	-	O
†**C. vulgare* (Savi) Tenore. Bull Thistle.	-	-	-	-	-	R
†*Conyza canadensis* (L.) Cronq. Horseweed.	-	O	-	-	-	O
Coreopsis major Walt. Forest Tickseed.	-	-	O	I	-	-
†*Erechtites hieracifolia* (L.) Raf. Fireweed.	-	R	-	-	-	I
Erigeron annuus (L.) Pers. Annual Fleabane.	O	O	O	-	-	O
E. philadelphicus L. Philadelphia Fleabane.	I	-	-	I	-	-
E. strigosus Muhl. Daisy Fleabane.	I	O	I	-	-	O
Eupatorium fistulosum Barratt. Hollow Joe-pye Weed.	O	O	-	-	-	O
E. purpureum L. Purple Joe-pye Weed.	-	-	-	R	-	-
E. rugosum Houtt. White Snakeroot.	F	O	O	A	-	-
E. serotinum Michx. Late Boneset.	O	-	-	-	O	-
†*E. sessilifolium* L. Upland Boneset.	R	-	-	-	-	-
Gnaphalium obtusifolium L. Old-field Balsam.	I	I	-	-	-	-
Helianthus microcephalus T. & G. Small-headed Sunflower.	I	I	R	O	-	-
†*H. tuberosus* L. Jerusalem Artichoke.	R	-	-	-	-	I
Hieracium gronovii L. Hairy Hawkweed.	R	-	I	-	-	-
H. paniculatum L. Panicled Hawkweed.	-	-	I	-	-	-
H. scabrum Michx. Rough Hawkweed.	-	R	R	-	-	-
H. venosum L. Rattlesnake Hawkweed.	-	-	I	-	-	-
Krigia biflora (Walt.) S. F. Blake. False Dandelion.	R	-	-	I	-	R
†*Lactuca biennis* (Moench) Fern. Tall Blue Lettuce.	-	-	-	-	-	R
L. canadensis L. Canadian Wild Lettuce.	-	-	-	I	-	O
L. floridana (L.) Gaertn. Florida Blue Lettuce.	-	-	I	I	-	-
†**L. saligna* L. Willow Lettuce.	-	-	-	-	-	R
†**L. serriola* L. Prickly Lettuce.	-	R	-	-	-	I
†*Polymnia uvedalia* L. Bear's Foot.	-	-	-	I	-	-
Rudbeckia hirta L. Black-eyed Susan.	I	R	-	-	-	R
†*Senecio anonymus* Wood. Appalachian Groundsel.	O	O	O	O	-	-
†*Silphium trifoliatum* L. Whorled Rosin-weed.	R	-	-	R	-	-
Solidago arguta Ait. Cut-leaved Goldenrod.	O	O	O	I	-	-
S. bicolor L. Silver-rod.	R	-	I	-	-	-
S. caesia L. Blue-stemmed Goldenrod.	R	-	-	-	-	-
†*S. canadensis* L. Canada Goldenrod.	F	O	F	I	-	I
S. erecta Pursh. Slender Goldenrod.	-	R	-	-	-	-
†*S. flexicaulis* L. Broad-leaved Goldenrod.	O	I	-	O	-	-
†*S. gigantea* Ait. Late Goldenrod.	-	-	-	-	I	-
S. nemoralis Ait. Old-field Goldenrod.	O	O	O	O	-	-

S. rugosa Mill. Rough Goldenrod.	I	I	I	O	-	-
†*S. sphacelata* Raf. Short-pappus Goldenrod.	R	-	-	-	-	-
**Sonchus asper* (L.) Hill. Spiny Sow Thistle.	-	-	-	-	-	I
**Taraxacum officinale* Weber. Common Dandelion.	-	-	-	I	-	I
†**Tussilago farfara* L. Coltsfoot.	-	-	I	-	-	-
Verbesina alternifolia (L.) Britt. Wingstem.	O	R	-	O	O	O
Vernonia gigantea (Walter) Trel. Tall Ironweed.	O	I	O	-	-	-
Balsaminaceae (Touch-Me-Not-Family)						
Impatiens capensis Meerb. Spotted Jewelweed.	I	-	-	F	-	-
I. pallida Nutt. Pale Jewelweed.	F	R	F	F	-	-
Berberidaceae (Barberry Family)						
Caulophyllum thalictroides (L.) Michx. Blue Cohosh.	-	-	-	R	-	-
Podophyllum peltatum L. May Apple.	R	-	-	I	-	
Betulaceae (Birch Family)						
†+**Alnus glutinosa* (L.) Gaertn. European Black Alder.	F	-	-	-	-	-
Betula lenta L. Sweet Birch.	R	-	-	R	-	-
Brassicaceae (Mustard Family)						
**Barbarea vulgaris* R. Br. Yellow Rocket.	I	I	-	-	-	R
†*Cardamine concatenata* (Michx.) Schwartz. Slender Toothwort.	I	-	-	R	-	-
**C. hirsuta* L. Hairy Bittercress.	O	O	-	-	-	O
†**Lepidium campestre* (L.) R. Br. Field Cress.	-	-	-	-	-	R
†*L. virginicum* L. Common Peppercress.	-	-	-	-	-	I
Campanulaceae (Bellflower Family)						
†*Campanula americana* L. Tall Bellflower.	O	-	-	O	-	-
C. divaricata Michx. Appalachian Bellflower.	-	-	R	-	-	-
Lobelia inflata L. Indian Tobacco.	I	-	-	-	-	O
†*L. siphilitica* L. Great Blue Lobelia.	I	-	-	-	I	-
Caprifoliaceae (Honeysuckle Family)						
**Lonicera japonica* Thunb. Japanese Honeysuckle.	I	I	-	O	-	-
Sambucus canadensis L. American Elderberry.	O	I	I	O	I	-
Triosteum perfoliatum L. Late Horse Gentian.	-	R	-	-	-	-
Caryophyllaceae (Pink Family)						
**Cerastium vulgatum* L. Mouse-ear Chickweed.	R	O	-	-	-	O
**Dianthus armeria* L. Deptford Pink.	-	-	-	-	-	R
†*Paronychia* canadensis (L.) Wood. Tall Forked Chickweed.	-	-	-	R	-	-
Silene ovata Pursh. Ovate-leaved Catchfly.	R	-	-	-	-	-
†*S. virginica* L. Fire Pink.	-	-	R	-	-	-
†**Stellaria media* (L.) Vill. Common Chickweed.	R	O	-	-	-	O
S. pubera Michx. Great Chickweed.	-	-	-	R	-	-
Celasteraceae (Staff Tree Family)						
†**Euonymus fortunei* (Turcz.) Hand. & Maz. Winter Creeper.	-	-	-	R	-	-
Chenopodiaceae (Goosefoot Family)						
Chenopodium album L. Lamb's Quarters.	-	R	-	-	-	R
Clusiaceae (Mangosteen Family)						
†*Hypericum canadense* L. Canadian St. John's Wort.	-	-	-	-	O	-
H. stragulum P. Adams & Robson. St. Andrew's Cross.	I	-	I	-	-	-
†*H. mutilum* L. Weak St. John's Wort.	R	-	-	-	O	-
H. punctatum Lam. Spotted St. John's Wort.	O	I	-	-	-	-
Commelinaceae (Spiderwort Family)						
**Commelina communis* L. Asiatic Dayflower.	I	-	-	-	-	-
†*Tradescantia subaspera* Ker-Gawl. Broad-leafed Spiderwort.	-	-	I	R	-	-
Convolvulaceae (Morning Glory Family)						
Calystegia sepium (L.) R. Br. Hedge Bindweed.	-	-	-	-	-	I
Cornaceae (Dogwood Family)						
Cornus florida L. Flowering Dogwood.	O	O	I	O	-	-
Nyssa sylvatica Marsh. Black Gum.	O	O	F	F	-	-
Crassulaceae (Stonecrop Family)						
Sedum ternatum Michx. Wild Stonecrop.	-	-	-	R	-	-

Cuscutaceae (Dodder Family)						
†*Cuscuta pentagona* Engelm. Prairie Dodder.	I	-	-	-	I	-
Cyperaceae (Sedge Family)						
†*Carex atlantica* L.H. Bailey. Atlantic Star Sedge.	-	-	-	-	O	-
†*C. blanda* Dewey. Common Wood Sedge.	R	-	I	-	-	-
C. digitalis Willd. Narrow-leaved Wood Sedge.	R	-	I	-	-	-
C. frankii Kunth. Frank's Sedge.	R	-	-	-	O	-
C. laxiflora Lam. Beech Wood Sedge.	-	-	R	-	-	-
C. lurida Wahl. Yellow-green Sedge.	O	-	-	-	F	-
†*C. purpurifera* Mack. Purple-leaved Sedge.	-	-	-	R	-	-
C. virescens Muhl. Slender Green Sedge.	-	-	R	-	-	-
C. vulpinoidea Michx. Brown Fox Sedge.	O	-	-	-	F	-
Cyperus strigosus L. Long-scaled Nut Sedge.	-	-	-	-	R	-
Eleocharis ovata (Roth) R. & S. Oval Spike Rush.	-	-	-	-	O	-
Fimbristylis autumnalis (L.) R. & S. Autumn Sedge.	-	-	-	-	O	-
Rhynchospora capitellata (Michx.) Vahl. Brown Beakrush.	-	-	-	-	O	-
†*Scirpus atrovirens* Willd. Dark Green Bulrush.	-	-	-	-	I	-
S. cyperinus (L.) Kunth. Woolgrass.	-	-	-	-	F	-
Dioscoreaceae (Yam Family)						
**Dioscorea batatas* Decne. Cinnamon Vine.	O	O	-	O	-	-
D. villosa L. Wild Yam.	R	-	-	I	-	-
Elaeagnaceae (Oleaster Family)						
†+**Elaeagnus umbellata* Thunb. Autumn Olive.	F	-	-	-	-	-
Ericaceae (Heath Family)						
Chimaphila maculata (L.) Pursh. Spotted Wintergreen.	-	-	I	-	-	-
Oxydendrum arboreum (L.) DC. Sourwood.	F	O	O	O	-	-
Vaccinium pallidum Ait. Lowbush Blueberry.	R	-	I	-	-	-
Euphorbiaceae (Spurge family)						
†*Acalypha rhomboidea* Raf. Three-seeded Mercury.	-	I	-	-	-	I
Euphorbia corollata L. Flowering Spurge.	-	R	-	-	-	-
E. maculata L. Spotted Spurge.	-	I	-	-	-	I
E. nutans Lagasca. Eyebane.	-	-	-	-	-	I
Fabaceae (Pea Family)						
Amphicarpaea bracteata (L.) Fern. Hog Peanut.	I	-	-	I	-	-
Cercis canadensis L. Eastern Redbud.	I	I	-	R	-	-
Chamaecrista nictitans (L.) Moench. Wild Sensitive Plant.	-	-	-	-	-	I
†+**Coronilla varia* L. Crown-vetch.	O	-	-	-	-	-
†*Desmodium glabellum* (Michx.) DC. Smooth Tick Trefoil.	I	O	-	-	-	-
D. glutinosum (Muhl.) Wood. Pointed Tick Trefoil.	-	-	-	R	-	-
D. paniculatum (L.) DC. Panicled Tick Trefoil.	O	F	-	-	-	-
†+**Lespedeza bicolor* Turcz. Bicolor Lespedeza.	F	-	-	-	-	-
†+**L. cuneata* (Dum.-Cours.) G. Don. Sericea Lespedeza.	F	A	-	I	-	I
†+**L. stipulacea* Maxim. Korean Lespedeza.	F	A	-	-	-	-
†**Medicago lupulina* L. Black Medick.	-	R	-	-	-	I
†+**Melilotus alba* Medic. White Sweet Clover.	I	O	-	-	-	-
†+**M. officinalis* (L.) Pallas. Yellow Sweet Clover.	I	I	-	-	-	-
†o*Robinia pseudoacacia* L. Black Locust.	A	A	O	A	-	-
**Trifolium campestre* Schreb. Low Hop Clover.	R	O	-	-	-	R
**T. hybridum* L. Alsike Clover.	R	I	-	-	-	-
+**T. pratense* L. Red Clover.	O	O	-	-	-	-
†**T. repens* L. White Clover.	O	O	-	-	-	R
Vicia caroliniana Walt. Carolina Wood-vetch.	-	R	-	-	-	-
†**V. sativa* L. Common Vetch.	R	I	-	-	-	-
Fagaceae (Beech Family)						
Fagus grandifolia Ehrh. American Beech.	R	-	-	-	-	-
Quercus alba L. White Oak.	R	R	-	-	-	-
Q. prinus L. Chestnut Oak.	R	-	-	-	-	-
o*Q. rubra* L. Northern Red Oak.	O	R	-	-	-	-
Q. velutina Lam. Black Oak.	R	-	-	-	-	-

Gentianaceae (Gentian Family)						
Gentiana decora Pollard. Appalachian Gentian.	R	-	-	-	-	-
Geraniaceae (Geranium Family)						
Geranium maculatum L. Wild Geranium.	I	-	-	R	-	-
Hamamelidaceae (Witch Hazel Family)						
Liquidambar styraciflua L. Sweetgum.	R	-	-	-	R	-
Hippocastanaceae (Buckeye Family)						
Aesculus flava Ait. Yellow Buckeye.	R	-	-	-	-	-
Hydrangeaceae (Hydrangea Family)						
Hydangea arborescens L. Wild Hydrangea.	O	I	I	O	-	-
Iridaceae (Iris Family)						
†*Sisyrinchium angustifolium* P. Mill. Narrow Blue-eyed Grass.	I	-	R	-	-	-
Juglandaceae (Walnut Family)						
†*Carya cordiformis* (Wang.) K. Koch. Bitternut Hickory.	R	-	-	-	-	-
C. ovalis (Wang.) Sarg. Sweet Pignut Hickory.	R	-	-	-	-	-
†o*Juglans nigra* L. Black Walnut.	R	R	-	-	-	-
Juncaceae (Rush Family)						
Juncus acuminatus Michx. Sharp-fruited Rush.	-	-	-	-	O	-
J. effusus L. var. *solutus* Fern. & Wieg. Soft Rush.	-	-	-	-	F	-
J. marginatus Rostk. Grass-leaved Rush.	-	-	-	-	O	-
J. tenuis Willd. Path Rush.	R	-	-	-	I	I
†*Luzula multiflora* (Retz.) Lejeune. Common Wood Rush.	R	-	-	-	-	-
Lamiaceae (Mint Family)						
Collinsonia canadensis L. Richweed.	I	-	-	-	-	-
†*Hedeoma pulegioides* (L.) Pers. American Pennyroyal.	-	-	R	-	-	-
**Lamium purpureum* L. Purple Dead Nettle.	-	-	-	-	-	R
Lycopus virginicus L. Bugleweed.	-	-	-	-	O	-
Monarda clinopodia L. Basil Bee Balm.	-	-	-	R	-	R
†*M. fistulosa* L. Wild Bergamot.	R	-	-	I	-	-
†**Prunella vulgaris* L. Selfheal.	O	O	-	-	-	R
†*Pycnanthemum incanum* (L.) Michx. Hoary Mountain Mint.	-	R	I	-	-	I
Salvia lyrata L. Lyre-leaved Sage.	I	-	-	-	I	-
Scutellaria elliptica Muhl. Hairy Skullcap.	R	-	-	R	-	-
†*Stachys nuttallii* Shuttlw. ex Benth. Nuttall's Hedge Nettle.	I	-	-	I	-	-
Lauraceae (Laurel Family)						
Sassafras albidum (Nutt.) Nees. White Sassafras.	O	O	O	O	-	-
Liliaceae (Lily Family)						
†**Allium vineale* L. Field Garlic.	I	R	-	-	-	-
†*Erythronium americanum* Ker-Gawl. Yellow Trout Lily.	I	-	-	-	-	-
†**Hemerocallis fulva* L. Orange Day-lily.	-	-	-	R	-	-
†**Narcissus pseudonarcissus* L. Daffodil.	-	-	-	R	-	-
†*Polygonatum biflorum* (Walt.) Ell. True Solomon's Seal.	R	-	-	I	-	-
Smilacina racemosa (L.) Desf. False Solomon's Seal.	I	-	-	I	-	-
Trillium grandiflorum (Michx.) Salisb. Large White Trillium.	R	-	-	-	-	-
Uvularia grandiflora Smith. Bellwort.	R	-	-	-	-	-
Linaceae (Flax Family)						
†*Linum striatum* Walt. Stiff Yellow Flax.	-	R	-	-	-	-
L. virginianum L. Slender Yellow Flax.	-	R	-	-	R	-
Magnoliaceae (Magnolia Family)						
o*Liriodendron tulipifera* L. Yellow Poplar.	F	F	O	O	-	-
†*Magnolia acuminata* L. Cucumber-tree.	I	-	-	I	-	-
Menispermaceae (Moonseed Family)						
†*Menispermum canadense* L. Canadian Moonseed.	R	-	-	-	-	-
Oleaceae (Olive Family)						
o*Fraxinus americana* L. White Ash.	A	F	O	A	-	-
Onagraceae (Evening Primrose Family)						
Circaea lutetiana L. var. *canadensis* L. Enchanter's Nightshade.	I	-	-	-	-	-
†*Epilobium coloratum* Biechler. Cinnamon Willow Herb.	-	-	-	-	I	-
†*Ludwigia alternifolia* L. Seedbox.	-	-	-	-	I	-

Oenothera biennis L. Evening Primrose.	-	R	-	-	-	R
Orchidaceae (Orchid Family)						
Aplectrum hyemale (Muhl.) Torr. Cranefly Orchid.	R	-	-	-	-	-
Goodyera pubescens (Willd.) R. Br. Rattlesnake Plantain.	R	-	-	-	-	-
Habenaria lacera (Michx.) Lodd. Ragged Fringed Orchid.	R	-	-	-	-	-
†*Liparis loeselii* (L.) L. Rich. Green Twayblade.	-	-	-	-	O	-
Spiranthes lacera (Raf.) Raf. Slender Ladies' Tresses.	-	-	R	-	-	-
Oxalidaceae (Wood Sorrel Family)						
**Oxalis corniculata* L. Creeping Wood Sorrel.	-	-	-	-	-	R
†*O. grandis* Small. Large Yellow Wood Sorrel.	I	-	-	-	-	-
O. stricta L. Common Wood Sorrel.	-	-	-	-	-	I
Paeoniaceae (Peony Family)						
†**Paeonia officinalis* L. Common Peony.	-	-	-	R	-	-
Papaveraceae (Poppy Family)						
Sanguinaria canadensis L. Bloodroot.	R	-	-	-	-	-
Phytolaccaceae (Poke Family)						
†*Phytolacca americana* L. American Pokeweed.	-	O	I	O	-	I
Plantaginaceae (Plantain Family)						
**Plantago lanceolata* L. English Plantain.	I	O	-	-	-	O
P. rugelii Decne. Rugel's Plantain.	I	I	-	-	-	O
Platanaceae (Plane Tree Family)						
†o*Platanus occidentalis* L. American Sycamore.	F	O	I	O	O	-
Poaceae (Grass Family)						
**Agrostis gigantea* Roth. Redtop.	O	I	-	-	O	-
A. perennans (Walt.) Tuckerm. Autumn Bentgrass.	-	-	O	-	-	R
†+**Alopecurus arundinaceus* Poir. Creeping Foxtail.	O	-	-	-	I	-
Andropogon virginicus L. Broomsedge.	F	F	O	-	-	I
Brachyelytrum erectum (Schreb.) Beauv. Awned Woodgrass.	-	-	-	R	-	-
†**Bromus japonica* Thunb. Japanese Chess.	I	O	-	-	-	I
B. pubescens Muhl. Woodland Brome.	-	-	-	R	-	-
†**Dactylis glomerata* L. Orchard Grass.	O	O	-	-	-	R
†*Danthonia compressa* Aust. Mountain Oatgrass.	O	I	F	R	-	-
†**Digitaria ischaemum* Schreb. Smooth Crabgrass.	I	I	-	-	-	O
†**Echinochloa crusgalli* (L.) Beauv. Barnyard Grass.	I	-	-	-	O	-
†*Elymus hystrix* L. Bottlebrush Grass.	R	-	-	-	-	-
†*E. virginicus* L. Virginia Wild Rye.	I	-	-	-	-	-
†**Elytrigia repens* (L.) Nevski. Quackgrass.	-	R	-	-	-	R
†*Eragrostis capillaris* (L.) Nees. Lacegrass.	-	-	-	-	-	I
†*E. frankii* C. A. Mey. Sandbar Lovegrass.	-	-	-	-	-	R
†+**Festuca elatior* L. Tall Fescue.	A	A	I	O	-	I
Leersia virginica Willd. White Grass.	-	-	-	-	O	-
**Microstegium vimineum* (Trin.) Camas. Eulalia.	A	A	O	A	-	I
†*Muhlenbergia frondosa* (Poir.) Fern. Satin Grass.	-	-	-	-	I	-
†*M. schreberi* J. F. Gmel. Nimblewill.	-	R	-	-	-	-
†*Panicum boscii* Poir. Wide-leafed Panic Grass.	O	O	-	I	-	-
†*P. capillare* L. Old Witch Grass.	-	R	-	-	-	R
†*P. clandestinum* L. Deer-tongue Panic Grass.	F	O	-	O	O	-
†*P. commutatum* Schultes. Ashe's Panic Grass.	O	I	-	O	-	-
P. dichotomiflorum Michx. Spreading Witch Grass.	-	-	-	-	I	I
P. dichotomum L. Forked Panic Grass.	-	-	-	-	F	-
†*P. lanuginosum* Ell. Hairy Panic Grass.	F	O	I	F	-	-
P. polyanthes Schultes. Small-fruited Panic Grass.	O	O	-	O	-	-
**Phleum pratense* L. Timothy.	I	O	-	-	-	-
†*Poa alsodes* A. Gray. Grove Bluegrass.	-	-	-	R	-	-
†**P. compressa* L. Canada Bluegrass.	I	I	-	-	-	I
†*P. cuspidata* Nutt. Short-leaved Bluegrass.	-	-	-	R	-	-
**P. pratensis* L. Kentucky Bluegrass.	O	O	-	-	-	I
†**Setaria faberii* Herrm. Nodding Foxtail.	O	-	I	-	-	I
**S. glauca* (L.) Beauv. Yellow Foxtail.	R	-	-	-	-	R

†*Sphenopholis obtusata* (Michx.) Scribn. Prairie Wedgegrass.	I	I	R	-	-	-
†*Sporobolus vaginiflorus* (Torr.) Wood. Dropseed.	-	-	R	-	-	O
†**Triticum aestivum* L. Wheat.	-	R	-	-	-	-
Polemoniaceae (Phlox Family)						
Phlox amplifolia Britt. Wide-leaved Phlox.	I	-	-	-	-	-
Polygalaceae (Milkwort Family)						
†*Polygala senega* L. Seneca Snakeroot.	I	I	-	-	-	-
Polygonaceae (Smartweed Family)						
**Polygonum aviculare* L. Common Knotweed.	-	-	-	-	-	R
+**Polygonum cuspidatum* Sieb. & Zucc. Japanese Knotweed.	O	O	O	O	-	-
P. pensylvanicum L. Pennsylvania Pinkweed.	-	I	-	-	-	I
**P. persicaria* L. Lady's Thumb.	-	-	-	-	-	I
†*P. punctatum* Ell. Dotted Smartweed.	-	-	-	-	R	I
P. sagittatum L. Arrow-leafed Tearthumb.	-	-	-	-	I	-
P. scandens L. Climbing False Buckwheat.	F	I	-	F	-	O
P. virginianum L. Virginia Knotweed.	R	-	-	-	-	-
†**Rumex crispus* L. Curly Dock.	I	I	-	-	R	I
**R. obtusifolius* L. Bitter Dock.	I	I	-	-	-	I
Portulacaceae (Purslane Family)						
Claytonia virginica L. Spring Beauty.	I	-	-	-	-	-
Primulaceae (Primrose Family)						
†*Lysimachia quadrifolia* L. Whorled Loosestrife.	-	-	I	-	-	I
L. tonsa (Wood) Kunth. Appalachian Loosestrife.	-	I	I	-	-	I
Ranunculaceae (Buttercup Family)						
†*Anemone virginiana* L. Thimbleweed.	I	O	-	I	-	-
†*Anemonella thalictroides* (L.) Spach. Rue Anemone.	I	I	-	-	-	-
†*Clematis glaucophylla* Small. Glaucous-leaved Clematis.	-	-	-	-	-	I
C. virginiana L. Virgin's Bower.	A	A	-	A	-	I
†*Delphinium tricorne* Michx. Dwarf Larkspur.	R	-	-	-	-	-
Ranunculus abortivus L. Small-flowered Buttercup.	O	O	-	-	-	I
†*R. hispidus* Michx. Hispid Buttercup.	I	I	-	O	-	-
R. recurvatus Poir. Hooked Crowfoot.	I	I	-	-	-	-
Thalictrum dioicum L. Early Meadow-rue.	I	I	-	O	-	-
Rosaceae (Rose Family)						
†*Agrimonia rostellata* Wallr. Beaked Agrimony.	O	I	-	O	-	I
Fragaria virginiana Duchesne. Virginia Strawberry.	F	F	O	-	-	I
Geum canadense Jacq. White Avens.	O	-	-	O	-	-
Potentilla canadensis L. Canada Cinquefoil.	I	O	O	-	-	I
†**P. norvegica* L. Rough Cinquefoil.	-	I	-	-	-	I
**P. recta* (L.) Raeusch. Upright Cinquefoil.	I	-	-	-	-	-
†*P. simplex* Michx. Common Cinquefoil.	O	O	I	-	-	I
Prunus serotina Ehrh. Wild Black Cherry.	I	I	-	-	-	-
†**Rosa multiflora* Thunb. Multiflora Rose.	I	I	-	I	-	-
Rubus allegheniensus Porter. Allegheny Blackberry.	A	A	I	F	O	I
†*R. flagellaris* Willd. Common Dewberry.	O	F	I	O	-	I
†*R. occidentalis* L. Black Raspberry.	O	O	I	O	-	-
†*Spiraea tomentosa* L. Pink Steeplebush.	-	-	-	-	O	-
Rubiaceae (Madder Family)						
Galium aparine L. Annual Cleavers.	F	F	-	A	-	I
G. latifolium Michx. Wide-leafed Bedstraw.	-	-	-	I	-	-
†*G. triflorum* Michx. Sweet-scented Bedstraw.	O	-	-	O	-	-
Hedyotis caerulea (L.) Hook. Spring Bluets.	O	O	O	-	-	R
H. purpurea (L.) T. & G. Large Summer Bluets.	I	I	-	-	-	-
Salicaceae (Willow Family)						
†*Salix nigra* Marsh. Black Willow.	F	-	-	-	F	-
†*S. sericea* Marsh. Silky Willow.	F	-	-	-	F	-
Saxifragaceae (Saxifrage Family)						
Heuchera americana L. Common Alum-root.	-	-	R	-	-	-

Scrophulariaceae (Figwort Family)						
Aureolaria laevigata (Raf.) Raf. Smooth False Foxglove.	-	-	-	R	-	-
†*Mimulus ringens* L. Common Monkey-flower.	-	-	-	-	I	-
Pedicularis canadensis L. Wood Betony.	R	-	-	-	-	-
†*Penstemon brevisepalis* Pennell. Appalachian Beard-tongue.	-	R	I	-	-	R
**Verbascus thapsus* L. Common Mullein.	-	-	-	-	-	R
†**Veronica arvensis* L. Corn Speedwell.	-	O	-	-	-	I
†**V. hederaefolia* L. Ivy-leafed Speedwell.	R	R	-	-	-	-
**V. officinalis* L. Common Speedwell.	I	I	-	-	-	-
Simaroubaceae (Quassia Family)						
**Ailanthus altissima* (Mill.) Swingle. Tree-of-heaven.	-	-	R	-	-	-
Smilacaceae (Greenbrier Family)						
†*S. glauca* Walt. Glaucous Greenbrier.	F	F	I	O	-	I
S. herbacea L. Herbaceous Carrion Flower.	R	-	-	-	-	-
†*S. rotundifolia* L. Common Greenbrier.	F	F	O	I	-	-
Solanaceae (Nightshade Family)						
†*Datura stramonium* L. Jimson Weed.	-	-	-	-	-	R
†*Solanum americanum* Mill. Black Nightshade.	-	-	-	-	-	R
S. carolinense L. Horse Nettle.	-	I	-	-	-	I
Tiliaceae (Linden Family)						
Tilia americana L. American Basswood.	I	-	-	I	-	-
Typhaceae (Cattail Family)						
†*Typha latifolia* L. Common Cattail.	-	-	-	-	A	-
Ulmaceae (Elm Family)						
†*Ulmus rubra* Muhl. Red Elm.	I	R	-	I	-	-
Urticaceae (Nettle Family)						
†*Laportea canadensis* (L.) Wedd. True Nettle.	O	-	-	F	-	-
Pilea pumila (L.) A. Gray. Clearweed.	O	O	-	O	-	-
Verbenaceae (Vervain Family)						
Verbena urticifolia L. Hairy White Vervain.	I	-	-	-	R	-
Violaceae (Violet Family)						
†*Hybanthus concolor* (Forster) Spreng. Green Violet.	R	-	-	-	-	-
†*Viola canadensis* L. Canada Violet.	O	I	-	F	-	-
†*V. eriocarpa* Schwein. Smooth Yellow Violet.	I	I	-	I	-	-
†*V. hirsutula* Brainerd. Southern Woolly Violet.	-	R	I	-	-	-
†*V. rostrata* Pursh. Long-spurred Violet.	I	-	-	-	-	-
V. rotundifolia Michx. Round-leafed Yellow Violet.	R	-	-	-	-	-
†*V. sororia* Willd. Common Blue Violet.	F	O	R	O	-	-
†*V. palmata* L. Lobed Violet.	-	I	I	I	-	-
Vitaceae (Grape Family)						
Parthenocissus quinquefolia (L.) Planch. Virginia Creeper.	I	R	I	O	-	-
Vitis aestivalis Michx. Summer Grape.	I	R	-	O	-	-
Habitat Total Species:	223	160	98	109	53	104

Code: (†) Bell County records; (+) Introduced planted taxa; (o) Native planted taxa; (*) Non-indigenous taxa.
Relative abundance values: R=rare; I=infrequent; O=occasional; F=frequent; A=abundant.
Habitats: 1=1963 bench; 2=1970 bench; 3=highwalls; 4=outslopes; 5=seeps, 6=access road.
[Note: Methods section].

4.0-4.9 which is the overall lowest pH sampled (Table 1). Virginia pine exhibits vigorous reproduction and growth. It has spread by seed to the outslopes, highwalls, the 1963 bench (Table 2), the 1970 bench and into the mixed hardwood forest. Nearly all the loblolly pine and eastern white pine and most of the shortleaf pine were destroyed by the mining of the 1970 area. Only three loblolly pines and 10 eastern white pine have persisted on this highwall plantation. Shortleaf pine has some seedling and sapling reproduction evident on the highwall and on the 1970 bench. Shrubs and herbs in this dry exposed area are representative of the species of the general highwall habitat.

Outslopes. The 1964 outslope plantings of tall fescue and Korean lespedeza have exhibited fair to good success. Black locust was heavily seeded in 1965 on the steep 1963 outslope after slumping of the bench occurred in 1964. Bench plantings of European black alder, autumn olive, and bicolor lespedeza have spread onto the outslope as have yellow poplar, white ash, sugar maple, and red maple. Shrubs and vines include wild hydrangea, poison ivy, common elderberry, virgin's bower, multiflora rose, and Japanese honeysuckle. The herbaceous layer is dominated by pioneer planted and volunteer species in the Poaceae, Asteraceae, Fabaceae, and Rosaceae. A total of 109 taxa were recorded from the outslopes (Table 3).

The black locust stand on the 1970 site B outslope at LMDA was one of several sites in the black locust provenance studies by Ashby et al. (1985). Black locust was initially hydroseeded in 1971 on the 1970 outslope and bench. By 1977, black locust seedlings had grown through the tall fescue and sericea lespedeza to form dense seedling thickets. This black locust stand had a density of 716 trees per hectare with a third of the trees dead in 1982. Most of the 482 live black locust trees per hectare were less than 1.0 dm dbh. The shade-tolerant tall fescue dominated about one-third of the open areas. A rich cover of white snakeroot, orange jewelweed (*Impatiens capensis*), pokeweed, eulalia, and other woodland herbs covered the remainder of the area (Ashby et al. 1985).

Sugar maple, red maple, white ash, and other more shade-tolerant volunteers, were replacing black locust. However, invasion was meager partly because of the vigorous herbaceous cover and scarcity of seed sources (Ashby et al. 1985). Other invading trees documented in the stand were yellow poplar, wild black cherry, sourwood, blackgum, sassafras, autumn olive, red elm, and Virginia pine. Vines and shrubs included virgin's bower, American elderberry, wild hydrangea, summer grape, Virginia creeper, poison ivy, and common greenbrier (Ashby and Vogel, USDA, unpublished data 1992).

LMDA Seeps. Wet seep pools are found on the 1963 site A bench in Scotch pine plantations of Area 5, yellow poplar plantations of Area 6, and on the 1970 site B bench at the highwall base. These wet depressions retain water throughout most of the growing season and support wetland vegetation. We collected 53 species from the seeps (Table 3). The woody plant zone is dominated by black willow (*Salix nigra*), silky willow (*S. sericea*), and pink steeplebush (*Spiraea tomentosa*). At the border of the woody zone is a common cattail (*Typha latifolia*)-woolgrass (*Scirpus cyperinus*) emergent zone. Other emergent species are soft rush (*Juncus effusus*), blunt spikerush (*Eleocharis ovata*), and blue-green bulrush (*Scirpus atrovirens*).

Wet meadow species adjacent to the tree and emergent zones comprise the sedge (*Carex*)-rush (*Juncus*) zone. Wetland indicator herbs include yellow-green sedge (*Carex lurida*), Frank's sedge (*C. frankii*), fox sedge (*C. vulpinoidea*), beakrush (*Rhynchospora capitellata*), sharp-pointed rush (*Juncus acuminatus*), seedbox (*Ludwigia alternifolia*), great blue lobelia (*Lobelia siphilitica*), marsh St. John's-wort (*Hypericum mutilum*), bugleweed (*Lycopus virginicus*), deer-tongue panicum, and common horsetail (*Equisetum arvense*).

Two unique plants for Kentucky are present in the sedge-rush zone. Creeping foxtail (*Alopecurus arundinaceus*), recorded only from North Dakota and Newfoundland, is becoming naturalized at LMDA (Thompson and Thieret 1986a). The rare green twayblade (*Liparis loeselii*), is a new distribution record which was previously known from only a single county in Kentucky (Thompson and MacGregor 1986b).

Coal Haul-Telephone Microwave Tower Road. This highly compacted access road is maintained in an early, pioneer seral stages from the continual disturbance by vehicles. Indicator herbs are mainly composed of many weedy native and exotic annuals, biennials, and perennials of the Asteraceae, Poaceae, Fabaceae, Rosaceae, and Polygonaceae. A few herbs found are common ragweed (*Ambrosia artemisiifolia*), daisy fleabane (*Erigeron annuus*), horseweed (*Conyza canadensis*), Korean lespedeza, sericea lespedeza, wild carrot, English plantain (*Plantago lanceolata*), milk spurge (*Euphorbia nutans*), smooth crabgrass (*Digitaria ischaemum*), nodding foxtail (*Setaria faberii*), and tall fescue. This ruderal habitat had 104 taxa recorded (Table 3).

LMDA Flora Summary

The annotated list is composed of 360 specific and infraspecific taxa in 224 genera from 82 families. A total of 74 species (20.6%) were non-indigenous introduced or naturalized (Table 3). Vascular plants consist of 1 Lycopodiophyta, 1 Equisetophyta, 8 Polypodiophyta, 7 Pinophyta, and 343 Magnoliophyta (82 Liliopsida, 261 Magnoliopsida). The largest plant families numerically are the Asteraceae (64), Poaceae (39), Fabaceae (20), Cyperaceae (16), Rosaceae (13), Lamiaceae (11), and Polygonaceae (10). Three threatened Kentucky species, Appalachian showy gentian, green twayblade, and ovate-leaved catchfly, are found in habitats created by surface mining. A total of 155 (43.1%) were Bell County distribution records (Table 3).

Discussion

Planted Species Evaluation

Twenty-four of 25 reclamation species (96.0%) originally planted during 1964, 1965, 1971, and 1984, are persisting at the LMDA through 1985. Several legumes are contributing to soil enrichment and community development. Several hardwoods and conifers are important to plant community structure and for potential forestry products. Woody and herbaceous exotic species are becoming locally naturalized and important in revegetation and plant succession on LMDA.

We recommend eleven successful planted species at LMDA for other contour surface-mined areas or similarly disturbed areas in the Cumberland Mountains. The plantings of tall fescue, sericea lespedeza, and Korean lespedeza, have been important for initial plant cover and site development throughout the LMDA habitats. Crown-vetch, red clover, white sweet clover, and yellow sweet clover, have enhanced overall cover, but to a lesser degree than tall fescue, sericea lespedeza, and Korean lespedeza. European black alder, autumn olive, and bicolor lespedeza are important soil enrichment species spreading on outslopes and benches. Black locust has significantly influenced the development of the young hardwood forest. This important early pioneering species contributes to soil nitrogen-fixation processes, litter production from dead trees, and serves as a nurse tree for other species.

Sugar maple, yellow poplar, and white ash are important volunteers in secondary succession throughout the LMDA habitats. Virginia pine is an important invader in other mine habitats. Shortleaf pine has been less successful in survival and migration. Eastern white pine and loblolly pine have not survived well partly because of destruction during remining in 1970. Scotch pine and Norway spruce on the 1963 bench have potential for forest products although these introduced trees are not reproducing from their own seed sources.

We do not highly recommend some woody plantings that have persisted at LMDA until more experimental planting research has been done or other potential choices have been tried. American sycamore and northern red oak have not grown as well as several other planted hardwood trees on the 1963 bench. American sycamore, northern red oak, and black walnlut are not growing well on the 1984 block plots on the 1970 bench, although it is too early for final evaluation. The only planted species not recommended for planting is saw-tooth oak which did not survive at LMDA.

LMDA as a Refugium for Rare Species

The Log Mountain Demonstration Area has provided new habitats for at least three Kentucky "endangered" plant species. The second Kentucky documentation of the green twayblade (*Liparis loeselii*) was recorded in the sedge-rush 1963 bench seeps (Thompson and MacGregor 1986b). The ovate-leaved catchfly (*Silene ovata*) and the Appalachian showy gentian (*Gentiana decora*), were found on the 1963 bench (Table 3).

Wade and Thompson (1993) reported that five pre-SMCRA surface-mined sites in Kentucky have provided important habitats for certain rare plant species not found in the undisturbed or non-mined environs. All five revegetated pre-SMCRA coal-mined sites in the Cumberland Mountains and Cumberland Plateau have provided habitats for one or more Kentucky endangered or threatened species listed by Warren et al. (1986). None of these rare Kentucky species were found in the adjacent non-mined environs of these coal mined sites during the initial or subsequent reconnaissances. While presence of these rare species outside of the immediate environs is unreported or unknown, the establishment of rare species has occurred in habitats created on the pre-SMCRA coal mined areas in the Cumberland Mountains and Cumberland Plateau. The potential status of these rare species is not predictable because of continual vegetation development, potential environmental changes, or anthropogenical effects in the future (Thompson and Wade 1991).

We classify these restricted habitats created by pre-law surface mining in Kentucky as "refugia" where rare species have become established. These unique habitats have served as suitable refuges for certain rare plants that might otherwise have become more rare or even extirpated from the region. In Europe, abandoned rock quarries (Usher 1979, Wartner 1983), and surface-mined lands (Bruns 1986, Dahl and Juerging 1982, Sanderson 1992) have served as refugia for many rare species not found elsewhere.

LMDA Floristic Richness

Species richness is a measure of success in the reclamation process in conjunction with natural plant succession. At the LMDA, the 360 species are comparable to two other pre-law coal mined sites in the Cumberland Plateau of Kentucky. A total of 350 species were found on an 18-year-old area-type surface mine of 14.0 ha in Laurel County (Thompson et al. 1984); 272 species were present on a 12-year-old contour surface-mined area of 2.5 ha in

Rockcastle County (Thompson and Wade 1991). The Asteraceae, Poaceae, Fabaceae, Cyperaceae, and Rosaceae, were also the largest families on the other four pre-SMCRA mines sites in eastern Kentucky (Wade and Thompson 1993). The 360 species at LMDA represent 11.5% of the total flora for Kentucky based on Browne and Athey (1992).

A species-area curve was derived to compare species that were "found" with those "expected" in different-sized areas within the Western and Mixed Mesophytic Forest Region (Wade and Thompson 1991, 1993). We found that the actual species richness at the LMDA is only two percent less than species richness expected. At four other pre-SMCRA Kentucky surface-mined sites, species richness ranged from only three to 12 percent less than predicted. The 155 distribution records indicate the sparseness of plant collections and botanical exploration in Bell County.

LMDA Vegetation and Secondary Succession

The contour-mining and reclamation efforts have created six physical habitats with minesoils significantly different from the non-mined contiguous ridgetops of the disturbed mixed mesophytic hardwood forest. While contour-mining disturbance at LMDA removed most of the original soils with the geologic substrate, seed bank diaspores or propagules were intermixed with surface minespoils. Planted nitrogen-fixing trees and herbs helped ameliorate the rapidly weathering minesoils. Planted exotics and native invaders have influenced the species composition during the early secondary successional stages. Shade-intolerant, pine and hardwood seedlings have become established from wind-transported seeds and fruits.

The disturbed mixed mesophytic hardwood forest above the highwall is important as diaspore or propagule source for overall high species richness and natural successional trends at the LMDA. This forest has a well-developed canopy with seedlings and saplings evident, a sparse subcanopy and shrub layers, and a rich herbaceous layer. Important trees are sugar maple, red maple, yellow poplar, American basswood (*Tilia americana*), yellow buckeye (*Aesculus flava*), white ash, sourwood, red pignut hickory (*Carya ovalis*), white oak (*Quercus alba*), chestnut oak (*Q. prinus*), and northern red oak. Flowering dogwood (*Cornus florida*) and eastern redbud are two important subcanopy trees. Shrubs and vines are common greenbrier, glaucous greenbrier, Virginia creeper, summer grape, and wild hydrangea.

The forest vegetation at LMDA is a complex mosaic of natural and semi-natural plant communities in various of secondary seral stages in the LMDA habitats. Plant communities are the result of habitat diversity created by surface-mining, reclamation, and minesoil characteristics. High species richness is a function of volunteering species from the environs, planted reclamation species, and the remnant seed bank of the pre-mining habitats at LMDA. Species richness, vegetation development, and habitat diversity at LMDA are comparable to other pre-SMCRA surface-mined area in the Cumberland Plateau of Kentucky (Thompson et al. 1984, 1986c, Thompson and Wade 1991, Wade and Thompson 1993).

Conclusions

Several interrelated factors have been ascertained from descriptive research data of the pre-SCMRA Log Mountain Demonstration Area:

1. Habitat diversity created by pre-law coal surface mining supports a high species richness comparable to the contiguous non-mined environs.
2. Habitats from pre-SMCRA mining serve as refugia for certain rare plants that are not present in the unmined contiguous habitats or environs.
3. The entire flora is derived from the remnant seed bank, native and exotic invaders from the environs, and the planted native and exotic reclamation species.
4. Plant communities are developing mainly through seral stages of secondary succession with the strong influence of the natural invading and planted native and exotic flora.
5. Vegetation is a complex mosaic of natural and semi-natural plant communities progressing toward a young mixed hardwood forest on the planted and unplanted surface-mined habitats.
6. Twenty-four of 25 native and exotic species have persisted from plantings by personnel of the USDA Forest Service.
7. Nitrogen-fixing species important for soil and site enrichment are black locust, European black alder, autumn olive, bicolor lespedeza, sericea lespedeza, Korean lespedeza, and red clover.
8. Native trees recommended as reclamation plantings for vegetational development are black locust, white ash, yellow poplar, sugar maple, and Virginia pine.
9. Naturalized herbaceous taxa recommended for reclamation ground cover and pioneer species are tall fescue, sericea lespedeza, Korean lespedeza, and red clover.
10. Introduced woody trees recommended for potential forestry products are Norway spruce and Scotch pine.

Acknowledgments

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THE RELATIVE MERITS OF DOLOMITIC AND CALCITIC LIMESTONE IN DETOXIFYING AND REVEGETATING ACIDIC, NICKEL- AND COPPER-CONTAMINATED SOILS IN THE SUDBURY MINING AND SMELTING REGION OF CANADA[1]

by
Derek McHale and Keith Winterhalder[2]

Abstract. Soils in the Sudbury mining and smelting region have been rendered phytotoxic and barren by acidification and particulate metal contamination, but can be detoxified and revegetated by the surface application of ground limestone. On certain barren sites, plant growth is better on soil treated with dolomitic limestone than with calcitic limestone, and greenhouse experiments using mung beans (*Vigna radiata*) have shown superior root and shoot growth on certain contaminated soils when the limestone is dolomitic rather than calcitic. Results of experiments with species used in revegetation (*Agrostis gigantea* and *Lotus corniculatus*) suggest that leguminous species are more sensitive to Ca:Mg ratio than grasses, that the plant response to this ratio is greater at lower liming levels, and that the response is more marked on more toxic soils. The effects of calcium:magnesium ratio of the limestone used in revegetating acidic, metal-contaminated soils are clearly complex, interactive and difficult to interpret. Further studies are needed, but meanwhile it is recommended that the practice of using dolomitic limestone to detoxify barren Sudbury soils be continued, since there is a risk of induced magnesium deficiency at certain sites when calcitic limestone is used.

Additional Key Words: Ca:Mg ratio, liming rate, *Agrostis gigantea*, *Lotus corniculatus*

Introduction

Until recently, 10,000 ha of barren, phytotoxic land encircled the three Sudbury smelters (Coniston, Copper Cliff and Falconbridge). The barren soils, ranging in pH from 3.5 to 4.5, and with copper and nickel contents frequently exceeding 1,000$\mu g \cdot g^{-1}$, are the result of the effects of a century of mining and smelting activity, including SO_2 fumigation and copper, nickel, and iron particulate fallout from the smelters.

The construction of a 380m smokestack at the Copper Cliff smelter, plant closures, legislated reductions in SO_2 emissions, and improved emission control technology, have brought about a steady decline in atmospheric SO_2 and in particulate emissions from the smelters since 1972. A detectable rise in soil pH and fall in soil metal levels has occurred (Gundermann and Hutchinson 1993; Dudka *et al.* 1995), but recolonization of the barrens has been mostly confined to species that have evolved metal tolerance, such as *Deschampsia caespitosa* (Cox and Hutchinson 1980), *Agrostis gigantea* (Hogan *et al.* 1977), *Agrostis scabra* (Archambault and Winterhalder 1995), *Poa compressa* (Rauser and Winterhalder

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1985) and *Betula pumila* (Roshon 1988), and there are still areas close to the smelters that are devoid of vegetation. In these acid soils, aluminum ions are liberated from aluminosilicate minerals, as are copper and nickel ions from particulate smelter fallout. Seeds germinate, but unless the plant is from metal-tolerant stock, root growth is inhibited and the roots are unable to penetrate the soil.

Ground limestone, when applied to the surface of the toxic soil, acts as a 'Trigger Factor' (Winterhalder 1983), initiating spontaneous colonization by metal-intolerant plants from a local seed source. The stony mantle that covers these eroded, glacial till-derived soils traps limestone particles and seeds, acting as a protective mulch to the young seedlings. The soil is effectively detoxified, despite a modest pH rise of as low as one unit (e.g. from 4.5 to 5.5), allowing the seedling to develop a deep root system, protecting it against drought and frost-heaving, and enabling it to reach bases and other nutrients deeper in the profile. Ultimately, the leaf litter provides insulation, thereby reducing frost damage, while the humus helps to complex potentially toxic metals.

Winterhalder (1995) has suggested five possible components of the complex mechanism of detoxification. Precipitation of copper and nickel from solution is probably relatively rare, since a pH of above 5.5 is rarely achieved. However, the Al^{3+} ions that result from the hydrolysis of clay minerals are hydroxylated and rendered less toxic as the pH rises. Calcium ions enhance the plasma membrane integrity of root hair cells (Epstein 1961), and the abundant calcium ions reduce metal uptake by the roots by competitive exclusion (Kinraide and Parker 1987).

The amelioration of metal toxicity by calcium is well documented. Increased calcium can reduce the detrimental effects of aluminum on soybean (Lund 1970; Brady *et al.* 1993) and nickel toxicity in corn (Robertson 1985). Calcium-nickel interactions can be complex, and Knight and Crooke (1956) found that the calcium uptake of oat plants, but not of tomato plants, was higher in nickel-contaminated soils .

Quite fortuitously, the limestone that has been used in Sudbury's Regional Land Reclamation Program has been dolomitic, with a Ca:Mg ratio of approximately 2:1.

Calcium and magnesium levels in Sudbury soils are within the normal range for soil in a temperate region (Hazlett *et al.* 1983), despite a century of acid leaching, probably because both Ca and Mg are components of the emissions of the Copper Cliff smelter, at rates of over 300 $kg \cdot day^{-1}$. It therefore seems unlikely

that plants on Sudbury soils would suffer from a simple deficiency of calcium or magnesium. Nevertheless, Winterhalder (1983) suggested that the beneficial effects of liming Sudbury barren soils involved both neutralization and a calcium nutrition effect. Later (Winterhalder 1993), it was suggested that calcium:magnesium balance might play a critical role in the nutrition of plants in limed Sudbury soils.

Tan *et al.* (1992), when examining a variety of sorghum genotypes which had different sensitivities to acid soils, found that the benefit of liming is increased when there is an adequate magnesium supply. It is therefore possible that the exclusive use of calcitic limestone on Sudbury soils would bring about a calcium-magnesium imbalance and consequent induced magnesium deficiency, in which the superabundance of calcium ions would exclude magnesium uptake by competing on the root exchange sites, since magnesium is less competitive than calcium and aluminum under acid conditions (Sumner *et al.* 1991). In addition to its nutritional role, possible beneficial effects of magnesium include interaction with potentially toxic metals. Proctor and McGowan (1976) showed a clear alleviation of Ni toxicity by Mg in oats, and Robertson (1985) showed that both calcium and magnesium provided protection to corn roots against nickel poisoning, although at higher levels Mg itself became toxic unless alleviated by the appropriate Ca balance.

Previous studies on the effects of Ca and Mg have focused on agricultural plant species growing in soils which are acidic and therefore contain elevated levels of soluble aluminum, or else on serpentine soils which have a neutral to basic pH, high levels of nickel, chromium and magnesium, and low levels of calcium (Johnston and Proctor 1981). The Sudbury area soils are unique in being acidic like the former but metal-contaminated like the latter. The purpose of this paper is to examine the relative merits of dolomitic and calcitic limestone, with respect to the Ca:Mg ratio within the limestone, for detoxifying and revegetating acidic, nickel- and copper-contaminated soils in the Sudbury region.

Plant Growth Experiments

At the beginning of the investigation, several exploratory experiments were carried out to test the hypothesis that the two limestone types could have differential ameliorative capacities. Two examples follow.

1. Mung bean (*Vigna radiata*) on semi-barren soil. Mung bean seedlings were grown in soil collected from a semi-barren birch woodland 6 km south-west of the Falconbridge smelter and 6 km

north of the Coniston smelter (Maki Road). The four treatments were: no limestone (control), calcitic limestone, 50:50 calcitic:dolomitic limestone, and dolomitic limestone. A total of 1.5 g of limestone was mixed into 150 g of soil in each pot. As shown in Figure 1, both root and shoot growth were significantly greater in the dolomitic containing treatments than in the calcitic or control treatments.

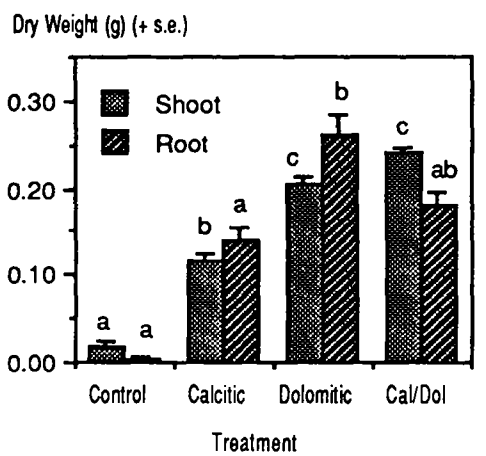

Figure 1. Mean *Vigna radiata* shoot and root dry weights in Maki Road soil under different limestone treatments (vertical bars represent standard error). Means for shoots or roots with the same superscript are not significantly different ($p<0.05$).

2. Redtop (*Agrostis gigantea*) on barren soil. Mung bean is a convenient experimental species, but has no relevance to the revegetation program. Therefore seedlings of redtop, a major revegetation grass species, were grown in barren soils from 10 km northeast of the Copper Cliff and 2 km south of the Falconbridge smelter, respectively, using the same protocol as above, except that 1.0 g of limestone was mixed into 150 g of soil. In neither soil did shoot biomass show significant response to dolomitic limestone, and the only significant root response was that seen in Copper Cliff soil (Figure 2), where the response was once again significantly greater in dolomitic treatments than in the control and calcitic treatments.

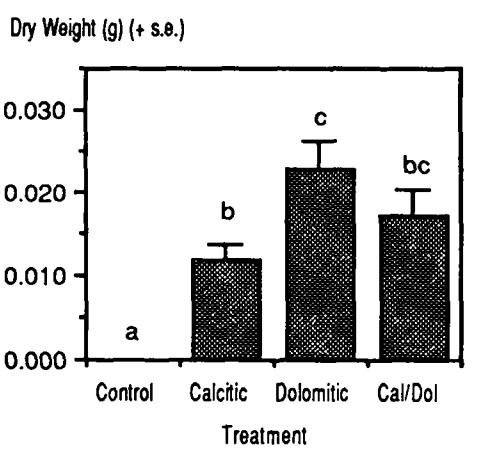

Figure 2. Mean *Agrostis gigantea* root dry weights in Copper Cliff soil under different limestone treatments (vertical bars represent standard error). Means with the same superscript are not significantly different ($p<0.05$).

Two additional, more comprehensive experiments were then carried out, the first looking at early seedling root and shoot growth under two lime types, and the second at longer term

growth under widely varying Ca:Mg ratios.

3. Petri dish bioassay using redtop (*Agrostis gigantea*) on three barren soils. Three barren soils from the vicinity of each of the three smelters (Table 1) were used in a Petri dish root/shoot growth bioassay.

Table 1. Chemical characteristics of soils used in Petri dish bioassay (Mean ± standard deviation for three samples at each smelter). Within rows, data followed by the same letter are not significantly different at the 5% level (Tukey Multiple Range Test). O.M. = Organic Matter, W.S. = Water Soluble, Ext. = Weak Acid Extractable, Tot. = Total.

	Falconbridge	Copper Cliff	Coniston
pH	3.75a	4.00b	3.85ab
O.M.	6.95%a	7.86%a	10.74%b
W.S.			
Ni	3.6±3.7a	11.9±5.5b	16.2±21.6c
Cu	2.95±1.4a	7.9±3.4b	16.8±4.5c
Al	11.0± 8.3a	29.24±9.1ab	64.8±60.4b
Ca	31.8±41.2a	42.35±41.4a	60.3±72.8a
Mg	4.1±2.7a	4.2±1.70a	6.2±6.8a
Ext.			
Ni	8.6±8.34a	20.0±12.0a	17.8±26.5a
Cu	71.0±32.5a	105.3±28.8a	176.3±76.2b
Al	381.3±230.4a	981.2±364.6b	1215.6±283.2b
Ca	74.0±77.7a	76.7±34.4a	105.2±110.7a
Mg	5.9±3.9a	6.9±5.5a	8.0±7.5a
Tot.			
Ni	410.3±191.0a	506.9±259.0a	611.6±314.2a
Cu	450.8±128.0a	484.3±131.9a	806.6±431.1b
Al (%)	4.3±0.3a	5.7±0.7b	5.3±0.8b
Ca (%)	2.9±3.2a	3.5±0.7a	2.7±1.7a
Mg (%)	2.0±0.3a	3.5±0.4b	3.6±1.7b

Values in $\mu g \cdot g^{-1}$ except where indicated.

40 g of each soil were placed in a filter paper lined perforated plastic Petri dish which was then placed in a larger glass Petri dish. The treatments were: no limestone (control), calcitic limestone, and dolomitic limestone, the limestone being applied by sprinkling 0.2 g onto the soil surface. Glass fibre filter paper was placed over the soil, and the soil was wetted from below. Twenty-five seeds of *Agrostis gigantea* were placed on the surface of each glass fibre filter paper.

The seeds were kept in darkness for 4 days, then in a 12 hour light/dark cycle for a further 12 days, after which the roots and shoots were measured to the nearest millimetre. In the case of the unlimed soils (data not shown), a gradient existed in the toxicity of the unlimed soils with respect to both the root and shoot growth of *Agrostis gigantea*. The roots in the Falconbridge soil were significantly longer than those in the other two soils. The shoot lengths were significantly the longest in the Falconbridge soil and the shortest in the Coniston soil, with the shoots in the Copper Cliff area soil being intermediate in length, but not significantly different from the others. In all cases there was a significant difference between the controls and the lime treatments.

In the limed treatments (Figure 3), there was no significant difference in root growth between limestone types for any

of the soils, but the fact that the Coniston soil was the most inhibitory to root growth, even after liming, was clear.

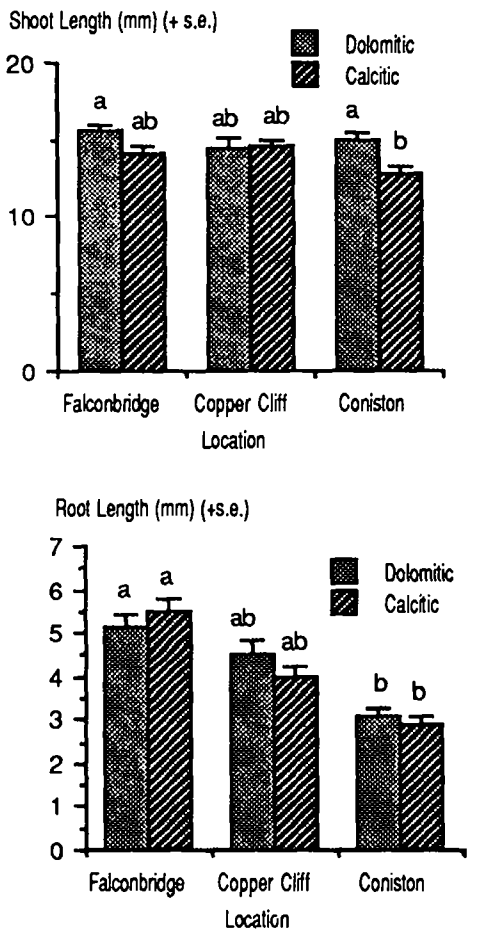

Figure 3. Mean shoot lengths (above) and root lengths (below) of young *Agrostis gigantea* seedlings grown on three Sudbury soils under different limestone treatments (vertical bars represent standard error). Means with the same superscript are not significantly different ($p<0.05$).

However, shoot growth showed a significant response to the dolomitic treatment on the Coniston soil, but not on the Falconbridge or Copper Cliff soil.

4. Ca:Mg Ratio Pot Experiment. Seeds of redtop and birdsfoot trefoil (*Lotus corniculatus* - a major leguminous species in revegetation) were sown in two contaminated soils (Coniston and Falconbridge) and a control soil (Cartier, 44 km northwest of the Copper Cliff smelter), each of which had been treated with finely-powdered calcium carbonate-dolomitic limestone combinations to give equal neutralization, but with Ca:Mg ratios of 1:0, 192:1, 96:1, 48:1, 24:1, 8:1, 4:1 and 2:1, respectively. Two neutralization levels were employed (0.2 g limestone•pot^{-1}, giving a pH of approximately 4.0, and 0.8 g limestone•pot^{-1} giving a pH of approximately 5.5). The experiment was replicated five times, and after 2 months of growth, roots and shoots were harvested and dry weights determined.

Results for the two plant species were analyzed separately, as were the two lime rates (moderate and low). Ca:Mg ratio had no significant effect on root or shoot growth of *Agrostis gigantea* at either lime rate on any soil, and for *Lotus corniculatus* there was no significant difference between treatments at the moderate lime rate for any soil. However, at the low lime rate, Ca:Mg ratio significantly influenced shoot mass in *Lotus corniculatus* in the Coniston area soil, with significantly higher shoot mass at Ca:Mg ratios of 4:1 or less (Figure 4).

Leaf chlorosis was consistently present in both contaminated soils at the low lime rate, but more intense under the higher calcium regimes.

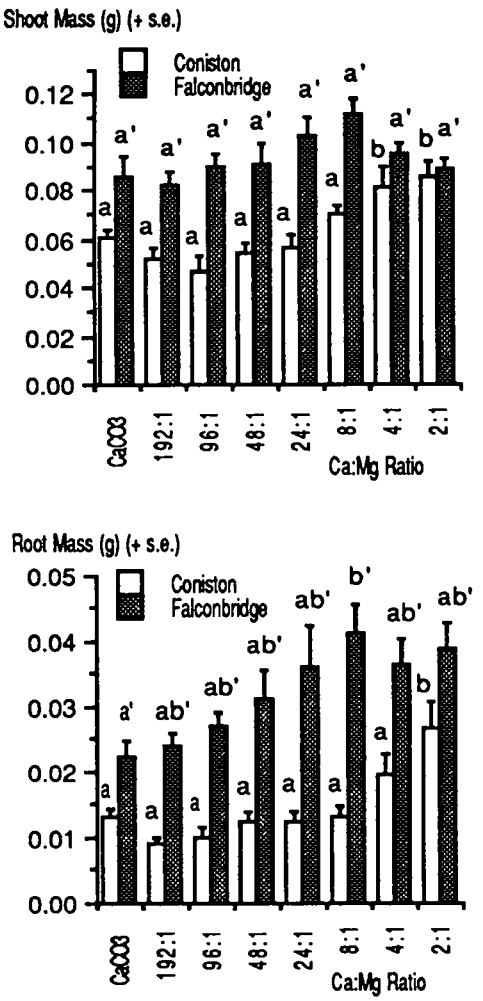

Figure 4. The effect of Ca:Mg ratio at low liming rate on the growth of birdsfoot trefoil shoots (above) and roots (below) on Coniston and Falconbridge soils (vertical bars represent standard error). Means with the same superscript are not significantly different ($p<0.05$).

Also at the low lime rate, Ca:Mg ratio significantly influenced root mass in *Lotus corniculatus* in both Coniston and Falconbridge soils (Figure 4). The uncontaminated Cartier soil showed no response to either the lime rate or the calcium-magnesium ratio for either species.

Discussion

The dramatic response of the mung bean seedlings to the dolomitic limestone treatment in the first experiment strongly supported the hypothesis, based on previous anecdotal evidence, that Ca:Mg balance can play an important nutritional and/or anti-toxic role on certain Sudbury area soils. However, redtop did not give the same consistent results on two different toxic soils in the second experiment. Indeed, the fact that the only significant response of redtop was that shown by roots in the Copper Cliff soil, whereas in the Maki Road soil the greater response of mung bean was evident in the shoots, suggests that the phenomenon is species and/or soil dependent.

There was a clear gradient in the toxicities of the three un-limed Sudbury area soils, with Coniston being the most toxic, followed by Copper Cliff, and finally Falconbridge. The Coniston area soil consistently had higher levels of copper and nickel, and a higher percentage of organic matter. Although all of the contaminated soils were barren when collected, they possessed different degrees of toxicity. It is still not clear

which of these factors is interacting with the limestone to produce the differential responses.

In the third experiment, a contradictory situation arose, in which there was no significant root response to dolomitic limestone in any of the soils, but a definite shoot response on the Coniston soil, which also appeared to be the most toxic. The difference here could have been soil-related. At higher soluble metal concentrations, the competition for exchange sites could reduce the magnesium available to the shoots, resulting in a lime induced magnesium deficiency, but at slightly lower metal levels the ameliorating effects on the metal toxicity in the rhizosphere would be the over-riding factor. Furthermore, the Coniston soil had the highest organic matter content. Although organically bound copper is less rhizotoxic than the ionic form, it is possible that it could be taken up by the plants and translocated where metal ions could not, as shown in the case of EDTA-chelated copper in *Triticum aestivum* by Taylor and Foy (1985). Alternatively, the difference could be the result of the fact that the organs measured were radicles and first leaves, rather than mature root and shoot systems, as well as the fact that length, rather than biomass, was measured.

In the fourth experiment, an interesting difference between the behaviour of the grass (redtop) and the legume (birdsfoot trefoil) appeared, in that there was no significant response to dolomitic limestone in redtop at any liming level in any soil, whereas birdsfoot trefoil shoots showed a significant response at the low liming level. Similar results were found by Edmunds *et al.* (1983) in a study on the effect of calcitic limestone on ryegrass and white clover grown on a New Zealand yellow-brown loam. After examining yield and internal magnesium concentrations, they showed that magnesium concentration in the grass decreased in proportion to increasing yield, suggesting a dilution effect. In the clover, however, magnesium concentration decreased with plant yield reduction, suggesting that the calcitic limestone had an antagonistic effect on magnesium uptake. One consistent feature in the Sudbury and New Zealand experiments has been the apparent greater sensitivity to Ca:Mg ratio of leguminous species, compared to grasses, possibly due to the lower ability monocot roots have to bind cations such as calcium (Rorison and Robinson 1984).

McNaught *et al.* (1973) were able to demonstrate that liming with $Ca(OH)_2$ in a pasture made up of primarily white clover could reduce the tissue magnesium levels compared to magnesium-containing liming materials. The legume used in the

present study, *Lotus corniculatus*, may be particularly sensitive to the Ca:Mg ratio in the soil. In a study on acidic, infertile Appalachian soils, Baligar *et al.* (1985) examined the differences in growth and nutrient uptake in six legume species. The soils were limed with $Ca(OH)_2$, and the only species that did not respond positively to the lime was the birdsfoot trefoil. However, the liming reduced the magnesium levels in all legume species to low to deficient rates, although, unlike birdsfoot trefoil, the other legumes, including white clover, responded with increased yield. Clearly, response of legumes to liming is species-specific, but the nature of the species-specific response factor or factors is not known.

It may be helpful to attempt to interpret the results of these experiments in the light of the two probable functions of the limestone - the amelioration of rhizotoxicity with its concomitant root growth inhibition, and the reduction of metal uptake (particularly of nickel) and concomitant effects on shoot growth and vigour. The amelioration of root toxicity would involve such things as the hydroxylation of aluminum ions and competitive exclusion of metals by base ions, whereas shoot effects would involve mechanisms that reduce nickel uptake such as magnesium:nickel interactions, plasma membrane integrity effects and (once again) competitive exclusion. On this basis one would expect response to dolomitic limestone to be expressed in the leaves more than in the roots, and this is indeed the case in three of the four experiments. The reduced shoot growth of *Lotus corniculatus* in the Coniston area soil at the higher calcium-magnesium ratios could be the result of a lime induced magnesium deficiency. Because the Mg^{2+} ion is also involved in the amelioration of nickel toxicity (Robertson 1985), its effectiveness is reduced when it is in lower concentrations, relative to calcium.

On the other hand, the presence of an increased response to the lower Ca:Mg ratios in the roots of the *Lotus corniculatus* growing in the Falconbridge soils, but not in the shoots, would indicate a reduction in copper and aluminum toxicity afforded by the higher magnesium concentrations.

Christenson *et al.* (1973) examined oat plants growing in acid soils amended with different calcium and magnesium sources. They showed that a high Ca:Mg ratio suppressed magnesium uptake, although low pH had a greater effect on magnesium uptake than did calcium source. This could explain, in part, the fact that birdsfoot trefoil only responded to dolomitic limestone at a low liming rate. Sumner *et al.* (1978) showed that, in highly weathered soils which have been limed with finely ground

limestone, the amount of exchangeable Mg increases to roughly pH 6, then decreases, as the pH approaches neutrality. Thus, as the pH of an acid soil is adjusted towards pH 6, a greater proportion of the magnesium will be held on exchange sites, rather than in the soil solution, making it more available to the plants (Sumner *et al.* 1991). While on the exchange sites, the competitive disadvantage of the magnesium is lost, and the lime induced deficiency no longer exists. This was observed in the last Sudbury experiment, in that there were no significant responses to Ca:Mg ratios at the higher lime rate. Tan *et al.* (1991) also found that, on aluminum toxic soils, there was a significant response of aluminum sensitive sorghum genotypes to the addition of magnesium at low rates of liming, but as the lime rate increased the effectiveness of the magnesium was reduced. In the light of Sumner *et al.*'s (1991) observations, it is possible that, if the Sudbury soils were limed to above pH 6, there would be a decrease in available magnesium, resulting in a reduced yield.

The importance of soil toxicity level was again apparent in the Ca:Mg ratio experiment. There was a strong response in the highly toxic Coniston soil, whereas there was little response in the less toxic Falconbridge soil. Shoot growth responded significantly to Ca:Mg ratio in the Coniston soil, but not in the Falconbridge soil. Root growth in the toxic Coniston soil was highest at the 2:1 ratio, while in Falconbridge soil it peaked at 8:1. Plants on the uncontaminated Cartier soil did not respond to either liming rate or Ca:Mg ratio.

Since other authors have demonstrated lime induced magnesium deficiency in various soil types and plant species, the fact that it is not evident in all Sudbury area soils indicates that there are other factors operating. From the experiments so far conducted, it is clear that the effects of the calcium-magnesium ratio in limestone are complex and interactive, and it is difficult to identify which factors in the soil are reacting with which elements of the limestone. There is evidence, however, that there is a relationship between the phytotoxicity of limed acid metal-contaminated soils and the calcium-magnesium ratio of the limestone employed. Further studies are needed to further define this relationship.

It is recommended that the practice of using dolomitic limestone to detoxify barren Sudbury soils be continued, since there is a risk of induced magnesium deficiency at certain sites when calcitic limestone is used.

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ONTOGENY OF ACCIDENTAL WETLANDS AND HYDRIC SOIL DEVELOPMENT IN SURFACE MINED LANDSCAPES

by

Robert B. Atkinson, W. Lee Daniels, and John Cairns, Jr.

Abstract. Reducing conditions are periodically present in hydric soils and are essential for chemical processes that support wetland functions and values. Indicators of these conditions, i.e., redoximorphic features, can be useful in determining the presence of a hydric soil. However, young wetlands, i.e., those recently formed, may not possess reducing conditions and/or may not exhibit redoximorphic features. Few studies have addressed the time needed for hydric soil development. In this study, we present data on redoximorphic features, including chroma and oxidized rhizospheres, gathered from two sets of wetlands in southwestern Virginia, including (1) constructed wetlands that are 3 years old and (2) accidental wetlands that are 10 to 30 years old. Under conditions described for these sites, there is strong evidence that discernable redoximorphic features form in accidental wetlands within 10 years, but not within 3 years in constructed wetlands. Since accidental wetlands have been in existence for longer than most man-made wetlands, they provide clues to the development of hydric soils in recently constructed wetland systems.

Additional Key Words: litter, hydrophytic vegetation, redoximorphic features, sedimentation, SMCRA, wetlands.

Introduction

Small depressions that formed accidentally following contour surface mining in southwestern Virginia may meet all three criteria to be considered wetlands, according to the Corps of Engineers Wetlands Delineation Manual (Environmental Laboratory 1987), which is the delineation manual currently in use. According to this manual, accidental wetlands (referred to as "man-induced") may not exhibit hydric soil indicators for "hundreds of years." A more accurate and precise estimate of time required for establishment of hydric soil indicators is needed (1) to aid in delineation of recently formed wetlands and (2) to aid in structural and functional assessment of constructed wetlands by providing evidence of functions that depend on reducing soil conditions.

Many wetland functions depend upon hydric soils. Hydric soils influence decomposition rate (Reddy et al. 1986) and distribution of organic matter (Megonigal and Day 1988), as well as plant distributions (Pearsall 1938). Many nutrients and other chemicals are transformed in wetlands, including nitrogen (Broadbent and Clark 1965), phosphorus, and iron (Gambrell and Patrick 1978). Pesticide and metal transformations also take place in hydric soils (Gambrell and Patrick 1978). Considerable methane is oxidized in hydric soils and may influence atmospheric gas balances (King et al. 1990).

Under certain conditions, soil redox potential decreases and hydric

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soil indicators begin to develop. The rate of oxygen diffusion in soil decreases by a factor of approximately 10,000 once a soil becomes saturated (Gambrell and Patrick 1978). Aerobic soil microbes consume organic matter and use oxygen preferentially as an electron acceptor. Some oxygen demand can also be attributed to oxidation of reduced elements diffusing up from anaerobic layers (Simpson 1978). Microbial activity and chemical transformations continue to occur under anaerobic conditions and the redox potential of the soil is lowered, i.e., reducing conditions become established (Ponnamperuma 1972).

Once reducing conditions become established for sufficient frequency and duration each year, several years may be required for redoximorphic features such as low chroma and oxidized rhizospheres (iron oxide coatings on fine roots) to develop.

Naturally occurring wetlands typically exhibit redoximorphic features within 25 cm of the surface or just below the A horizon, which are used in wetland delineation (Vepraskas 1995). Specifically, matrix chroma must be 1 or less if no color pattern (depletions and masses) occurs (Environmental Laboratory 1987; Federal Interagency Committee for Wetland Delineation 1989). According to the Environmental Laboratory (1987), wetlands that have been "incidentally created by human activities" are considered "atypical situations." This section states that hydric soil indicators may be lacking, and delineation may not require that hydric soil indicators be present. Thus, assessments of constructed wetlands, including wetlands constructed as mitigation for destruction of natural wetlands, may not currently consider soil chroma and oxidized rhizospheres for measuring success. This paper provides information on the time required for development of redoximorphic features in small depressions in southwestern Virginia.

Site Description

Accidental Wetlands

Contour surface mining for coal disturbed over 385,000 ha in the Appalachian Mountains prior to enactment of the Surface Mining Control and Reclamation Act of 1977 (SMCRA). Topographic features left by mining included vertical "high walls" and fairly flat "benches" consisting of severely compacted spoil with bulk density of 1.7 g/cc (Daniels and Amos 1982). Sediment was deposited over compacted mine spoil in many small depressions on benches. As a result, wetland hydrology became established and hydrophytic vegetation colonized the depressions (Atkinson and Cairns 1994). Over time, these conditions led to lowered soil redox potential and establishment of redoximorphic features in the accidental wetlands we studied.

Table 1. Indicator categories and probability of a plant species occurring in a wetland (Reed 1988)

Indicator Category	Wetland Frequency
Obligate wetland	>99%
Facultative wetland	67-99%
Facultative	34-66%
Facultative upland	1-33%
Obligate upland	<1%

Sites were located in Wise County in southwestern Virginia. Selection criteria included the presence of hydrophytic vegetation and depressional wetland hydrology and the absence of significant acid mine drainage inputs. Each wetland exhibited two plant community types: (1) a community dominated by obligate wetland plants and (2) a community dominated by facultative wetland plants (Table 1).

Year of accidental wetland formation ranged from 1965 to 1986. Maximum water depth in accidental wetlands ranged from 89 cm in site 3 to 8 cm in site 2 (Table 2). Aboveground biomass estimates in 1994 for the facultative wetland community were 442 g/m^2 and were 398 g/m^2 for the obligate wetland community. Total precipitation over a 7-year period

Table 2. Site reference number, hydrology (sources of water, duration dry, and hydrology modifier), and age (year formed) for 12 accidentally formed wetlands in this study

Site	Inlets/ outlets	Water depth[1]	Draw-down[2]	Hydrology[3]	Year formed
1	No/no	40	0.0	PEMe	1978
2	Temp/temp	7.9	66	PEMb,a	1986
3	Temp/perm	89	0.0	PEMe	1974
4	No/temp	18	25	PEMc	1974
5	No/no	18	13	PEMc	1973
6	No/temp	62	13	PEMc	1973
7	Temp/temp	20	4.2	PEMd	1983
8	Perm/perm	44	0.0	PEMe	1974
9	Temp/temp	8.4	21	PEMc	1965
10	Temp/temp	60	4.2	PEMd	1970s
11	No/no	32	8.3	PEMd	1970s
12	Temp/temp	73	0.0	PEMe	1970s

[1]Maximum measured depth in cm
[2]Percent of three growing seasons without inundation
[3]Modifiers based on Cowardin et al. (1979)include PEM=palustrine system and emergent class; water regime modifiers include: a=temporarily flooded, b=seasonally flooded, c=semipermanently flooded, d=intermittently exposed, and e=permanently flooded

(1988 to 1994) in Wise, VA, showed significantly low precipitation in 1989 and 1993 (Fig. 1). Mean monthly maximum temperatures during that period ranged from a low in February of 7°C to a high in July of 27°C(National Weather Service Cooperative Observer, Wise, VA; Fig. 2).

Most of the adjacent uplands were characterized by herbaceous species, including *Festuca elatior*, *Lespedeza cuneata*, *Trifolium repens*, and *Solidago gigantea*. Terrestrial vegetation dynamics in the region are discussed elsewhere by Holl and Cairns (1994).

Constructed Wetlands

In 1992, six wetlands were constructed at two sites by excavating depressions and creating small berms. Mean wetland size was approximately 10 m x 30 m, and maximum depth was approximately 1 m. Because of the small size, data collection was limited to nondestructive techniques, e.g., vegetation cover estimates and soil probes. Species with highest modified importance values (sum of relative cover and relative frequency) were *Echinochloa walteri*, *Eleocharus obtusa*, *Scirpus cyperinus*, and *Typha latifolia*.

The Powell River Project (PRP) in Wise County, VA, was the site of three constructed wetlands. Sedimentation rates were low since wetland construction occurred after the site had been revegetated with the perennial ryegrass *Lolium perenne* and various legumes (Fabaceae). Three wetlands were also constructed near Wise, VA. Sedimentation rates were much higher at this site since wetland construction coincided with final grading and preceded revegetation efforts.

Methods

Twelve accidental wetlands in Wise County, VA, were studied between 1992 and 1995. Parallel transects and permanent plots were established and monitored for 3 years. Accidental wetland sampling was conducted within two communities: (1) a community dominated by obligate wetland plants and (2) a community dominated by facultative wetland plants. Constructed wetland sampling was not conducted on a community basis due to

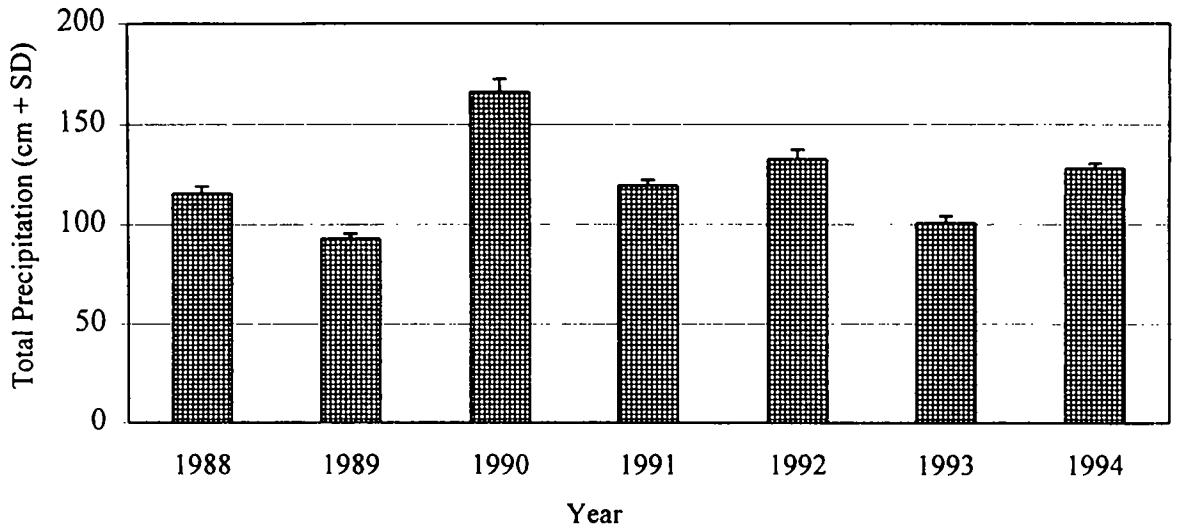

Fig. 1. Annual precipitation totals during a 7-year period (1988-1994) recorded in Wise, VA

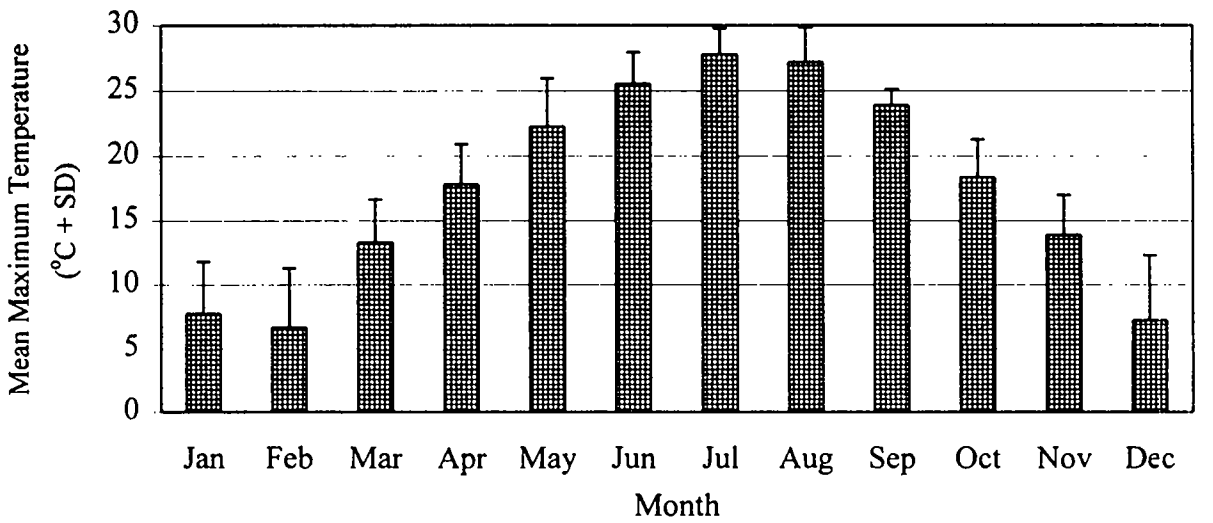

Fig. 2. Mean monthly maximum temperatures during a 7-year period (1988-1994) recorded in Wise, VA

small dimensions and poorly defined vegetative zonation. Several parameters were studied and those that aid in characterizing soils are described here.

Soil value and chroma, prevalence of oxidized rhizospheres, and presence of iron depletions and masses were assessed at a minimum of two plots within each accidental wetland community and within the deeper portion of each constructed wetland. Soil samples were extracted by shovel and were broken open manually for inspection. Value and chroma estimates were made using a wetted sediment sample in full sun using a Munsell Soil Color Chart. Oxidized rhizosphere frequency and intensity were assessed by visual inspection. No iron depletions and masses were detected in any soil samples in this study, although low chroma matrices were commonly described.

Water depth at the deepest point in each accidental wetland was measured biweekly during each growing season from 1992 through 1994. Each accidental wetland was assigned a water regime modifier based on the classification scheme set forth in Cowardin et al. (1979). Elevation in all plots was measured using a transit and stadium rod, and the mean of three estimates was calculated for each plot. Plot depth was calculated based on elevation and biweekly water depth estimates. Water depth in constructed wetlands was also measured biweekly. Sediment depth was measured using a soil probe penetrated to depth of compacted mine spoils, and the mean of three estimates was determined for each plot. Grand means were calculated for each community in the accidental wetlands and for each constructed wetland site. Soil particle size analyses were conducted at each accidental wetland and were compared to published data on upland soils that was also collected at PRP by Daniels and Amos (1982).

Litter estimates were made in both communities within the 12 accidental wetlands in 1994. Litter, including surficial organic material as well as standing dead biomass, was collected in 0.25-m^2 quadrats, dried, and weighed. No quantitative estimates of litter were made in constructed wetlands; however, cover estimates were made from 1-m^2 quadrats along two belt transects in each of the six constructed wetlands.

Belowground biomass estimates were made in both communities within the 12 accidental wetlands in 1994. All plant biomass between underlying compacted mine spoils and the litter layer were collected within 0.25-m^2 quadrats. No attempt was made to separate live and dead roots. Material was washed and sieved through a #10 seive, dried, and weighed. Because of the small dimensions of constructed wetlands, no estimates of belowground biomass were made at these sites.

Results

Accidental Wetlands

Redoximorphic features were present in all 12 accidental wetlands in this study. Low chroma (1 or less) matrices and oxidized rhizospheres were common in accidental wetlands; however, no pattern of iron depletions and masses was observed in any wetlands in this study (Fig. 3).

Hydric soil criteria for chroma were met in all 12 obligate wetland communities and 11 of 12 facultative wetland communities (Fig. 4). A chroma of 0 was common in the obligate wetland community of all four accidental wetlands that exhibited permanent inundation and in site 11, which was inundated throughout the study except for July, 1993, during a drought. Only the facultative wetland community at site 10 failed to meet the chroma criterion of 1 or less. Mean soil chroma in the obligate wetland community (0.67; SD 0.70) was somewhat lower than for the facultative wetland community (1.1; SD 0.68)(Fig. 4).

Oxidized rhizospheres were present at all accidental wetlands except site 12. Obligate wetland communities that exhibited chroma greater than 0 were found to have "prevalent and bright" oxidized rhizospheres. Oxidized rhizospheres were described as "absent" (sites 8, 10, and 12) or "present and dull" (sites 1, 3, and 11) in obligate wetland communities that exhibited chroma of 0. Oxidized rhizospheres in the facultative wetland community were described as "prevalent and bright" for all accidental wetlands except site 12, which exhibited dense growth of a moss, *Sphagnum* sp., associated with negligible mineral material above compacted mine spoil.

The duration criterion for wetland hydrology (Environmental Laboratory 1987) was exhibited by all accidental wetlands except site 2. Water regime modifiers for accidental wetlands included permanently flooded (4 sites), intermittently exposed (3 sites), semipermanently flooded (4 sites), and seasonally flooded (site 2)(Table 2). After August, 1992, inundation was not detected at site 2, which is the youngest of the 12 accidental wetland sites (formed 1986)(Table 2). Precipitation totals for 1993 were the second lowest on record in Virginia (National Oceanic and Atmospheric Administration 1993). Surface water disappeared from 8 of the 12 accidental wetlands during

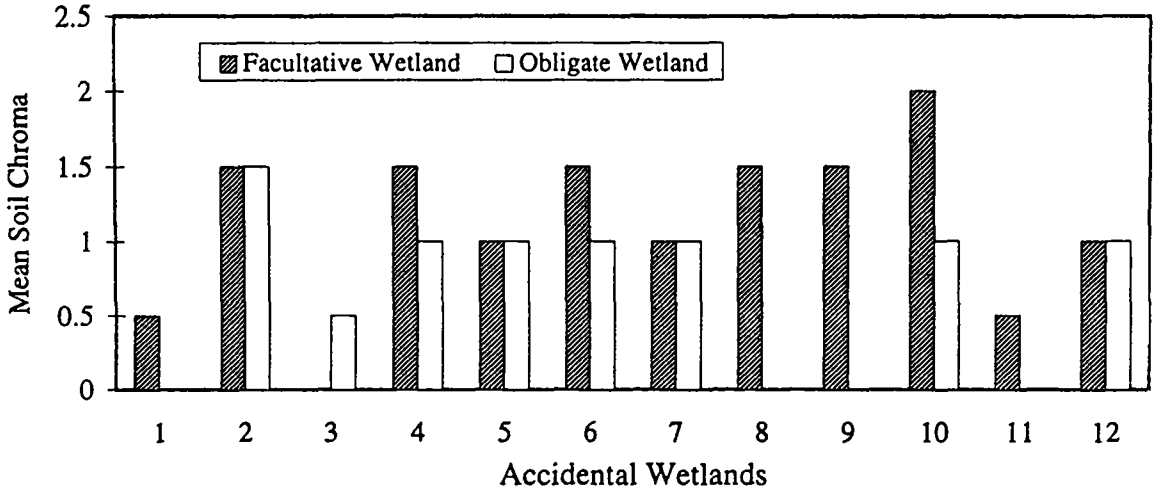

Fig. 3. Mean soil chroma for facultative wetland and obligate wetland communities at 12 accidental wetlands

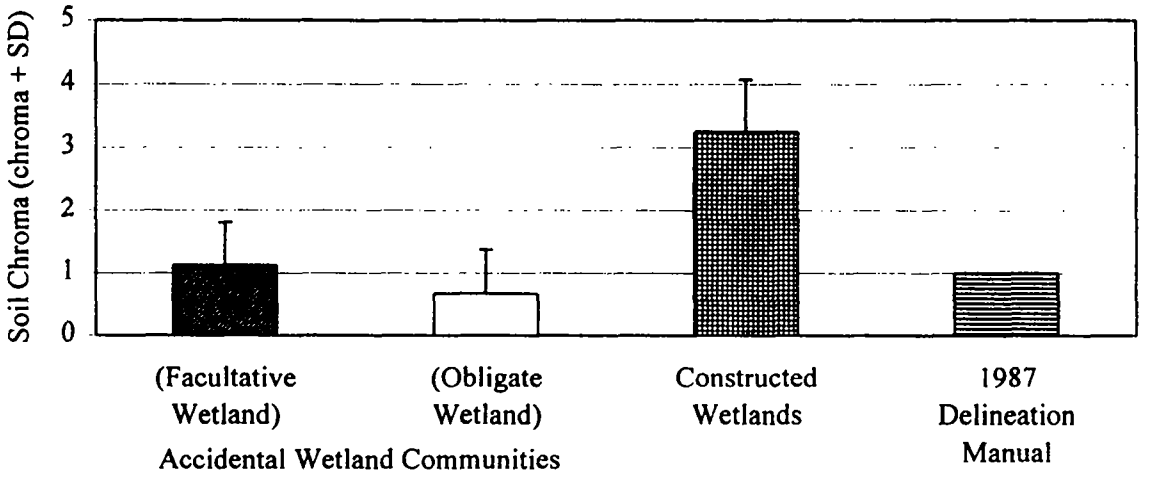

Fig. 4. Mean soil chroma for accidental wetlands, constructed wetlands, and maximum according to the 1987 delineation manual

1993, but inundation was reestablished during the 1993 growing season except at sites 2, 4, and 9. Precipitation in 1994 was within the range considered normal, and inundation was reestablished at sites 4 and 9, but not site 2.

Hydrology also differed within accidental wetlands. Mean water depth in obligate wetland communities (25 cm) was 24 cm deeper than mean facultative wetland community water depth (1 cm)(Fig. 5). Thus, inundation periods were shorter for facultative wetland communities than for obligate wetland communities in most sites. Soil saturation may have persisted longer than inundation, but was not quantified in this study.

Sediment depth in accidental wetlands differed both among and within the two communities. Mean sediment depth for both communities was greatest at site 10, which functioned as a sediment retention pond throughout this study. Sediment depth exceeded the maximum measured depth of 80 cm in five of the six

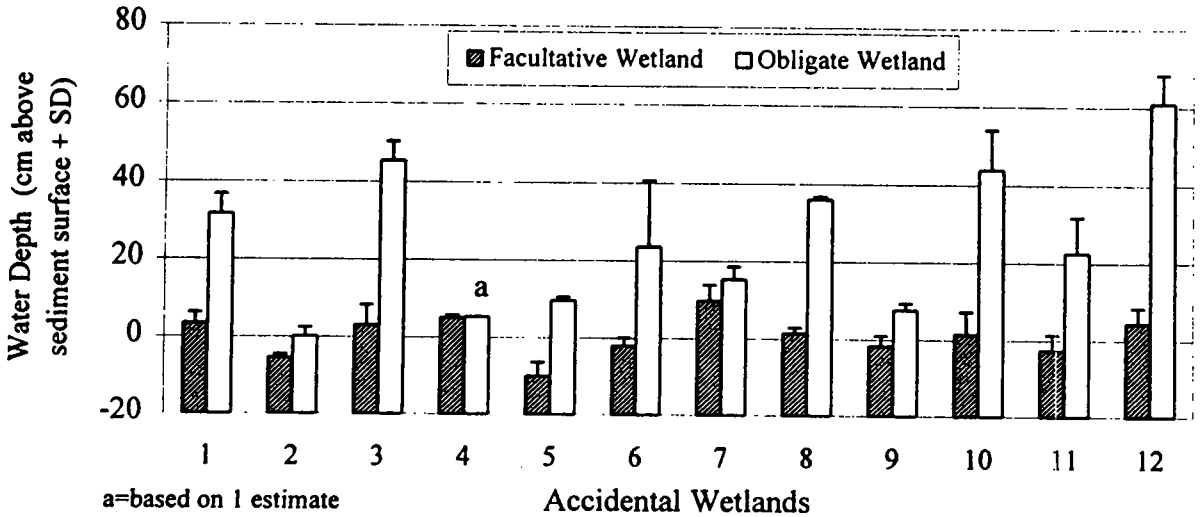

Fig. 5. Mean water depth from April through September, 1994, for facultative wetland and obligate wetland communities at 12 accidental wetlands

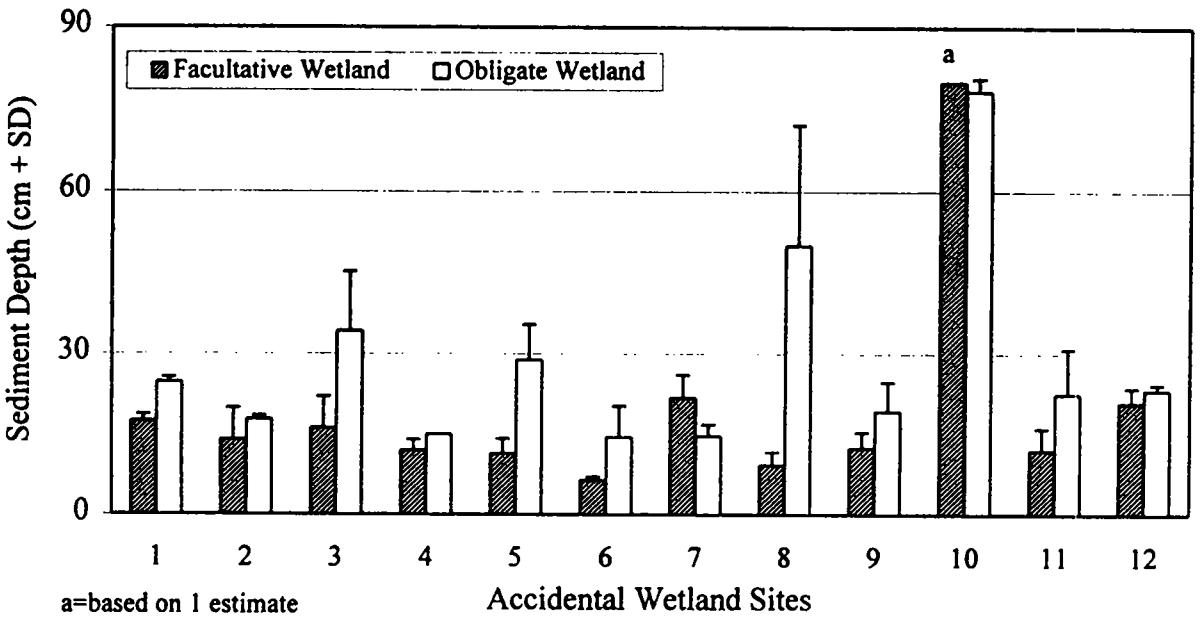

Fig. 6. Sediment depth for facultative wetland and obligate wetland communities at 12 accident wetland sites

plots where measurements were taken at site 10. Mean sediment depth was lowest at site 6 for both the facultative wetland (6 cm) and obligate wetland (14 cm) communities. In 10 of the 12 accidental wetlands, obligate wetland sediment depth exceeded that for the facultative wetland. Median sediment depth for facultative wetland communities was 13 cm compared to 23 cm for obligate wetland communities (Fig. 6). Particle size analyses for accidental wetlands indicated a significantly higher percent clay (31%) than that for upland areas (15%). Silt content in accidental wetlands (58%) was also somewhat higher than that for uplands (41%)(Fig. 7).

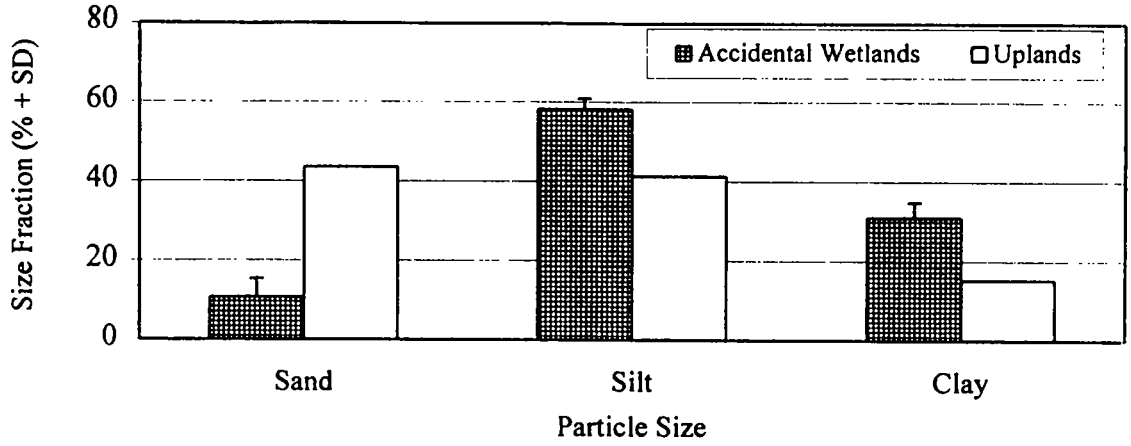

Fig. 7. Particle size analysis for accidental wetlands (this study) and for uplands at the Powell River Project (Daniels and Amos 1982)

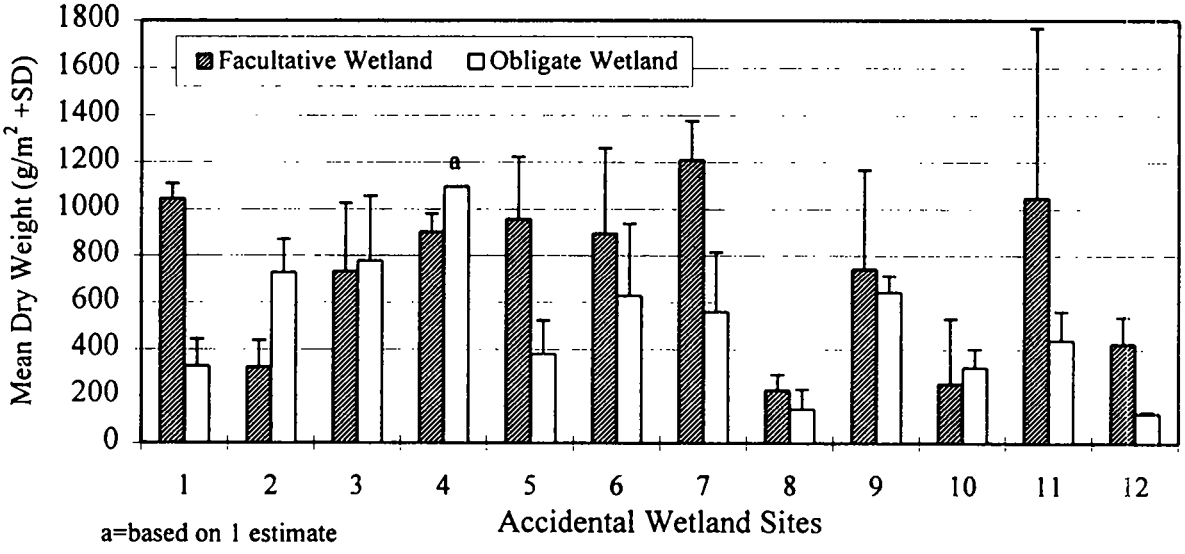

Fig. 8. Mean dry weight of litter for facultative wetland and obligate wetland communities at 12 accidental wetlands

Mean litter content in accidental wetlands was 621 g/m² in 1994. Facultative wetland litter content (730 g/m²) was significantly greater than litter content in obligate wetland communities (514 g/m²). Litter content was lowest in site 8 (184 g/m²), which was found to have the highest diversity and abundance of aquatic macroinvertebrates (Jones 1995; Fig. 8).

Dense root mat formations with negligible accumulated mineral material were common in the facultative wetland plant community. Belowground biomass in accidental wetlands was higher in the facultative communities, 442.5 g/m², than in obligate wetland communities, 398.4 g/m². Site 2 was the most recently formed accidental wetland (1986) and exhibited the lowest mean belowground biomass in the facultative wetland community (135.2 g/m²)(Fig. 9).

Constructed Wetlands

Soil chroma was 3 to 4 in all six newly constructed wetlands in this study. In all three constructed

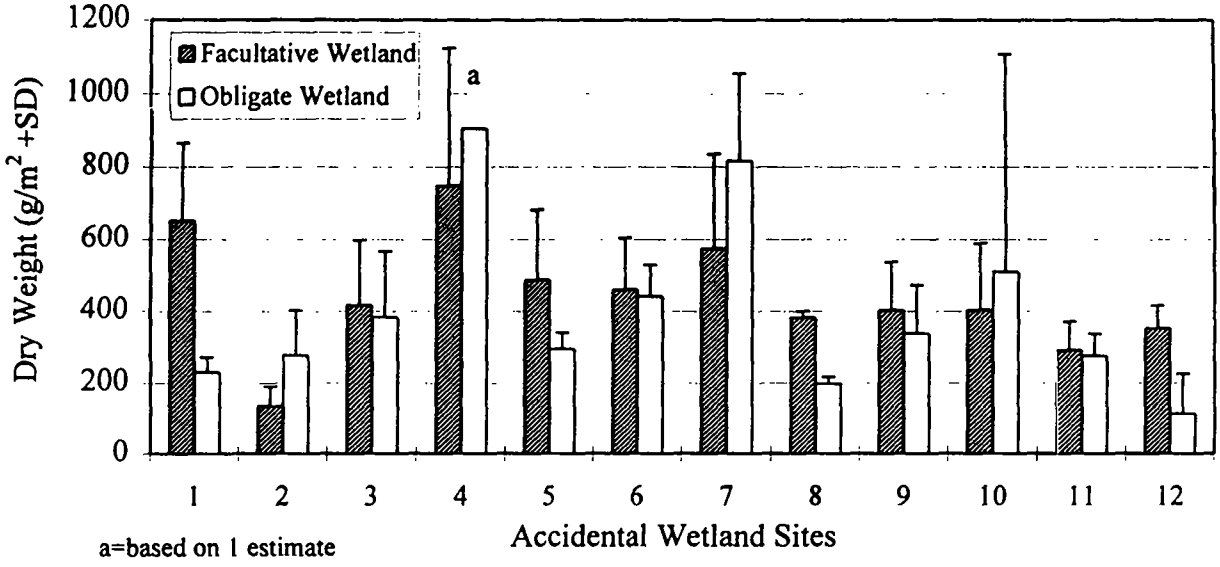

Fig. 9. Mean dry weight of biomass (belowground) for facultative wetland and obligate wetland communities at 12 accidental wetlands

wetlands near Wise, VA, oxidized rhizospheres were present, but were described as "infrequent and dull." No oxidized rhizospheres were found in the three constructed wetlands at PRP. Mean vegetative cover (68%) was significantly greater at the Wise constructed wetlands than at the PRP constructed wetlands (9%).

Sediment depth in constructed wetlands was significantly different among the PRP and Wise constructed wetlands. The mean sediment depth for PRP constructed wetlands was 5.8 cm compared to 19.3 cm for the Wise constructed wetlands (Fig. 10).

Discussion

The hydric soil criteria for chroma (1 or less) and, to a lesser extent, oxidized rhizospheres (Environmental Laboratory 1987) were met by both facultative and obligate wetland communities in all 12 accidental wetlands, including site 2. Since site 2 was formed in 1986, one can infer that a low soil chroma can be exhibited in less than 10 years after a depression becomes established. The minimum time required for chroma of 1 to be exhibited within small depressions is greater than 3 years, based on the survey of six constructed wetlands in this study. Oxidized rhizospheres may form more quickly. They were described as "present and dull" in two of the newly constructed wetlands.

The minimum time required for development of low chroma is probably even shorter, since site 2 failed to exhibit inundation during or after drought conditions began in spring 1993; and, since precipitation in 1989 was even lower than for 1993 in Wise County (National Weather Service Cooperative Observer, Wise, VA), site 2 probably was not inundated in 1989.

The issue of low chroma color development in these materials is complicated by the fact that some of the mine spoil strata encountered are reduced and gray when mined and freshly deposited (Daniels and Amos 1981). However, these reduced sandstones and siltstones are typically chroma 2 or higher, and weathering for even a few years raises the matrix chroma to 3 or 4 under well drained conditions. Thus, the higher chroma sediment colors observed in the constructed wetlands corroborates our assumption that the low chromas (<1.0) observed in the accidental wetlands we surveyed reflect the development of hydric soil conditions rather than simple preservation of relic parent material colors.

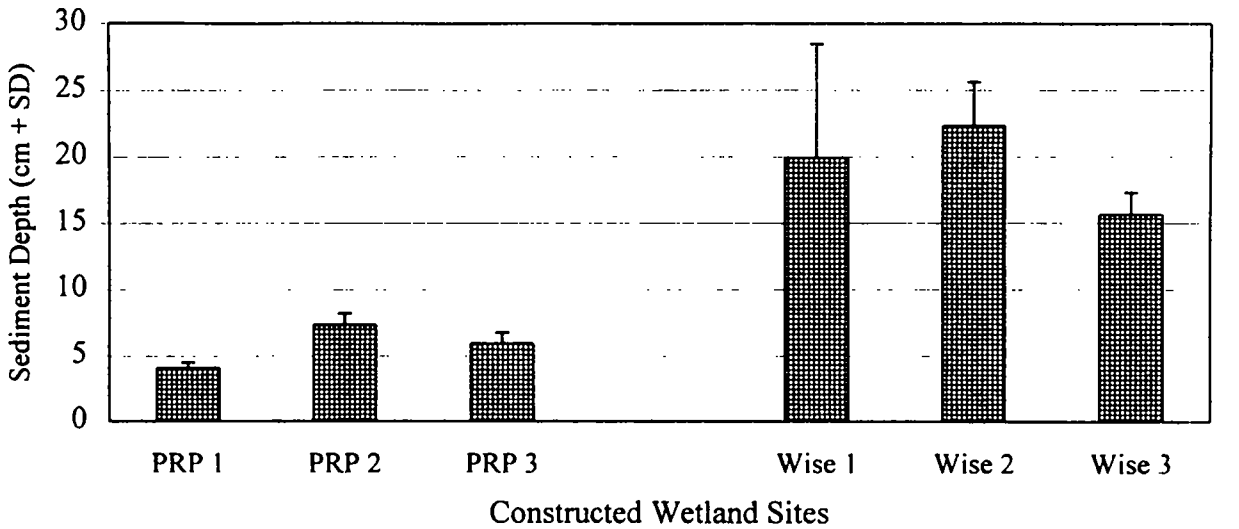

Fig. 10. Mean sediment depth in constructed wetlands at two sites, Power River Project (PRP) and near Wise, VA (Wise)

Mitsch and Gosselink (1993) list three conditions for establishment of low chroma: (1) sustained anaerobic conditions, (2) sufficient organic matter to support microbial activity, and (3) adequate soil temperature. These conditions were met to differing extents in all 12 accidental wetlands and influence the time required for development of hydric soil indicators.

Wetland hydrology, especially the duration of inundation, appeared to influence soil chroma. Sites having permanently flooded water regimes were found to exhibit lower chroma than sites with either intermittently exposed or semipermanently flooded conditions (Fig. 11). Water depth did not appear to influence chroma, except for the fact that deeper portions of a wetland may be inundated for longer periods of time. For example, facultative wetland communities were not inundated for as long as the associated obligate wetland communities and exhibited higher chroma.

The actual duration of wetland hydrologic conditions cannot be quantified for the accidental wetlands since there is no data to indicate how much time was required for an impervious layer to form. Particle size analysis in accidental wetlands indicated a significantly higher clay content when compared to spoil material on upland benches (Daniels and Amos 1981), which may have limited infiltration rates. Constructed wetlands at PRP failed to form an impervious layer in the first 2 years after construction, and there was considerable fluctuation in water depth. The shrink-swell characteristics of sediments appeared to permit infiltration following drawdown events.

Vepraskas and Wilding (1983) noted that soils saturated for 120 days failed to show chroma below 3 when organic matter content was less than 1%. Organic matter was measured in both litter and belowground biomass compartments for both communities in the 12 accidental wetlands. The supply appears to be ample to support microbial metabolism, but may have been limiting during early stages of wetland development. There are two possible roles for organic matter in hydric soil development in accidental wetlands. First, organic matter is required for microbial metabolism, which leads to iron reduction and lower chroma. Organic matter may have been available early in accidental wetland development since wind dispersed seeds, e.g., *Typha latifolia* and *Scirpus cyperinus*, could colonize sediments quickly.

Second, the water holding effects of organic matter can increase the duration of soil saturation. Since accidental wetlands form in

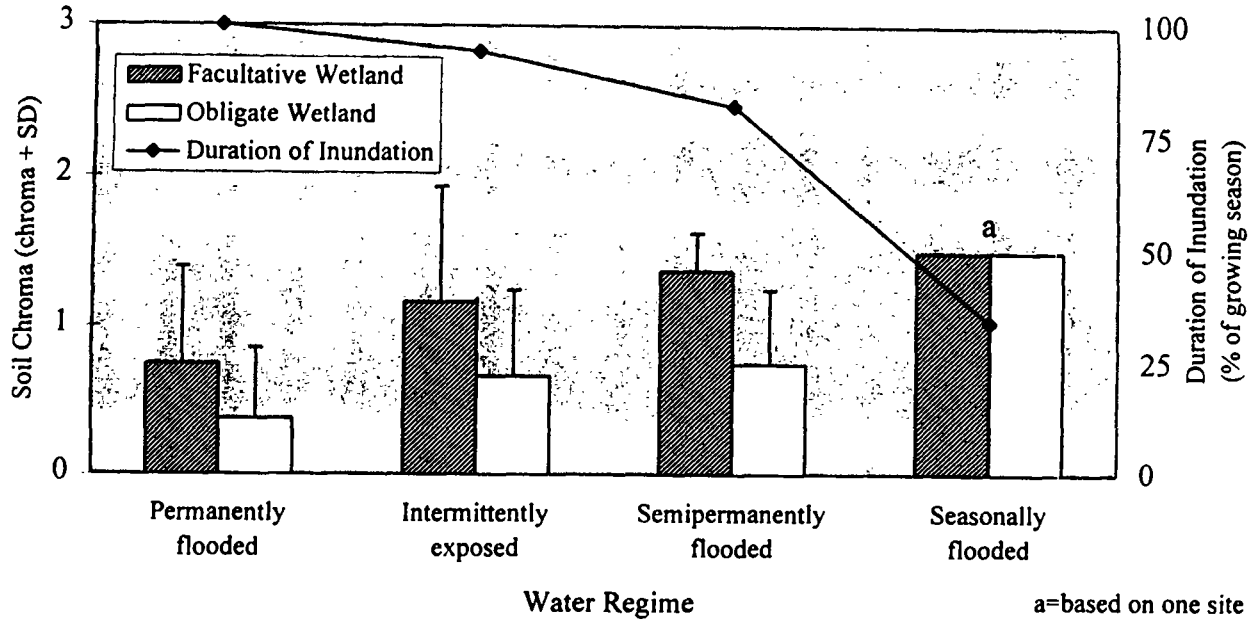

Fig. 11. Soil chroma for four accidental wetlands arranged by water regime (based on Cowardin et al. 1979)

small depressions, export of organic matter is low. Since mean accidental wetland litter content was high (621 g/m^2 in 1994), absence of inundation for short periods during the growing season may not have led to significant soil aeration. Macroinvertebrates were abundant in some accidental wetlands (e.g., site 8), but they should not inhibit development of hydric soil indicators for several reasons. By reducing the detritus size and increasing surface area, macroinvertebrates facilitate microbial action. The faster mineralization of nutrients should also yield greater primary productivity and organic matter accumulation.

Thus, sedimentation appears to play an important role in accidental wetland formation and development. Sediment deposition can influence hydrology by creating an impervious layer that establishes inundation. Sediment provides a substrate for seeds of hydrophytes and a plant growing medium. If plant growth provides adequate litter following senescence, there will be sufficient substrate to support microbial metabolism and adequate water holding capacity to limit diffusion of oxygen. In addition, it is the supply of iron with sediment that supports color development and provides an indication of oxidation and reduction processes.

Other Hydric Soil Indicators

Oxidized rhizospheres were found to be "prevalent and bright" in most accidental wetlands. The ability of hydrophytes to aerate the soil adjacent to roots has been described for many species (Conway 1940, Bartlett 1961, Armstrong and Boatman 1967). The two communities found in accidental wetlands in this study were identified based on the dominant species. The obligate wetland community was dominated by species which occur in wetlands >99% of the time, and the facultative wetland community is dominated by species that occur in wetlands 67-99% of the time (Reed 1988).

Oxidized rhizospheres were missing under three distinct conditions observed in this study. First, oxidized rhizospheres were either lacking, infrequent, or dull in the six newly constructed wetlands. Oxidation of iron adjacent to roots of hydrophytes appears to require time to become visible. Bartlett (1961) extracted reduced iron from soil to show that some iron oxidizes around

the surface of roots during a single growing season, but did not address the establishment of a visible layer of oxidized iron around roots. Second, oxidized rhizospheres were absent or dull in obligate wetland communities that exhibited a permanently flooded water regime. Oxygen may have been released by roots more slowly at these greater depths, and, since oxygen and redox potential decreases with depth, oxygen may be consumed more quickly under these conditions. Third, oxidized rhizospheres were absent at the facultative wetland community in site 12. This community exhibited high organic matter content and negligible mineral material; thus, iron was probably not available for oxidation.

There was no evidence of iron depletions and masses in this study. These features either have not had sufficient time to form, or conditions are not conducive to mottle development. Color patterns associated with iron depletions and masses become established where water levels fluctuate (Ponnamperuma 1972). The absence of these color patterns may, therefore, be a result of stable water tables in the small depressions, modulated by litter content and slow infiltration rates.

Soil temperature was not measured in this study, but several years of temperature data were obtained from the local Climatological Observer in Wise, VA. Mean monthly maximum temperatures ranged from a low in February of 7°F to a high in July of 27°F). Since water depth in the accidental wetlands ranged from 8 to 89 cm, light penetration to the bottom seems likely. However, vegetative cover of *Typha latifolia* was extensive in many obligate wetland communities, and shading could reduce soil temperatures. It appears that the 12 accidental wetlands were in similar geographic and topographic positions and would experience similar soil temperatures; however, future studies should monitor soil temperature.

Conclusion

Wetland construction has increased in the United States following implementation of Section 404 of the Clean Water Act (1977). However, most construction attempts have occurred in the last 10 years, and monitoring has been limited. Accidental wetlands provide an age class from 10 to 30 years post formation and provide clues to depressional wetland ontogeny. In particular, redoximorphic features, including low chroma matrices and oxidized rhizospheres, were formed within this time period. Newly constructed wetlands showed that certain hydric soil indicators, i.e., oxidized rhizospheres, can form within 3 years.

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ACIDIC MINE DRAINAGE ABATEMENT IN AN ANAEROBIC SUB-SURFACE FLOW WETLAND ENVIRONMENT - CASE HISTORY OF THE TREATMENT SYSTEM AT CORSICA, PA[1]

by
Jonathan M. Dietz[2]
and
Dennis M. Stidinger, P.E.[3]

Abstract. Wetland Treatment Systems (WTS) have been constructed over the past decade for the treatment of Acidic Mine Drainage (AMD). Potential benefits of anaerobic sub-surface flow treatment were initially identified from attempts to improve effectiveness of a surface flow wetland at the Jennings Environmental Education Center in western Pennsylvania (Dietz and Stidinger 1993) which, although not completely successful, resulted in acidity removal rates two to four times greater than pre-modification rates. This study was conducted to investigate the potential benefits of sub-surface flow design, which utilizes a sub-surface collection system, in comparison to conventional surface flow design. Two field-scale WTS, each containing anaerobic surface flow and sub-surface flow treatment cells (approximately 100 m^2 each), were constructed along an existing AMD discharge, near Corsica, Pennsylvania, with a 3-4 pH, 250-400 mg/L (as $CaCO_3$) acidity, 20-40 mg/L iron, 15-40 mg/L manganese and 10-30 mg/L aluminum. Sampling of the WTS included two elements: a weekly monitoring program from March 1992 through November 1992 to collect discharge water quality data; and a post-monitoring pore water sampling program conducted on December 8, 1992. Sub-surface flow design was found to provide statistically greater acidity removal with an average rate of 61.8 grams per day per square meter (GDM), in comparison to surface flow which had an average rate of 4.78 GDM, and provided effluent alkalinities greater than 100 mg/L (as $CaCO_3$) and pH greater than 6.5 at the flows and loadings evaluated. Removal rates for iron and aluminum removal of 3.02 and 2.48 GDM in the sub-surface flow design were also significantly greater than surface flow design which had removal rates of 1.89 and 0.20 GDM, respectively. Manganese removal was ineffective in both surface and sub-surface flow designs. Based on pore water results of two sub-surface flow units receiving different AMD flow and loading, an acidity removal rate of 25 GDM is recommended for future wetland treatment systems constructed with the sub-surface flow design evaluated in this study.

Key Words: Acidic mine drainage, passive treatment, wetland treatment, water quality, remediation

Key Phrases: Acidic Mine Drainage remediation utilizing passive (wetland) treatment systems.

Introduction

Wetland Treatment Systems (WTS) have been constructed over the past decade for the treatment of Acidic Mine Drainage (AMD). These WTS have achieved variable and at times inconsistent success at improving and meeting established effluent limits. The variable success of WTS is attributable to differences in design, size and treatment material. Although treatment success is variable, all constructed WTS have, at a minimum, been successful in reducing chemical costs associated with AMD treatment and pollutant loading to receiving streams.

Recent investigations by Hedin *et al.* (1991) have proposed design (sizing) criteria for surface flow compost WTS based on removal rates for iron, manganese and aluminum. In a study by Dietz and Stidinger (1993) on constructed surface flow wetlands, the removal of acidity was correlated with the removal of iron and aluminum, but

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did not find any relationship between acidity removal and manganese removal. Based on these relationships and the consistency of acidity removal in the different wetlands, Dietz and Stidinger (1993) recommended the use of acidity as a design criteria.

Potential benefits of a new WTS design were originally identified at a WTS constructed at the Jennings Environmental Education Center (Dietz and Stidinger 1993). The system was originally constructed as a surface flow system; however, due to poor performance, the WTS was modified by placing limestone and drainline in trenches, to promote greater subsurface flow. The modifications decreased acidity, iron and aluminum and resulted in acidity removal rates 2 to 4 times greater than surface flow WTS.

The following is a summary of several components of the study "Technical Investigation of the Abatement of Acid Mine Drainage in a Sub-surface Wetland Environment" (Dietz *et al.* 1993). This study was to investigate the potential benefits of this WTS design, which utilizes a sub-surface collection system, in comparison to conventional surface flow WTS design and position (i.e., before or after) of surface flow when used in combination with the sub-surface flow systems.

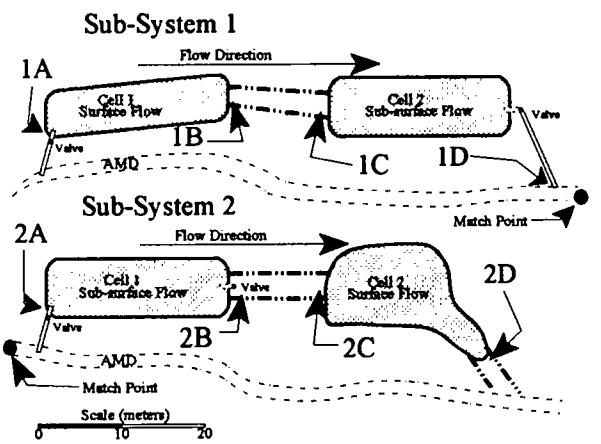

Figure 1. Plan view for the AMD wetland treatment system near Corsica, PA.

Wetland Design

The experimental system, designed to compare surface flow to sub-surface flow design and the two types of designs when used in combination, contained two wetland treatment systems, each containing a sub-surface and surface flow treatment cell. The order in the first treatment system (Sub-system 1) was surface flow followed by sub-surface flow. The order was reversed in the second treatment system (Sub-system 2) to permit evaluation of the two design combinations. Plan view of the two systems and cross-sections of surface flow and sub-surface flow units are contained on Figure 1 and 2.

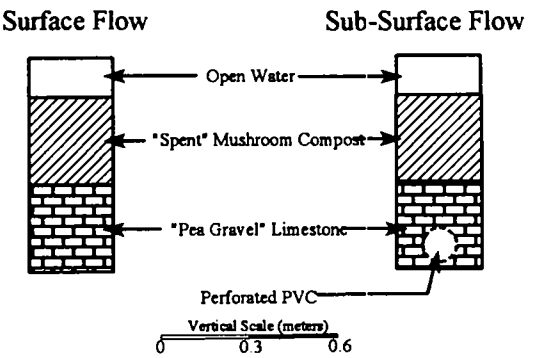

Figure 2. Typical cross-sections of units at the wetland treatment system near Corsica, PA.

The physical parameters of the treatment units consisted of at least 0.5 m of freeboard, a minimum slope of a 1:1 ratio for the inside and outside slopes, and 0.6 m berm width. The cell bottoms and banks of the treatment units were lined with clay found on-site during the excavation process. The surface area of Sub-system 1 totaled 303 m^2 with Cell 1 containing 113 ^{2}m and Cell 2 containing 130 m^2. The surface area of Sub-system 2 totaled 318 m^2 with Cell 1 containing 116 ^{2}m and Cell 2 containing 176 m^2.

After installation of the underdrain collection system in the sub-surface flow cells, "pea gravel" limestone purchased from Central Valley Aggregates was placed in each cell to a depth of 30 cm. The stone was covered with "spent" mushroom compost, obtained from Moonlight Mushrooms (Indiana, PA), to a depth of 30 cm. Once the compost was in place, the cells were allowed to fill and all areas disturbed by the construction activities were mulched and seeded with a grass/legume mix. After a two week waiting period the cells were planted by placing dug plant units, consisting of one mature cattail (*Typha latifolia*) and roughly a 30 cm

diameter of soil 25 cm thick, on approximately 0.6 m centers. Planting was completed in June of 1991.

Each sub-system received AMD from the same source and at the same flow rate of 11.4 L/min, which were controlled by intake gate valves. In addition to influent flow, gate valves and/or pipe weirs were located between treatment cells and at discharge locations to control and monitor flow through the WTS.

Sampling Program

The sampling program included several elements: a weekly monitoring program to collect discharge water quality data; and a pore water sampling program. The later program was conducted to evaluate wetland processes and functions. The different programs are discussed below.

Weekly Monitoring

For the weekly monitoring program, sampling stations were located in the two sub-systems to collect influent AMD, effluent from each sub-system and intermediate points between individual treatment units. Sampling stations, 1A through 2D on Figure 1, were located to sample influent, discharge, and intermediate locations in each sub-system at weirs and gate valves.

Sampling of the wetland was initiated in May of 1991; however, the severe drought during the early spring and summer of 1991 resulted in a loss of AMD flow in August. Flow did not return until early March of 1992 and normal water levels and sufficient AMD flow did not return until late March 1992. The weekly sampling was re-instated on March 25, 1992, continued through the spring, summer and fall of 1992 and was concluded on November 19, 1992. A total of 35 weekly samples were collected during the second year of the program.

Field measurements for temperature and flow were determined weekly at each sampling station. Temperature was measured with a YSI Model 51B Dissolved Oxygen meter. Estimates of flow were performed using a calibrated bucket (in liters) and a stop watch.

Water samples for laboratory analysis were collected weekly at each station, except influent stations in each sub-system, which were collected on alternating weeks. Samples were collected in 500 mL polyethylene bottles and, due to close proximity of the project (approximately 15 min.), transported to the laboratory in a cooler at the collected temperature. Water samples were analyzed for a number of parameters including: pH by electrometric method using an Orion (Ross) combination electrode; conductivity using a Markeson Model 10 and electrode cell; alkalinity and acidity by the potentiometric titration (H_2O_2) method; sulfate by the turbidimetric method; iron by the phenanthroline method; aluminum by the eriochrome cyanine R method; manganese by the periodate oxidation method; and calcium and magnesium (hardness), directly and indirectly, using the EDTA titrimetric method (APHA 1989 and HACH 1990).

Pore Water Monitoring

Pore water sampling was conducted on December 8, 1992 upon the completion of the weekly monitoring program. Pore water was extracted at the influent and effluent of each treatment pond at various depths from the compost layer; this was dependent on the depth of compost found at each sampling location. In general, pore water was extracted just below the surface, at 15 cm and 30 cm at each location. Cell 2 of Sub-system 2 contained only 15 cm of compost and could only be sampled at the surface and 15 cm depth.

Pore water samples were collected using a negative pressure apparatus that consisted of an 125 mL erlenmeyer flask, a hand held vacuum pump, and a variety of tubing, connectors and rubber stoppers. Collected pore water samples were numbered according to station and stored in air tight containers and transported to the laboratory. The samples were immediately analyzed (within 12 hours of collection) for Eh, pH, ferrous iron and sulfide. Eh was measured with an Orion combination redox electrode in a closed cell. Sulfide was measured on extracted water by the methylene blue method (APHA 1989). The remaining parameters, pH, alkalinity, acidity, total iron, ferrous iron, sulfate and conductivity were analyzed in accordance with procedures previously identified.

Results and Discussion

The results of the research project are summarized below. As reported earlier, a period of no flow

Table 1. Summary of averages and standard errors (in parenthesis) for water quality parameters of special interest monitored at the Corsica, PA wetland treatment system (n=35).

Sample Number	Lab pH	Alkal. mg/L	Acidity mg/L	Total Fe mg/L	Total Mn mg/L	Total Al mg/L	SO_4^{-2} mg/L	Ca mg/L	Mg mg/L
Sub-system 1									
Influent	3.41 (0.0002)	0 (0)	375 (96.8)	34.4 (12.7)	33.6 (6.80)	19.9 (8.55)	1579 (292)	315 (47.)	148 (33.)
Cell 1 Effluent	3.26 (0.0002)	0 (0)	340 (84.7)	20.1 (8.75)	31.8 (6.75)	18.4 (7.81)	1481 (294)	314 (61.)	144 (36.)
Cell 2 Influent	3.27 (0.0002)	0 (0)	232 (60.8)	15.8 (5.61)	32.4 (5.18)	10.8 (4.19)	1458 (285)	320 (47.)	134 (32.)
Cell 2 Effluent	6.76 (1e-7)	112 (21.0)	0 (0)	7.1 (5.44)	33.0 (4.29)	0.06 (0.03)	1351 (260)	377 (61.)	134 (35.)
Sub-system 2									
Influent	3.41 (0.0002)	0 (0)	375 (96.8)	34.4 (12.7)	33.6 (6.80)	19.9 (8.55)	1579 (292)	315 (47.)	148 (33.)
Cell 1 Effluent	6.62 (1.e-7)	112.3 (14.2)	0 (0)	11.13 (7.72)	38.21 (7.76)	0.055 (0.03)	1555 (288)	416 (67.)	160 (37.)
Cell 2 Influent	6.69 (1.e-7)	103.3 (13.7)	0 (0)	6.94 (6.04)	37.37 (7.62)	0.034 (0.02)	1537 (259)	422 (65.)	155 (34.)
Cell 2 Effluent	6.87 (8.e-8)	98.1 (16.5)	0 (0)	0.511 (0.32)	33.72 (5.93)	0.026 (0.02)	1496 (311)	408 (73.)	154 (40.)

occurred during the drought of 1991, which resulted in lowered water levels in several of the treatment units and caused air exposure to the mushroom compost and limestone layers. Therefore, only the data collected in the second year is included.

Effluent Water Quality

Discharge water quality is an important consideration in evaluating WTS performance and the ability of the treatment system to meet required effluent goals and regulatory limits (e.g., pH, iron, aluminum and manganese). The averages of the 35 samples collected for each of the monitoring stations for key parameters included in the sampling program are summarized in Table 1. To evaluate performance, effluent water quality from sub-systems and cells within sub-systems were statistically compared to influent water quality using an ANOVA procedure (Sokal and Rohlf 1981). A probability (p) of 0.05 was selected as the criteria for significant differences.

Temperatures monitored at stations within the sub-system averaged 13°C (55°F) during the monitoring program and varied over the course of the study from 4 to 26°C (40 to 79°F) in the treatment units. No significant effect of temperature was observed on discharge quality of any monitored parameter.

Three parameters that are closely related, pH, acidity and alkalinity, were similarly effected by the sub-systems. The pH in both Sub-systems increased significantly from influent pH of approximately 3 to effluent pH averages greater than 6.5, which is an almost complete removal of hydrogen ions by both sub-systems. The observed pH increases were associated with the sub-surface flow units, Cell 2 in Sub-system 1 and Cell 1 in Sub-system 2. Acidity followed a similar pattern as pH, significantly decreasing from influent concentrations of greater than 350 mg/L (as $CaCO_3$) to effluent aciditities of 0 mg/L in both sub-systems, with the majority of acidity decreases occurring in the sub-surface flow units. The remaining parameter, alkalinity, significantly increased across both sub-systems from 0 mg/L to approximately 100 mg/L (as $CaCO_3$) effluent alkalinity which entirely occurred in the sub-surface flow units of both sub-systems. Slight decreases

in alkalinity were observed across Cell 2, the surface flow unit in Sub-system 2, however, the decreases were not significant.

Two of three permit metals, iron and aluminum, decreased significantly across both sub-systems. Manganese, the third metal, did not significantly decrease across either sub-system; although slight increases and decreases of less than 2 mg/L were observed across individual units. Total iron decreased to approximately 7 mg/L in Sub-system 1 and less than 1 mg/L in Sub-system 2. With respect to individual treatment performance, total iron removal was similar across both units in each sub-system, ranging between 20 and 40 percent. A slight increasing trend in total iron was observed in effluent samples from sub-surface flow cells over the 35 weeks of the study, from initial concentrations of less than 5 mg/L to slightly greater than 10 mg/L. Total aluminum levels were found to decrease in both Sub-system 1 and 2 by almost 100 percent to effluent concentrations of less than 0.5 mg/L. Decreases in aluminum occurred in all treatment units, but predominately occurred in the sub-surface flow units and were directly related to the pH increases observed these units which is likely the result of aluminum pH dependent solubility (Stumm and Morgan 1981).

Calcium and magnesium were two metals monitored in the study to evaluate wetland processes. Calcium concentrations significantly increased across both sub-systesms by between 50 and 100 mg/L. The increases in calcium were associated with the sub-surface flow units in both sub-systems; no calcium concentration increases were observed across surface flow units. The observed calcium increases are likely the result of limestone dissolution in the sub-surface flow units, contributing at least 60 percent of the observed alkalinity increases from each sub-system. This alkalinity is important for the neutralization of acidity and hydrolysis and precipitation of metals. Conversely, magnesium concentrations were not significantly affected across either sub-system. Slight decreases of less than 15 mg/L were observed across Sub-system 1 and slight increases of less than 10 mg/L were observed across Sub-system 2, however, neither were significant. The magnesium data provide additional support that the increases in calcium were in fact due to limestone dissolution and not from some other process, such as evaporation.

Sulfate, monitored to evaluate processes within the wetland treatment system, was found to significantly decrease in Sub-system 1 by greater than 200 mg/L. Slight decreases of approximately 80 mg/L in Sub-system 2 were not significant. Sulfate exhibited decreases in both Cell 1 and Cell 2 in Sub-system 1 averaging decreases of approximately 100 mg/L in each cell. Seasonal increases and decreases in effluent sulfate concentrations were apparent in both sub-systems, however, influent sulfate data appeared to follow a similar seasonal change. The sulfate removal observed in the treatment units may reflect acidity removal processes (e.g., biological sulfate reduction) in the treatment systems which can be important for metal sulfide precipitation and alkalinity generation.

The effluent results indicate that the passive treatment systems remediate the majority of pollutants in the AMD. With respect to compliance levels associated with the mining industry, both sub-systems adequately reduced aluminum and raised pH to compliance levels. Only Sub-system 2, the sub-surface flow followed by surface flow configuration, adequately removed iron to levels below permit effluent limits. Neither sub-system lowered manganese which suggests additional treatment for manganese removal would be required to achieve compliance.

Removal Rates

Influent and effluent water quality data from each sub-system were used to determine loadings, removal rates and fluxes for each sub-system and units within each sub-system for iron, aluminum and acidity parameters. These three parameters were of interest since they were all significantly removed by the treatment systems. Loading rates, in grams per day (GPD) were calculated using influent concentration and flow data collected at each cell. Removal rates, also in GPD, were calculated using influent and effluent flow and concentration data. Average removal rates were adjusted to unit area rates or fluxes, in grams per day per square meter (GDM), using the surface areas of each unit.

Loading and removal rates for the sub-systems and individual units are contained in Table 2. The loading rates to the two sub-systems and surface flow unit, Cell 1 in Sub-system 1, and sub-surface flow unit, Cell 1 in Sub-system

2, were similar for all parameters, differing by less than 4 percent. Removal rates for each of the units were much more variable, differing by greater than 100 percent. A summary for each parameter is contained below.

Table 2. Average loading, removal, and flux rates for the Corsica, PA wetland treatment system.

Unit	Factor	Loading GPD	Removal Rate GPD	Flux Rate GDM
Sub-System 1				
Cell 1	Fe	528	214	1.89
	Al	295	22.5	0.20
	Acidity	5740	540	4.78
Cell 2	Fe	148	79.7	0.61
	Al	97.9	97.4	0.75
	Acidity	2170	3210	24.7
Total	Fe	528	338	1.12
	Al	295	187	0.62
	Acidity	5740	4760	15.7
Sub-System 2				
Cell 1	Fe	515	351	3.02
	Al	289	288	2.48
	Acidity	5530	7170	61.8
Cell 2	Fe	75.4	70.7	0.4
	Al	0.32	0.093	0.01
	Acidity	-1000	-42.7	-0.2
Total	Fe	515	465	1.46
	Al	289	288	0.91
	Acidity	5530	7030	22.1

Removal rates for acidity which reflect the addition of alkalinity to the effluent for Sub-system 2 were greater than Sub-system 1. The differences between sub-systems were not the result of differences in sub-system performance, but due to an uncontrollale loss of flow between Cell 1 and 2 of Sub-system 1, resulting in lower acidity loading and removal potential in the sub-surface flow unit (Cell 2). Removal rates adjusted for treatment area or fluxes for the sub-systems were similar, approximately 20 GDM, however, individual treatment unit fluxes were much more variable ranging from 0 to 60 GDM. As was indicated by the concentration results, acidity removal was predominately associated with the two sub-surface units. Fluxes for Cell 1 in Sub-system 2 (sub-surface flow) was 61 GDM, which was almost three times greater than the flux of 25 GDM for Cell 2 in Sub-system 1; however, the later unit received less than half the loading (2170 GPD vs. 6630 GPD). The surface flow unit from Sub-system 1 had a flux of 4.8 GDM which is similar to values obtained by other investigators (Dietz and Stidinger 1993 and Hedin *et al.* 1994) for anaerobic surface flow design. The removal rate for Cell 2 in Sub-system 2 is of limited value due to the absence of influent acidity.

Total iron removal rates were slightly greater in Sub-system 2, by slightly more than 100 GPD, than removed in Sub-system 1 which corresponds to slightly greater effluent iron measured from this sub-system (see Table 1). Total iron fluxes for Sub-system 2 were also slightly greater than Sub-system 1, 1.12 GDM versus 1.46 GDM, respectively. Fluxes were greatest in the first treatment units of both sub-systems and were coupled with the highest iron loadings. This relationship between iron removal and iron loading was alsc observed by Dietz and Stidinger (1993) in surface flow wetlands receiving various iron loadings.

Effluent total aluminum was essentially zero from both sub-systems, and differences in removal of aluminum, removal rates of 187 GPD for Sub-system 1 and 288 GPD for Sub-system 2, were due to previously discussed flow loss in Sub-system 1. Aluminum decreases across each sub-system were primarily associated with the sub-surface flow units, which had fluxes at least three times greater than the highest flux observed in the surface flow units, Cell 1 of Sub-system 1.

Pore Water Results

Pore water results from each treatment unit, average of influent and effluent samples, are summarized in Table 3. The pore water pH found in the compost of Cell 1, the surface flow unit in Sub-system 1, increased within the first several centimeters of compost depth from less than 4 to near 7. Cell 2, the sub-surface flow unit of Sub-system 1, did not have observable pore water pH increases until near the bottom of the compost layer which had

Table 3. Average of influent and effluent pore water results from the treatment units at the Corsica, PA wetland treatment system (n=2).

Cell	Depth cm	Eh mV	LAB pH	Alkal. mg/L	Acidity mg/L	Fe^{+2} mg/L	SO_4^{-2} mg/L	H_2S mg/L
Sub-System 1								
1	300	220	3.55	0	230	40.3	1000	0.84
	15	-275	6.45	460	0	16.7	1060	6.0
	30	-205	6.75	900	0	38.0	800	1.32
2	0	66.	4.41	2	244	7.7	785	0.84
	15	-168	4.49	2	236	10.5	1130	3.59
	30	-208	6.28	50	0	21.6	1250	0.78
Sub-System 2								
1	0	470	3.37	0	374	3.78	1350	0
	15	173	3.62	0	248	9.66	1300	0.55
	30	-55	4.20	0	332	15.0	1200	1.46
2	0	-67	6.53	159	0	25.5	1090	0.03
	15	-204	6.95	356	0	4.94	1190	3.21
	30	-197	7.11	826	0	5.46	955	0.67

pH levels near 6. Cell 1, the sub-surface flow unit of Sub-system 2, only had increases in pore water pH from slightly above 3 to approximately 4, less than a 1 unit pH increase. Similarly the increases in pore water pH in Cell 2 of Sub-system 1 were less than 1, however, surface water pH was well above 6 which did not allow for more substantial increases.

Pore water Eh, a measure of redox potential, decreased with depth in the compost in all units from values greater than 0 mV to less than -200 mV, except Cell 1 of Sub-system 2 which had decreases only to approximately -100 mV. The Eh decreases were fairly rapid occurring in the upper 15 cm of compost.

Acidity in pore waters tended to decrease with depth in the treatment units while alkalinity tended to increase with depth. Only pore waters from Cell 1 of Sub-system 2 contained acidity concentrations and no alkalinity at all depths. Surface flow units contained highest alkalinities in pore waters with maximum concentrations exceeding 800 mg/L. This is of particular interest for Cell 1 of Sub-system 1 since alkalinity was not measured in the effluent during the weekly monitoring program. Sub-surface flow units contained higher acidity concentrations and the lower alkalinity concentrations than surface flow units. This observed effect is likely the result of increased penetration of the AMD into the substrate, as a result of the underdrain systems, than would occur in surface flow design where diffusion across the water/compost barrier is relied upon.

Sulfate concentrations in the pore water of the two surface flow units were similar. Concentrations increased by less than 100 mg/L from surface concentrations to the 15 cm depth before decreasing to well below surface water concentrations at the 30 cm depth. Sulfate concentrations in the sub-surface flow units differed from each other and the surface flow units. Cell 2 of Sub-system 1 contained increasing concentrations of sulfate with depth, whereas pore water sulfate in Cell 1 of Sub-system 2 decreased by approximately 150 mg/L throughout the 30 cm compost depth. Sulfide

concentrations, the other sulfur form monitored, were detected at depths in the compost in all treatment units with maximum concentrations ranging between 2 and 8 mg/L. Maximum sulfide concentrations tended to occur at the middle depth of 15 cm.

The low Eh (less than -100 mV) and elevated pH (greater than 4) in pore waters collected from the treatment units indicate that conditions are conducive for biological sulfate reduction (Dietz 1989). Sulfate was found to decrease with respect to depth in several units which, in combination with the elevated sulfide concentrations, clearly indicates this process occurs in the units. The processes benefits in improving AMD quality are reflected in the higher pH and alkalinity observed in pore waters.

The differences in pore water data between surface flow and sub-surface flow design are apparent for all the parameters evaluated. Surface flow units contained higher pH, lower Eh, higher alkalinity and higher sulfide in pore waters than sub-surface flow units. The differences are likely the result of increased interaction of AMD with the compost layer in sub-surface flow units from directing the AMD through the compost with the underdrain system. The importance of this greater interaction is apparent in effluent results from the sub-surface flow units which had effluent higher in pH, lower in acidity, and higher in alkalinity than the surface flow unit.

Differences in sub-surface flow unit pore water Eh, pH and acidity were also observed. These differences are likely the result of differences in AMD loading to the two units, as identified in the previous section (see Table 2), where Cell 1 in Sub-system 2 received higher AMD loading than Cell 2 in Sub-system 1.

Sub-surface Flow vs. Surface Flow Comparison

A statistical analysis was conducted to compare water quality monitoring data from surface flow design (Sub-system 1 - Cell 1) and sub-surface flow design (Sub-system 2 - Cell 1) in order to evaluate the beneficial effects of the sub-surface flow treatment design over conventional surface flow design. The analysis consisted of comparing water quality monitoring data for individual parameters using an ANOVA procedure (Sokal and Rohlf 1981). Parameters included in this analysis were selected based on effluent requirements of wetland systems (i.e., pH, acidity, alkalinity, total iron, total manganese, and aluminum) and wetland processes (i.e., sulfate, calcium and magnesium).

The average pH discharged from the sub-surface flow design was significantly ($p<0.001$) higher, greater than 3 units, than the average pH discharged from the surface flow design. In comparison to surface flow, which discharged significantly ($p<0.001$) higher concentrations of acidity and no alkalinity, sub-surface flow discharged significantly ($p<0.001$) greater concentrations of alkalinity and no acidity. Iron and aluminum concentrations discharged by sub-surface flow were also significantly ($p<0.001$) lower than the surface flow units; iron and aluminum concentrations were approximately 10 mg/L and 20 mg/L lower, respectively. Conversely, manganese concentrations were significantly higher, approximately 6 mg/L, from sub-surface flow than surface flow. Calcium, analyzed to evaluate processes, was significantly different between the two designs. Calcium measured significantly ($p<0.001$) greater, approximately 100 mg/L, in the discharge from sub-surface flow than surface flow. The remaining two parameters, sulfate and magnesium, had averages that were slightly different between flow designs; however, the differences were not significant.

This comparison clearly indicates the anaerobic sub-surface flow design has superior AMD remediation capabilities than surface flow design for this type of AMD discharge. Water quality improvements were significantly better for all parameters evaluated except manganese, however, neither design afforded any measurable decreases in manganese concentrations. In addition, the superior effluent water quality results in higher removal rates for most of the parameters, except manganese, in sub-surface flow which would be reflected in reduced treatment area for sub-surface flow design. The improved effluent water quality from sub-surface flow may be, in part, the result of improved interaction of the AMD with the compost. In addition, the importance of higher calcium concentrations from sub-surface flow, which are the result of limestone solubilization, may be the result of improved contact of the AMD with the limestone layer.

Sub-System Comparison

A statistical analysis was also conducted, similar to the sub-surface flow and surface flow comparison, to compare water quality monitoring data from Sub-system 1 to Sub-system 2 to determine which configuration afforded better effluent water quality; see Table 1 for comparison of effluent means.

The average pH values discharged from the sub-systems were not significantly different. Both systems provided complete removal of acidity throughout the monitoring program. A slightly higher (approximately 14 mg/L) average alkalinity was discharged by Sub-system 1 in comparison to Sub-system 2; this was found to be significant ($p<0.005$). Sub-system 2 exhibited an average total iron of less than 1 mg/L, which was significantly ($p<0.001$) lower than the 7.1 mg/L average total iron exhibited by Sub-system 1. In addition, the average aluminum concentration from Sub-system 2 was significantly ($p<0.001$) lower than the level from Sub-system 1; however, in both sub-systems the average aluminum levels were well below 1 mg/L. The remaining metal analyzed, manganese, was not significantly different between the two sub-systems.

Sulfate, a process parameter, was significantly ($p<0.025$) different between the two sub-systems. On average, sulfate concentrations were approximately 150 mg/L lower in Sub-system 1 than in Sub-system 2. The remaining two parameters, calcium and magnesium, had averages that were slightly higher for Sub-system 2 than Sub-system 1; however, the differences were not significant.

The only effluent water quality parameter effected by placement of the sub-surface and surface flow units was iron. The Sub-system 2 configuration, sub-surface flow followed by surface flow, yielded iron concentrations significantly lower than Sub-sytem 1 and to levels necessary to meet compliance concentrations. This suggests sub-surface flow followed by surface flow will be the preferred configuration in a two cell treatment design.

Summary

Wetland design has been evolving over the past decade. Recent studies have begun to recommend sizing criteria that are intended to provide guidance in designing wetland systems (e.g., Hedin *et al.* 1994 and Dietz and Stidinger 1993), however, this design information has been limited to surface flow designs. The results from this research treatment project provided water quality and design information regarding sub-surface flow design. In addition, the study provided guidance with respect to placement of sub-surface flow and surface low treatment units when used in combination.

The water quality monitoring data clearly indicated that sub-surface flow design produces water quality that is superior to surface flow design for a number of parameters that include pH, iron, aluminum, acidity and alkalinity. Neither wetland design significantly removed manganese; however, manganese removal would not be expected in strongly reducing environments, as is found in compost wetland treatment systems, where solubility of manganese tends to increase (Stumm and Morgan 1981).

The water quality monitoring data also identified the appropriate alignment of a two cell system utilizing sub-surface and surface flow units. Water quality was similar between the two sub-systems for pH, aluminum, acidity and alkalinity. Effluent iron levels (approximately 0.5 mg/L) in Sub-system 2 were significantly lower than effluent iron levels (approximately 7 mg/L) in Sub-system 1 which indicates sub-surface flow followed by surface flow is the better alignment.

Processes in the wetlands important for remediating the high acidity in AMD were qualitatively evaluated. Alkalinity generating processes identified in the wetlands were sulfate reduction and limestone dissolution. Based on calcium increases in sub-surface flow units limestone dissolution accounted for greater than 60 percent of the alkalinity generated.

Acidity removal rates, identified by Dietz and Stidinger 1993 as the most reliable design parameter for determining size of wetland systems constructed for AMD treatment, were 24.7 GDM and 61.8 GDM in the two sub-surface flow cells. Lower pH, higher Eh, and higher acidities observed in pore water collected in the compost of the sub-surface flow cell in sub-system 2, which had a flux rate of 61.8 GDM, suggest the cell may have been overloaded. Therefore, a design criteria of 25 GDM for acidity removal, based on acidity flux measured in Cell

2 of Sub-system 1, is recommended for predicting size of future sub-surface flow wetland treatment systems.

The intent of the study was not to evaluate life expetancy of the sub-surface flow design. Life expetancy of sub-surface flow systems will likely be affected by a number of factors associated with the chemistry of the AMD, materials placed in the sub-surface flow system and design features (e.g., aerobic pre-treatment) intended to reduce excessive precipitation of metals in sub-surface flow systems. Long-term operational data from systems based on the above design guidance will be required to assess long-term effectiveness of the sub-surface flow design.

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CHEMICAL CHARACTERIZATION OF IRON OXIDE PRECIPITATES FROM WETLANDS CONSTRUCTED TO TREAT POLLUTED MINE DRAINAGE[1]

by

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Abstract. The passive treatment of abandoned mine drainage using wetlands will produce a significant amount of iron rich sludge which will require costly removal and disposal. An alternative to disposal may be the use of this iron oxide material as pigments which could defray some of these costs. In this research, iron deposits from five alkaline mine drainage wetlands were collected and a series of standard tests were run. The tests included loss on ignition, moisture, pH, acid soluble metals, oil absorption, and water soluble matter. The results of these tests were compared to those acheived using commercially available natural and synthetic iron oxides. The results indicate that iron oxides from constructed wetlands have chemical properties that are intermediate to those of natural and synthetic iron oxide products.

Additional Key Words: Acid mine drainage, constructed wetlands, iron oxides

Introduction

Passive methods of treatment for iron contaminated mine waters have been developed in recent years (Hedin et al. 1994). The principle technique used to treat this mine drainage is to pass the water through a constructed wetland, where the iron precipitates as iron oxide. Because passive methods are cost effective when compared to chemical alternatives (Brodie 1990), many private companies, public reclamation agencies, and non-profit conservation organizations are adopting this technology. Interest in passive techniques has been tempered, however, by concerns about the long-term costs of managing the iron oxide sludge that collects in the wetlands. It is anticipated that mine operators and reclamation groups will need to remove the metal-rich precipitates from passive systems every 5 to 20 years (the period depends on the design of the individual systems). Since the sludges are not classified as hazardous, they can be legally disposed of in an active mine backfill or in a public landfill. The costs associated with collecting and legally disposing of the iron oxide wastes, however, may be substantial.

The possible recovery and utilization of iron oxides from passive treatment systems has not been previously considered in detail. A significant iron oxide industry exists in the United States. In 1990, 123,000 mtons of iron oxides were

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produced for use as pigments in construction materials and coatings, as industrial catalysts, in foundry sands and in animal feeds. These iron oxides were produced from natural sources, by synthetic processes, and as a by-product of steel manufacturing. Natural iron oxides are produced by mining hemitite, geothite, or magnetite deposits. Synthetics are produced through the oxidation and neutralization of solutions containing ferrous or elemental iron, or as a by-product of analine production. Because synthetics are of higher purity than the natural iron oxides, they demand a higher price (Mauney 1994, Templeton 1990).

The purpose of this study was to evaluate precipitates from alkaline mine drainage wetlands as a potential source for iron oxide. The study focused on alkaline mine water (pH > 6.0) due to its faster precipitation rates, existing treatment sites, and potential for future wetlands construction. In Western Pennsylvania, constructed wetlands are currently being designed for several discharges that will precipitate more than 250,000 pounds of iron oxide per year. These wetlands will provide a large source of passively precipitated iron oxides.

Experimental

Sample Collection

Samples were collected during July 1995 from five alkaline mine drainage wetlands (Table 1) which were either natural or constructed wetlands used for the treatment of abandoned mine

Table 1:Descriptions of Sampling Sites

Site	Year	Builder	Design	Flow gal/min	Iron Conc. (mg/L) Influ.	Iron Conc. (mg/L) Efflu.	Substrate	Water Source	Coal Seam
Bubbler	1970	Unknown	Bore Hole	300	80	80	Wood	Deep Mine	Pgh.
Saint Vincent	1982	Boy Scout Troop	Natural	30	80	5	Soil	Deep Mine	Pgh.
Cedar Grove	1989	Natural Resource Conservation Service	Settling Pond 2-3 Wetlands LS Cell	41.2	92	41	Clay	Deep Mine	Pgh.
Howe Bridge	1991	Mill Creek Coalition	ALD Ditches 2 Ponds	34	265	185	Clay LS SMC	Abandoned Well	Clarion
Morrison	1990	Glacial Mining Company	ALD Ditch Settling Pond 2 Cells	1.7	151	56	Clay Manure	Surface Mine	Clarion
Tennessee	1992	Confidential Company	Pump Discharge Ditch	750	45	<1	Clay	Surface Mine	Sewanee

Pgh. - Pittsburgh
LS - Limestone
SMC - Spent Mushroom Compost
ALD - Anoxic Limestone Drain

drainage. Four of these sites (Saint Vincent, Cedar Grove, Howe Bridge, and Morrison) are located in western Pennsylvania. The Tennessee wetland is located near Chattanoga, TN. A sixth site (locally known as the Bubbler) was a wooden weir receiving mine drainage from a bore hole. It was chosen to determine the effect of precipitation in a non-vegetative area.

In each wetland site, three sampling areas were chosen: one close to the influent point of the wetland, one in the center of the wetland, and one near the effluent point of the wetland. A flow of water was present at each sampling point. Grab samples of the solid iron oxide precipitates were collected and placed into two gallon polyethylene bags. Two bags were collected at each sampling area. Each bag was filled with precipitate and water from one third to one half of its capacity. The Bubbler sample was taken directly from the first and second sections of the weir.

Commercial Iron Oxides

Samples of commercial iron oxides were obtained from the Bayer Corporation, (Pittsburgh, PA) and Hoover Color Corporation (Hiwasee, VA). The Bayer samples, Bayferrox 130M and Bayferrox 940, were synthetics produced by the analine process. The Hoover samples, 202RS Raw Sienna and 308BS Burnt Sienna, were natural iron oxides.

Sample Preparation

All samples were air dried in a greenhouse (maximum daytime temperatures 38° - 47° C) for 5-7 days. Once dried, the samples were crushed to a powdery consistency using a pestle. Most of the plant debris was removed by passing the samples through a 0.850 mm The sieved material was then further ground to pass through a 0.425 sieve. The fine material resulting from this sieving was placed in polyethylene bags and sealed for future use.

Analytical Methods

American Standard Test Methods (ASTM) procedures for pigments were done in duplicate to test the precipitates (Table 2) and the commercial iron oxide.

Table 2: ASTM Pigment Methods

Designation	**Title**	**Procedure**
D 1280-84	Standard Test Methods for Common Properties of Certain Pigments	4. Loss on Ignition and Ash 5. Matter Soluble in Water 6. Hydrogen IonConcentration*
D 280-81	Standard Test Method for Hygroscopic Moisture (and Other Matter Volatile Under the Test Conditions) in Pigments	Method A-For Pigments that do not Decompose at 110°C
D 1483-84	Standard Test Method for Oil Absorption of Pigments by Gardner-Coleman Method	
D 185-84	Standard Test Method for Coarse Particles in Pigments, Pastes, and Paints	Insoluble Dry Pigments, Except Metallic Aluminum and Bronze Powders

* Revisions in this method are noted in text.

Revisions were made to method D 1280-84 (Procedure 6 - Hydrogen Ion Concentration) upon consultation with an iron oxide manufacturer. The electrode used was an Orion Ag/AgCl Sure-Flow Electrode, Model number 9172BN. First, the cool sample solution was stirred with a glass stir rod sixty times. It was then allowed to settle for two minutes. After that time, the electrode was placed into the sample, stirred slightly and allowed to sit for an additional two minutes. The pH reading was taken after the two minute wait.

Metal Analysis

Sample Digestion. Each solid sample was digested by adding 100 ml of concentrated nitric acid (69.0-70.0% Baker Analyzed Nitric Acid) to a weighed portion of the precipitate in a 400 ml beaker and heated on a hot plate to induce boiling. The solution was reduced to 25 ml or less and filtered through Whatman 40 ashless filter paper to remove the acid insoluble material from the solution. The residue was then air dried and weighed to determine the percentage of acid insoluble found within the iron oxide sample. During filtering, the solution was transferred into a volumetric flask and diluted to 100 ml. After dilution, the flasks were sealed, labeled, and placed into a 3-5 °C refrigerator for storage.

Analysis. The sample digestates were analyzed for iron and twelve trace metals - cobalt, nickel, lead, aluminum, calcium, manganese, chromium, magnesium, sodium, zinc, copper and cadmium. The analysis was done using the inductively coupled plasma atomic emission spectrophotometer (ICP-AES) (Varian Liberty 200). All of the samples were analyzed in duplicate on the ICP. A standard curve for each element was determined at the beginning of each run using two multi-element standards.(Table 3) Since the ICP is linear over several orders of magnitude only two standards were necessary. The standard concentrations were determined from trial runs with representitive precipitated iron oxide. Wavelengths for each element were chosen to minimize interference from the high concentrations of iron.

Table 3:Elemental Wavelengths and concentrations of Standards

Element	Wavelength (nm)	Standards (ppm)
Co	345.350	1,10
Ni	352.454	1,10
Pb	368.348	1,10
Al	396.152	1,10
Ca	317.933	50,100
Mn	257.610	5,10
Fe	260.709	100,500
Cr	267.716	1,10
Mg	279.553	5,100
Na	588.995	5,10
Zn	213.856	1,10
Cu	324.754	1,10
Cd	228.802	1,10

A control check was also run for each set of samples. To analyze for iron, the samples were diluted 1 to 100.

Results and Discussion

Pigment Tests

The dried and sieved (0.425 mm sieve) precipitates were subjected to a variety of pigment tests similar to those used in the pigment industry to test the commercial iron oxide. The results from these tests on the wetland precipitate are shown in Table 4. The results from commercial iron oxide are given in Table 5. Loss on ignition for the wetland precipitates is in the same range as the yellow iron oxide (Bayferrox 940) and much higher than the red iron oxide (Bayferrox 130M). Since this test is in part an indication of the purity of the iron oxide, it is encouraging. The oil absorption test is also encouraging since this is an indication of the suitability of the precipitates pigments in paint applications. For the most part the wetland precipitates are very similar to the commercial iron oxides in terms of oil absorption.

Table 4: Pigment Tests for Wetland Precipitates

	Bubbler	Saint Vincent	Cedar Grove	Howe Bridge	Morrison	Tennessee
0.850 Sieve (% retained)		$3.16^{0.23}$	$11.56^{5.76}$	$20.08^{13.99}$	$17.14^{12.19}$	41.68^{nd}
Ignition Loss (% lost)	$13.74^{0.04*}$	$16.82^{2.13}$	$16.83^{0.35}$	$16.68^{0.31}$	$16.98^{0.17}$	$14.79^{0.95}$
Soluble Matter (%)	$0.80^{0.09}$	$1.00^{0.05}$	$0.95^{0.05}$	$1.00^{0.09}$	$0.36^{0.10}$	$0.65^{0.11}$
pH	$6.38^{0.04}$	$6.82^{0.06}$	$6.94^{0.07}$	$5.26^{0.13}$	$6.06^{0.42}$	$7.20^{0.11}$
Oil Absorp. (%)	$118.80^{2.88}$	$83.90^{7.91}$	$88.60^{2.14}$	$81.40^{10.47}$	$81.19^{3.63}$	$65.50^{10.24}$
Moisture (%)	$9.08^{1.76}$	$5.17^{0.54}$	$8.25^{1.65}$	$8.74^{4.12}$	$2.90^{0.40}$	$6.24^{3.72}$
0.045 mm sieve (% retained)	21.00^{nd}	$17.33^{6.87}$	$10.40^{10.07}$	$18.60^{16.68}$	$10.85^{11.04}$	$9.12^{5.31}$

Superscripts are 95% confidence limits

Table 5: Pigment Tests for Commercial Iron Oxide

	Bayferrox 130M	Bayferrox 940	Hoover 308BS	Hoover 202RS
Ignition Loss (% lost)*	0.6	13		
Soluble Matter (%)*	0.2	0.25		
pH	$6.47^{0.09+}$	$4.1^{0.02}$	$7.07^{0.12}$	$6.97^{0.02}$
Oil Absorption (%)	$37.84^{0.84}$	$89.4^{0.72}$	$64.1^{3.43}$	$45.9^{6.09}$
Moisture (%)*	0.4	0.25		
0.045 mm sieve*	0.002	0.04	0.1	0.05

*Taken from product literature
+Superscripts are 95% Confidence limits

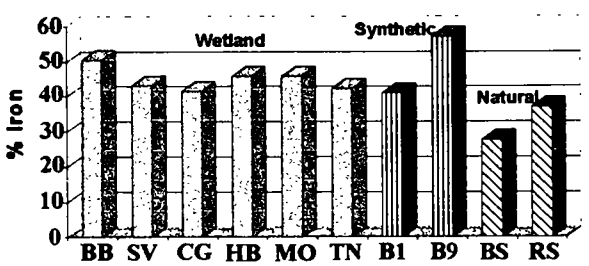
The meaning of the two sieve tests should be clarified. The 0.850 mm test is not a pigment test but a measurement of the miscellaneous material that is found in the wetland precipitates and is removed with a large size sieve. This material included leaves, grass and other biotic material. The 0.045 mm test is a critical test for the pigment industry since the particle size must be consistent to provide consistent blending by the customers. However, it should be noted that this is a test for final product iron oxide that is sold to the customer and the wetland precipitates were not processed except in a rudimentary fashion. Further processing (i.e. grinding) would certainly decrease the values for this test.

The final two tests (moisture and water soluble matter) also have much higher values for the wetland precipitates than the commercial iron oxide. Once again the results from these tests would change considerable with some simple processing such as rinsing the precipitates and then more thoroughly drying them.

Metals Analysis

The results of the metal analysis for the precipitates are shown in Table 6 and the commercial iron oxides in Table 7. The most important metal analysis is, of course, the percent iron in the precipitates. As can be seen in Figure 1, the percent iron is similar in the precipitates to the synthetic iron oxide and somewhat higher than the natural iron oxide.

Figure 1: Concentration of Iron in Precipitates. BB = Bubbler, SV = Saint Vincent, CG = Cedar Grove, HB = Howe Bridge, MO = Morrison, TN = Tennessee, B1 = Bayferrox 130M, B9 = Bayferrox 940, BS = Hoover 308BS, and RS = Hoover 202RS .



Results from four of the other metals analyzed are shown in Figure 2. For aluminium and manganese the concentrations are somewhat higher than the synthetic iron oxide but lower than the natural. The analysis for copper shows all the wetland precipitates to be lower than both the synthetic and natural iron oxides. The zinc results indicate some diversity in the wetlands with zinc concentrations from the Bubbler, Saint Vincent, Cedar Grove and Tennessee sites being lower than the concentrations in the commercial iron oxide. The values from the Howe Bridge site are close to the concentrations of the high zinc commercial products. The Morrison site is substantially higher in zinc concentration than any of the other samples. This illustrates that there is some diversity in the wetland precipitates and that critical commercial applications may require careful screening of the precipitates used.

Metals in the mine drainage are most likely the source of the trace metals in the precipitates. Therefore, mine drainage with high concentrations of metals would produce precipitates with high metal concentrations. The water soluble matter is probably associated with ions such as Na^+, SO_4^{2-}, and Cl^- present as water soluble salts. The concentration of sodium in the Bubbler, Saint Vincent, and Cedar Grove sites is much higher than at the other sites. The water soluble matter is also much higher at these sites. The Howe Bridge site does not fit into this pattern, however, with relatively low sodium concentrations and high water soluble matter.

Conclusion

In general the wetland precipitates have many similar characteristics to commercial iron

Table 6: Concentration of Metals in Wetland Precipitates

Metals	Bubbler	Saint Vincent	Cedar Grove
Co (ppm)	23.66 +/- 40.32	45.77 +/- 12.60	51.66 +/- 7.86
Ni (ppm)	20.71 +/- 2.70	16.86 +/- 1.94	22.70 +/- 25.05
Pb (ppm)	27.24 +/- 1.94	72.88 +/- 36.92	68.28 +/- 34.71
Al (ppm)	259.9 +/- 8.6	3322 +/- 1639	820.1 +/- 57.12
Ca (ppm)	3825 +/- 113	5310 +/- 664.1	5004 +/- 287.4
Mn (ppm)	328.4 +/- 11.4	735.2 +/- 386.7	144.1 +/- 37.43
Cr (ppm)	2.97 +/- 0.07	74.04 +/- 136.3	0.18 +/- 0.35
Mg (ppm)	347.6 +/- 11.1	728.6 +/- 216.1	359.2 +/- 25.6
Na (ppm)	422.1 +/- 4.6	1141 +/- 1182	1284 +/- 137
Zn (ppm)	125.0 +/- 5.1	93.44 +/- 9.06	69.71 +/- 5.05
Cu (ppm)	18.87 +/- 1.56	22.24 +/- 1.71	15.34 +/- 0.75
Cd (ppm)	< 0.10	<0.10	3.01 +/- 1.96
%Fe	50.45 +/- 0.08	43.11 +/- 6.44	41.41 +/- 2.28

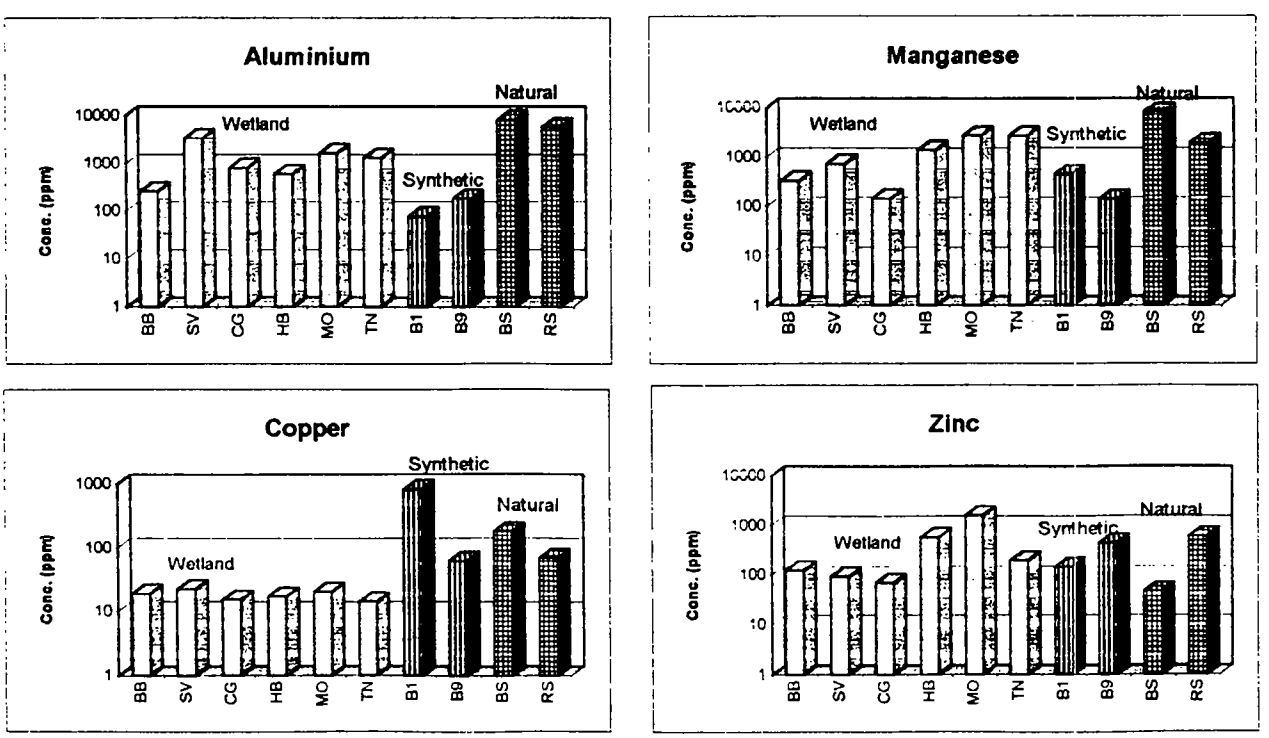
Metals	Howe Bridge	Morrison	Tennessee
Co (ppm)	72.13 +/- 20	276.4 +/- 28.0	108.3 +/- 9.5
Ni (ppm)	156.16 +/- 63.53	325.3 +/- 73.8	200.6 +/- 24.5
Pb (ppm)	86.97 +/- 35.85	84.76 +/- 34.86	17.93 +/- 20.71
Al (ppm)	600.0 +/- 180.6	1660 +/- 511	1321 +/- 894
Ca (ppm)	1060 +/- 31.20	1401 +/- 203	6038 +/- 1392
Mn (ppm)	1383 +/- 193	2716 +/- 278	2694 +/- 715
Cr (ppm)	<0.10	0.15 +/- 0.30	1.35 +/- 1.74
Mg (ppm)	343.8 +/- 50.20	355.9 +/- 42.4	586.8 +/- 164.5
Na (ppm)	73.88 +/- 17.24	88.17 +/- 9.80	40.95 +/- 5.50
Zn (ppm)	569.4 +/- 108.2	1570 +/- 184	195.6 +/- 45.9
Cu (ppm)	17.10 +/- 1.68	20.21 +/- 1.80	14.21 +/- 0.62
Cd (ppm)	0.05 +/- 0.097	<0.10	<0.10
%Fe	45.88 +/- 4.68	45.74 +/- 10.58	42.30 +/- 8.67

Table 7: Concentration of metals in commercial iron oxide

Metals	Bayferrox 130M	Bayferrox 940	Hoover 308BS	Hoover 202RS
Co (ppm)	43.62 +/- 15.25	42.29 +/- 6.06	62.21 +/- 2.01	45.48 +/- 6.74
Ni (ppm)	182.6 +/- 38.7	34.93 +/- 7.38	146.2 +/- 21.2	173.13 +/- 4.20
Pb (ppm)	34.50 +/- 1.89	45.61 +/- 1.64	73.39 +/- 15.8	69.17 +/- 22.43
Al (ppm)	77.21 +/- 18.46	183.7 +/- 16.9	8153 +/- 268	5757.8 +/- 94.7
Ca (ppm)	152.5 +/- 45.7	259.5 +/- 23.8	7512 +/- 210	1264.9 +/- 108.4
Mn (ppm)	450.6 +/- 89.0	143.1 +/- 10.7	8094 +/- 69	1958.4 +/- 57.6
Cr (ppm)	479.6 +/- 86.5	105.1 +/- 13.0	159.6 +/- 1.3	190.7 +/- 3.2
Mg (ppm)	20.07 +/- 4.47	76.08 +/- 5.74	10343 +/- 156	10577 +/- 303
Na (ppm)	160.4 +/- 15.4	622.1 +/- 23.7	393.5 +/- 1.5	264.4 +/- 1.0
Zn (ppm)	145.7 +/- 17.6	434.9 +/- 29.9	46.99 +/- 2.94	597.7 +/- 308.5
Cu (ppm)	826.6 +/- 73.6	61.69 +/- 3.60	177.9 +/- 2.1	69.09 +/- 0.10
Cd (ppm)	<0.10	0.50 +/- 0.35	68.52 +/- 0.36	1.58 +/- 2.79
%Fe	40.99 +/- 0.97	57.11 +/- 0.83	27.63 +/- 0.47	37.06 +/- 0.44

Figure 2: Selected Metals Concentrations in Wetland Precipitates and Commercial Iron Oxide.

Wetland: BB = Bubbler, SV = Saint Vincent, CG = Cedar Grove, HB = Howe Bridge, MO = Morrison, TN = Tennessee
Synthetic: B1 = Bayferrox 130M, B9 = Bayferrox 940
Natural: BS = Hoover 308BS, RS = Hoover 202RS



oxide. The iron concentrations, oil absorption, and ignition loss are all reasonably close to the commercial iron oxide. Several of the other parameters such as moisture, 0.045 mm sieve, and water soluble matter are significantly different. However, this difference may be due to a lack of processing such as grinding, washing, or vigorous drying. The inclusion of some of these processing steps may bring these values closer to the commercial products.

The results of the metal analysis are interesting in that they show the precipitates to be between the synthetic and the natural iron oxides in concentration for several metals. They are also significantly lower in copper and chromium concentration than both the natural and the synthetic material.

These results are encouraging and would indicate that the wetland precipitates do have some potential as a raw material for commercial iron oxide. There is some additional processing that will be necessary to make a consistent product, but additional evaluations are justified.

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THE BARK CAMP RUN DEMONSTRATION CONSTRUCTED WETLANDS: FINDINGS AND RECOMMENDATIONS FOR FUTURE DESIGN CRITERIA

by

William W. Hellier

Abstract The performance of six wetlands built to ameliorate acid mine drainage has been analyzed over 806 days. The wetlands are each 52 m x 8 m at the water surface, with a 5 cm surface water column and 60 cm depth of substrate consisting of horse manure mixed with an equal volume of SiO_2 stone. Each wetland has a 1 mm polyethylene liner and wooden deflectors every 2 m to promote uniform flow. The dominant macrophyte is *Typha latifolia*. Flows are ~10 m^3/d. Median influent pH = 3.0; acidity = 202 mg/L as $CaCO_3$; $[Fe^{+3}]$ = 22.9 mg/L; $[Fe^{+2}]$ = 0.8 mg/L; [Al] = 10.9 mg/L; [Mn] = 2.0 mg/L; and $[SO_4^{-2}]$ = 592 mg/L. Water quality was determined biweekly for each wetland's influent and final effluent and at 84 internal sampling stations built to sample at depths 5, 25, 45, and 60 cm below the water surface. Tracer studies showed that the mine drainage flows through both the surface water column and the substrate, and a one-dimensional flow model appears satisfactorily to explain the performance of the wetlands. Wetlands should be designed to achieve uniform flow and to maintain flow through the substrate A polishing area of a few m^2 with a substrate consisting only of limestone chips appears to be helpful in maintaining effluent alkalinity. Simple mathematical models were developed to assess the performance of the constructed wetlands. Removal of Fe^{+3} is by hydrolysis, followed by reduction to Fe^{+2} and incorporation into the substrate. Removal of Al^{+3} is by hydrolysis. The observed removal rates of 1g d^{-1} m^{-2} for Fe^{+3} and 0.5g d^{-1} m^{-2} for Al^{+3}, combine with concurrent neutralization of the resulting H^+, to give an acidity removal rate of 5.5g d^{-1} m^{-2}. A zero-order rate law can be used to describe the observed rate of Fe^{+3} removal First-order rate laws also produce good representations for the rates of Fe^{+3} and Al^{+3} removal. Effluent Mn concentration > influent [Mn], and declines in accordance with a first-order rate law, indicative of desorption of Mn^{+2} initially present in the substrate. This may explain past difficulties in assessing Mn removal in anaerobic constructed wetlands.

Additional Key Words: water pollution abatement, constructed wetlands design, mathematical modeling.

Introduction

The PA Department of Environmental Protection (DEP 1988) has reported that discharges from coal and clay mining operations account for pollution of approximately 2700 km of Pennsylvania's streams. At least 794 discharges are being treated by current coal operators to conform to effluent limits (Hellier, *et al.* 1994). A large number of discharges are from operations that were abandoned before current mining laws were passed; for example, the Muddy Run watershed receives 15.75 metric tons of acid per

[1]Paper presented at the 13th annual meeting of the American Society for Surface Mining and Reclamation, Knoxville TN, May 19-25, 1996.

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day from abandoned mines (Skelly and Loy 1972). To restore the quality of the polluted streams, a technologically feasible and cost-effective means must be found to abate the pollution from these discharges.

The United States Environmental Protection Agency (EPA 1982) has approved chemical treatment for mine discharges; however, because of the cost, passive treatment methods, including constructed wetlands, are also being used to treat mine discharges of different qualities. While there have been several studies and reports on the subject of constructed wetlands, and design guidance manuals have been published (Hedin *et al.* 1994), there is little theoretical discussion on the basis for the design of constructed wetlands.

Objectives

The objectives of this study were: (1.) To study the performance of wetlands built to take advantage of anaerobic processes in treating a mine discharge with moderate acidity and moderately elevated metals; (2.) To obtain transient and long term water quality data at key sample points in the wetlands.; (3.) To determine flow patterns within the wetlands; (4.) To assess the effects of moderate pretreatment of the wetland influents with anhydrous NH_3; (5.) To develop simple mathematical models from the data obtained, upon which future designs can be based; and (6.) To compare the design criteria produced by the mathematical models to previous design criteria, and to compare the performance of the demonstration wetlands to the performance of existing wetlands.

Site Selection

The site of the abandoned Bark Camp #1 and #2 deep mines and coal processing facility was selected for the study because: (1.) The discharge had a moderate acidity of < 300 mg/L as $CaCO_3$ and a moderate Fe concentration of < 30 mg/L, indicating a reasonable likelihood of successful treatment and that there was a reasonable chance to improve the quality of the degraded stream; (2.) The landowner, the Pennsylvania Bureau of Forestry, approved of the project and provided assistance; and (3.) It was within reasonable travel time of the office.

Bark Camp Run is located in Huston Township, Clearfield County, PA, and has been impacted by the abandoned mining operation. as shown by the following results obtained 9/4/90, before the study was initiated: pH = 5.5; acidity = 22 mg/L as $CaCO_3$ (alkalinity = 3 mg/L); $[Fe^{+3}]$ = 7.1 mg/L; [Al] =2.7 mg/L; [Mn] = 0.9 mg/L; $[SO_4^{-2}]$ = 186 mg/L; no fish are present.

Upstream of the study area, the respective parameters were: pH = 6.9; acidity = 0 mg/L as $CaCO_3$ (alkalinity = 18 mg/L); $[Fe^{+3}]$ = 0.2 mg/L; [Al] = 0.2 mg/L; [Mn] = 0.1 mg/L; $[SO_4^{-2}]$ = 161 mg/L, and around 50 trout/km were found. Bennetts Branch Sinnemahoning Creek, to which Bark Camp Run is tributary, is stocked with trout by the PA Fish and Boat Commission.

The Bark Camp Run watershed area is 726 ha, and the stream flow is on the order of 7000 m^3/day downstream of the study area. The combined mine discharges from the study area contribute nearly 700 m^3/day of this flow. During the study, the following median values were found for the discharge: pH = 3.0; acidity = 202 mg/L as $CaCO_3$; $[Fe^{+3}]$ = .22.9 mg/L; $[Fe^{+2}]$ = 0.8 mg/L; [Al] = 10.9 mg/L; [Mn] = 2.0 mg/L; and $[SO_4^{-2}]$ = 592 mg/L.

Constructed Wetlands Design

Six constructed wetlands were built, each with a 1mm polyethylene liner on the bottom and 2 (horizontal) :1 (vertical) in-slopes to prevent groundwater exchange. The wetlands were 52 m long and 8 m wide at the water surface. The design freeboard is 35 cm, surface water column depth 5 cm, and substrate depth 60 cm. The top 5 cm of substrate is topsoil, and the remaining 55 cm is a mixture that is 50% by volume horse manure and 50% by volume river gravel with approximately 5 cm

dimension and composed primarily of SiO_2. This rock, rather than limestone, was added to the substrate in order to assess the effects of allowing all the alkalinity to be generated through dissolution from the substrate and by biological reactions, without supplementation of the substrate with limestone. The manure itself had been freshly generated over the past winter and had a measured alkalinity of approximately 15000 mg $CaCO_3$ equivalent per kg dry weight. The substrate in the initial 2m length at the influent barrel consisted of the aforesaid stone alone, while the substrate in the last 2m immediately preceding the effluent barrel consistent of crushed limestone of about 1.5 cm dimension.

From three manhole/mixing chambers, influent was directed into pairs of wetlands via a barrel around whose bottom 15 cm periphery were drilled 6 mm holes. The effluent barrels were of the same design. Wooden deflectors were built every 2m along the wetlands' length to promote uniform flow. Sample stations were built to measure water quality at depths of 5 cm, 25 cm, 45 cm, and 60 cm after 128 m^2, 256 m^2, and 384 m^2 of wetland area for all six wetlands; also after 96 m^2 for wetland #2 and after 48 m^2 for wetlands #4 and #6. The dominant macrophyte was *Typha latifolia*. Figure 1 is a plan view of the wetlands. The areas given are the cumulative wetland areas from the inlet to the sample stations depicted.

Anhydrous ammonia was used for a one year period to adjust influent pH. Mixing chamber pH was adjusted in an effort to deliver an influent having pH = 3.0 to wetlands #1 and #2; pH = 4.0 to wetlands #3 and #4; and pH = 5.0 to wetlands #5 and #6.

Methods

Biweekly water quality samples were obtained upstream and downstream of the study area and from the final treatment basin, at the wetland influent and effluent, and at the 12 internal sample points of the odd numbered wetlands and 16 internal sample stations of the even numbered wetlands. Each sample was separated into two aliquots to facilitate laboratory processing; one aliquot to be analyzed for pH , alkalinity, acidity, and $[SO_4^{-2}]$ and the other to be analyzed for $[Fe^{+2}]$, total [Fe], [Mn], [Mg], and [Al]. The latter aliquot was acidified to pH < 2.0. Additionally, the effluents were analyzed for Cl^-, P, K, BOD, and forms of N. The samples were put into iced containers and transported to the Department's laboratory in Harrisburg, where they were analyzed following EPA approved methods (APHA 1992).

Influent flows were measured with a beaker and stop watch on most occasions when the water quality samples were taken. When influent flow rates were not measured, they were estimated by interpolation between measured values. Because influent flows remained very steady from measurement to measurement, the instantaneous measurement was considered to be representative. Effluent flow rates, while measured, proved less representative because they were more sensitive to daily evaporation and rainfall.

A tracer study was conducted to determine flow patterns in the wetlands (Donahue 1994). On June 9, 1993, 11.6 kg of NaBr was dissolved in water and poured into the influent barrels of wetlands #1 and #2 during one half hour. Over 68 days, samples from all of the two wetlands' sample points were analyzed at the Pennsylvania State University for $[Br^-]$, and breakthrough curves of $[Br^-]$ as a function of time were developed.

Results and Discussion

The performances of the six wetlands are illustrated and discussed in the report to EPA (Hellier 1996). Flows varied over time, but usually were less than 15 m^3/day. The effluent pH was usually near 6.0, contrasted with an influent pH of 3.0. The wetlands tended to impart alkalinity as shown by an effluent which was alkaline to neutral as contrasted with an influent with a median acidity of 202

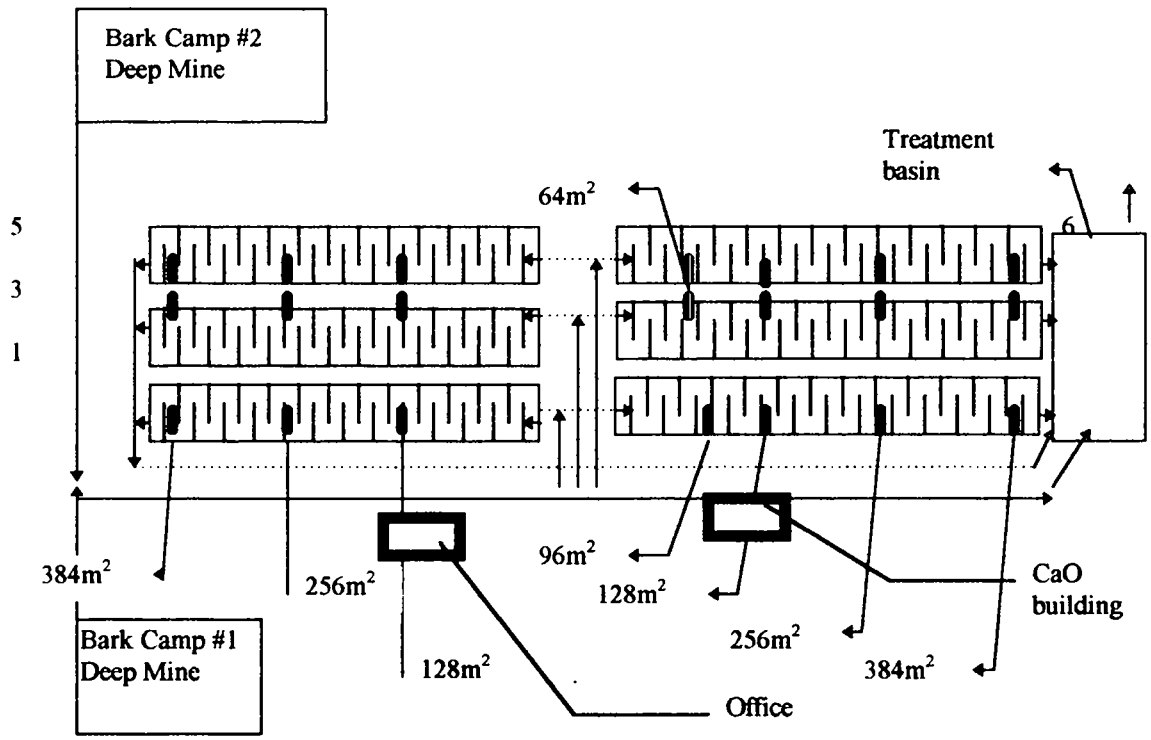

Figure 1.Plan View of Bark Camp Run Constructed Wetlands.

mg/L. Effluent [Fe^{+3}] tended to be about an order of magnitude lower than influent [Fe^{+3}], and most of the effluent Fe was Fe^{+2}. Total Fe removal was effective for the period from about 130 days to 400 days. After 400 days, Fe continued to be removed, although not as efficiently. The wetlands have been effective in removing Al, but effluent [Mn] has consistently exceeded influent [Mn]. For the purpose of illustration, the overall performance of wetland #1 is shown (Figure 2).

Graphs were made of the pollution indicators as a function of time at constant depth in the substrate for the different cumulative treatment areas, and at constant cumulative treatment area for the four different depths within the substrate. For the present discussion, Figure 3 presents the [Fe^{+3}] data for wetland #1. The results for the indicators are summarized as follows: (1) Net acidity tended to decline and pH to increase with increasing depth, although no regular pattern is apparent. Deeper parts of the substrate tended to maintain alkaline conditions better over time compared to the surface water column and the shallower parts of the substrate. (2) Total [Fe] tended to decline with depth, with [Fe^{+3}] being lowered relative to influent [Fe^{+3}] and [Fe^{+2}] being elevated; (3) [Al] tended to be lowered relative to influent concentrations, but (4) [Mn] was elevated relative to influent concentrations; (5) [SO_4^{-2}] fluctuated, and no apparent pattern was observed.

The tracer breakthrough curves at the four different depths at a cumulative treatment area of 128 m^2 are shown for wetland #1 (Figure 4). Based on the breakthrough curves, detention times were calculated. The flow characteristics of wetland #1 were assumed to apply to wetlands #3 and #5; the characteristics of wetland #2, to wetlands #4 and #6. To apply the results to a given water quality sampling period, detention time was assumed to be proportional to influent flow. Detention times behaved regularly

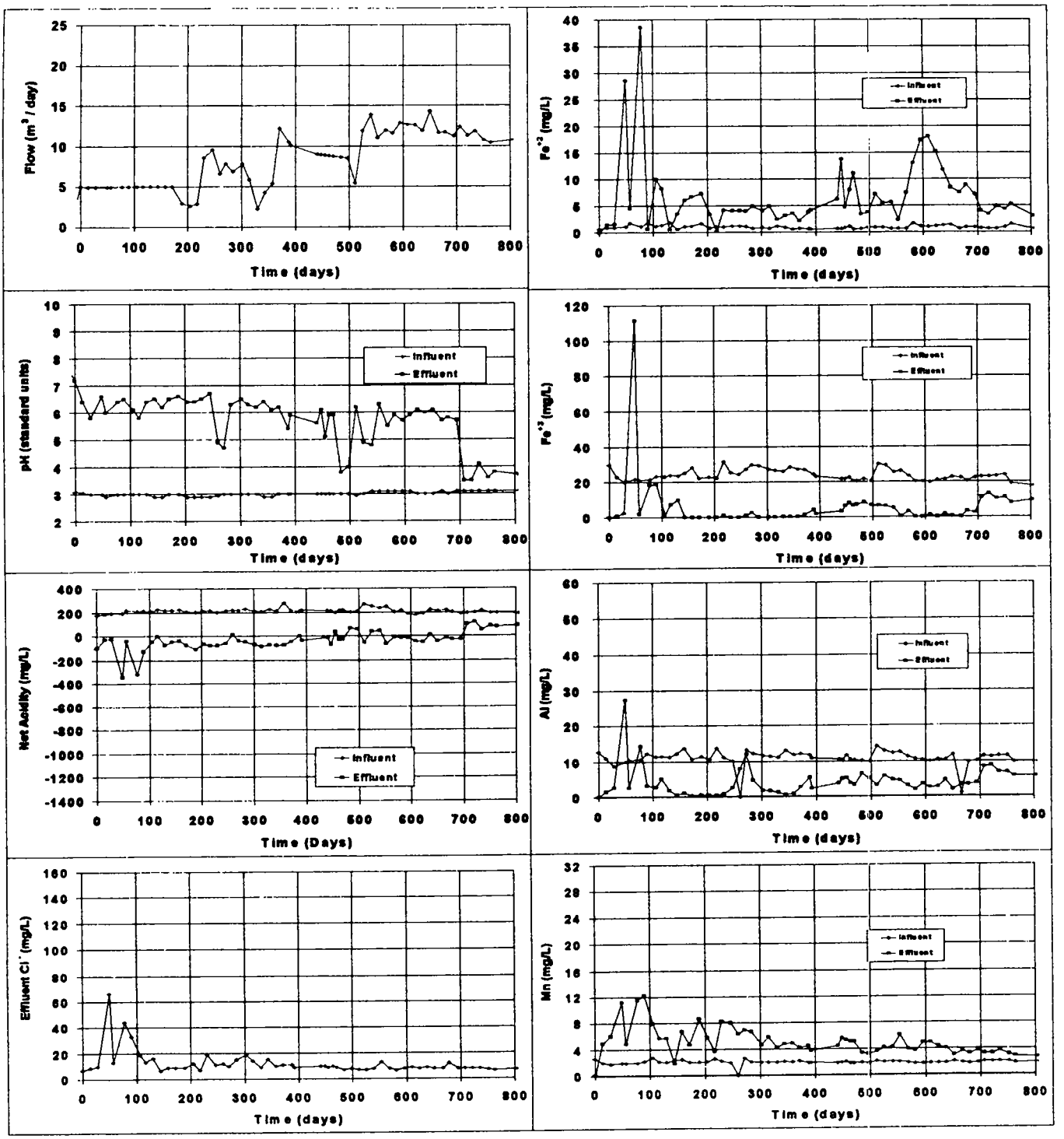

Figure 2. Overall Performance of Wetland #1.

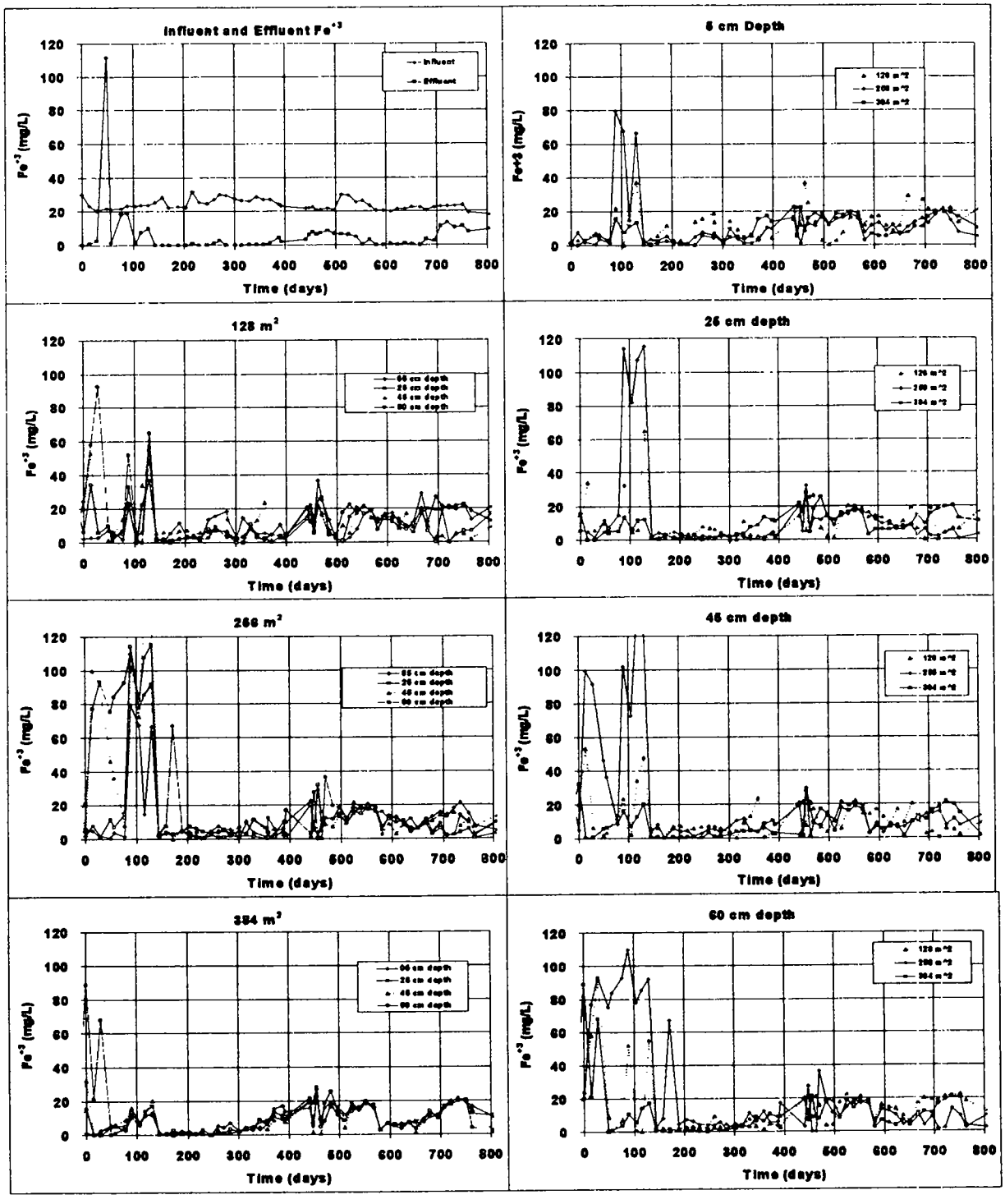

Figure 3. Wetland #1 Removal of Fe^{+3}.

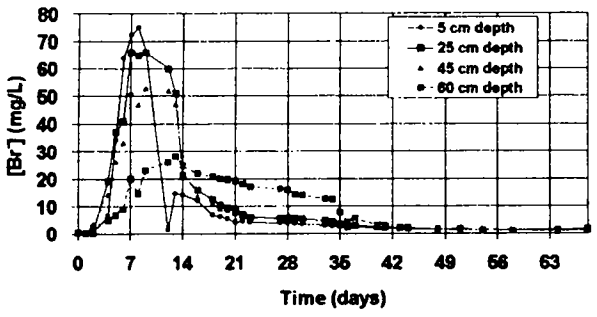

Figure 4: Wetland #1 [Br⁻] after 128 m² Area

and increased with increasing depth, suggesting the first approximation of a one-dimensional flow pattern in which flow was uniform and varied only in the vertical direction throughout most of the wetlands. The irregularities exhibited near the influent can be attributed to a drawdown effect resulting from the effluent being taken from the bottom of the effluent barrels.

The study period appears to be divided into three phases: (1.) A transient startup period in which the wetland acclimates to the mine drainage flow and reaches steady operation. During this phase, the effluent exhibits elevated [Cl^-] and [Fe], indicative of possible desorption and flushing from the substrate. The transient period is not identical for all pollutants; for example, [Mn] continues a downward trend. (2.) A steady period in which the wetlands generate alkalinity, remove acidity, and are very effective in removing Fe; (3.) A declining effectiveness period in which the wetlands begin to lose their alkalinity generation capabilities and Fe removal becomes less effective. During this period, the limestone chip polishing area appears to remove between 30% and 100% of the Fe^{+3}. The declining wetland effectiveness may be due to the loss of the alkaline material originally present in the horse manure. Both of these facts suggest that limestone, rather than inert stone (SiO_2), should be incorporated into the substrate to maintain the performance of the wetlands.

The anhydrous NH_3 addition accelerated the deposition of FeOOH at the entrance to the wetlands, as evidenced by the coloration of the substrate surface: wetlands #1 and #2 had no coloration; wetlands #3 and #4 had some coloration; and wetlands #5 and #6 had substantial coloration. Influent pH in all six wetlands returned almost instantaneously to 3.0 as the Fe^{+3} was hydrolyzed.

Mathematical models

The development of a comprehensive mathematical model to explain the behavior of all pollutant indicators is beyond the scope of this report. Model development will be illustrated by considering the removal of Fe in the first 128 m^2 of treatment area during the steady phase of the study.

The 22.90 mg/L median influent [Fe^{+3}] accounts for 96% of the influent Fe. Mechanisms of Fe^{+3} removal from the water include (1) Hydrolysis of Fe^{+3} followed by precipitation as FeOOH from the surface water column and retention as a solid on the surface of the substrate; (2) Hydrolysis of Fe^{+3} and retention of the resulting FeOOH within the substrate; (3) Reduction of Fe^{+3} to Fe^{+2} within the substrate, followed by its Fe^{+2} retention in the substrate as organically bound Fe^{+2}, FeS, or $FeCO_3$.

The following assumptions were made to develop a simple model: (1.) The transient startup period having passed, on any sample day during the steady period of operation, the influent flow and indicator concentrations are steady. (2.) Plug flow conditions apply in the surface water column and within the substrate. This assumption is justified by considering that the flow is low relative to the wetland volume, and that the deflectors and the *Typha latifolia* enhance uniform, plug flow. (3.) Flow components in the x direction (along the length of the wetland) are large relative to flow components in the y or z directions; (4.) To permit the use of differential equations, the water column and the substrate are

considered mathematically to be continuous media; (5.) The rate expression for a given indicator depends only on the concentration of that indicator.

Zero order model

Historically, expressions of rate for removal of pollutants in constructed wetlands have been simple linear representation based on the mass of pollutant removed per unit area per unit time; i.e., g day^{-1} m^{-2} (US Bureau of Mines, *op. cit.)*. If the wetland depth is constant, or if an average depth is used, this representation considers the removal rate *independent* of concentration:

$$d[Fe^{+3}]/dt = -k_0 \quad (1)$$

The zero-order rate constant k_0 is given by:

$$k_0=([Fe^{+3}]_{in}-[Fe^{+3}])/(\tau-\tau_{in}) \quad (2)$$

where the subscript "in" stands for conditions at the beginning of the wetland section being considered, and τ stands for detention time as determined from the tracer breakthrough time (Figure 4) and influent flow rate. For each wetland, the rate constant k_0 was determined between sampling stations at the different depths, (Figure 3), removal rates were calculated at those depths, and the composite rate of removal in each section of the wetland was calculated by averaging among the rates at the four depths. The rate of Fe^{+3} removal, g day^{-1} m^2 is shown in Table 1. The removal rate for the first 128 m^2 of the wetlands is consistent with removal rates that DEP has observed in other wetlands receiving acidic influents. However, the calculated removal rates in this first one-third of the wetland is sensitive to the position within the wetland. This suggests, particularly for the first 128 m^2 of the wetland, that a different rate expression might be more valid. The calculated removal rates in the remainder of the wetland are less consistent, and may indicate a change in mechanism to which a different model must be applied.

First-order model

The use of a first-order rate law to describe pollutant removal is common practice in sanitary engineering. The first order rate expression:

$$d[Fe^{+3}]/dt = -k_1[Fe^{+3}] \quad (3)$$

can be combined with the simplifying assumptions given above to calculate the first-order rate constant k_1:

$$k_1=\ln([Fe^{+3}]_{in}/[Fe^{+3}]_{out})/(\tau-\tau_{in}) \quad (4)$$

The procedure for calculating removal rates in g day^{-1} m^{-2} was similar to that used for the zero-order model. The removal rates are given in Table 2. They are again sensitive to the position in the wetland. The tendency of the removal rate to decline as the position of a particular treatment area of the wetland becomes further removed from the influent suggests that the removal rate does depend on $[Fe^{+3}]$ and lends credence to a first or higher order model.

The removal of Al was also considered, using a first-order rate expression, (Table 3). The behavior of [Mn] (Fig. 2) indicates that desorption from the substrate is occurring, possibly in accordance with a first-order rate law.

Conclusions

The procedures outlined above give a reasonable, simple predictive tool for the design of constructed wetlands. If a conservative figure of 1.0 g day^{-1} m^{-2} removal rate is taken for Fe^{+3} and the similarly derived 0.5 g day^{-1} m^{-2} removal rate is taken for Al^{+3}, and both removals are converted into acidity equivalents:

$$\text{Acidity equivalent} = 2.6883\ Fe^{+3}] + 5.5643\ [Al^{+3}] \quad (5)$$

an acidity removal rate of 5.5 g day^{-1} m^{-2} as $CaCO_3$ equivalent is calculated for the Bark Camp Run wetlands. This agrees well with the US Bureau of Mines (Hedin *et al.* 1994) guideline of 5.0 g day^{-1} m^{-2} as $CaCO_3$ and the 6 g day^{-1} m^{-2}

Table 1. Removal Rate (g day^{-1} m^{-2}) for Fe^{+3} Using Zero-order Model.

Cumulative Treatment Area

Wetland	64 m^2	96 m^2	128 m^2	256 m^2	374 m^2
1	nd	nd	1.01	+0.03	1.34
2	nd	3.04	1.21	+0.87	1.13
3	nd	nd	1.15	-0.03	2.92
4	2.90	nd	1.72	-.036	0.35
5	nd	nd	1.25	+0.55	1.67
6	1.93	nd	0.35	-0.45	12.10

Table 2. Removal Rate (g day^{-1} m^{-2}) for Fe^{+3} Using First-order Model.

Cumulative Treatment Area

Wetland	64 m^2	96 m^2	128 m^2	256 m^2	374 m^2
1	nd	nd	1.08	+0.04	0.71
2	nd	4. 93	0.38	+0.85	0.79
3	nd	nd	1.47	-0.02	0.52
4	5.90	nd	4.44	-0.33	0.17
5	nd	nd	3.78	-0.03	0.77
6	1.28	nd	0.81	-0.03	0.54

Table 3: Removal Rate (g day^{-1} m^{-2}) for Al^{+3} Using First-order Model.

Cumulative Treatment Area

Wetland	64 m^2	96 m^2	128 m^2	256 m^2	374 m^2
1	nd	nd	4.42E-01	-2.42E-02	-1.50E+00-
2	nd	3.09E+00	2.20E-01	-1.07E-02	-2.93E-01
3	nd	nd	4.58E-01	-2.30E-03	-1.19E+00
4	3.64E+00	nd	5.58E-01	+1.22E-01	-4.13E-01
5	nd	nd	2.08E+00	-1.66E-03	-1.30E-01
6	9.80E-01	nd	3.89E-02	+7.28E-02	-7.85E-01

In all three tables, the symbol "nd" means "not determined."

guideline given by Dietz *et al.* (1994). We conclude that the sizing guidelines for a system of this type receiving acidic influent can be explained by simple kinetic models. The designer is cautioned that the wetlands will experience a transient startup period, during which a steady model will not apply. Desorption of pollution indicators from the substrate during the transient startup period, particularly Mn, appears to be important, and the simplified models should not be applied to this period.

The declining performance of the wetlands indicated that the acid neutralizing properties of a substrate fashioned only of horse manure and inert, preponderantly SiO_2 stone will not be sustained for more than about 1 to 1½ year.

The performance of wetlands designed in a specific way to take advantage of anaerobic processes for treating a mine discharge with moderate acidity and moderately elevated metals has been studied. Water quality data

have been presented representing the first 806 days of operation, including transient startup conditions and short term performance following startup. Long-term data continue to be collected. The flow patterns in the wetlands were studied with a Br^- tracer, and the presence of flow through the substrate was demonstrated. Anhydrous NH_3 as pretreatment has the effect of accelerating the hydrolysis of Fe^{+3} and thus has the advantages of chemical treatment with other alkaline materials. The simple mathematical models we have developed compare favorably with previous design criteria. Further modeling efforts continue, and the results of this effort will be presented at a later time.

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MANGANESE AND TRACE METAL REMOVAL IN SUCCESSIVE ANAEROBIC AND AEROBIC WETLANDS

by

F.J. Sikora, L.L. Behrends, G.A. Brodie, and M.J. Bulls

Abstract. A microcosm study was conducted to evaluate the use of anaerobic wetlands preceding aerobic wetlands for removal of Mn, Cu, Ni, Zn, and Pb in wastewater. Initial concentrations for Mn, Cu, Ni, Pb, and Zn were 20, 2.0, 1.5, 2.1, and 2.0 mg/L, respectively. Each experimental unit consisted of three cattle-feeding troughs (cells) set in series. The first cell was anaerobic and the last two were aerobic. Water was delivered to the wetland cells at 20 mL/min for a period of 380 days starting August 24, 1994. The anaerobic wetlands consisted of three treatments replicated two times. The aerobic wetlands consisted of two treatments replicated three times. One anaerobic treatment contained organic matter and limestone (SP). Another anaerobic treatment contained organic matter, limestone, and canarygrass (SP&CG). The third anaerobic treatment consisted of canarygrass planted in river gravel (RG). Water flowed into the top and was discharged from the bottom of each anaerobic wetland. The aerobic treatments consisted of reciprocating or not reciprocating water between two cells containing river gravel. The anaerobic troughs with organic matter were effective in reducing sulfate to sulfide and producing alkalinity in the range from 80 to 300 mg/L. Manganese removal in the anaerobic systems decreased with time with the effluent anaerobic waters near equilibrium with respect to MnS and $MnCO_3$ toward the end of the experiment. Removal of Cu, Ni, Zn, and Pb was very effective in the anaerobic cells with organic matter due to precipitation of metal sulfides. Since Mn removal was ineffective in the long-term in the anaerobic system, aerobic wetlands would be necessary for further water treatment. Manganese removal in the reciprocating aerobic cells was quicker than in the nonreciprocating aerobic cells with removal due to precipitation of Mn oxides. Coupled anaerobic-aerobic wetlands appear to hold promise for removing trace metals via metal sulfide precipitation and Mn via Mn oxide precipitation.

Additional Key Words: alkalinity, compost, copper, lead, nickel, sulfate, sulfide, lead, zinc.

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[3] Publication in this proceedings does not prevent authors from publishing their manuscript, whole or in part, in other publication outlets.

INTRODUCTION

Due to federal limits placed on Fe and Mn concentrations in acid mine drainage effluent, considerable research has been conducted on the use of wetlands for removal of these metals. Monthly averages for Fe and Mn concentrations generally must be below 3 and 2 mg/L, respectively (Code of Federal Regulations, 1995). Both aerobic and anaerobic surface-flow wetlands have been used. Aerobic wetlands consist of a layer of water over soil with O_2 supplied via diffusion. Iron and Mn removal occurs via precipitation of metal oxides. Anaerobic wetlands consist of a layer of organic matter placed on top of soil. Anaerobic conditions are imposed with high C in the organic layer and rapid depletion of available O_2. Anaerobic conditions promote sulfate reduction to sulfide and subsequent precipitation of metal sulfides.

Removal rates for Fe and Mn in surface flow wetlands range from 10 to 20 and 0.5 to 1 $g/m^2/d$, respectively (Hedin et al., 1994). The reason for less efficient removal of Mn is due to slower kinetic processes in the oxidation of Mn^{2+} (Stumm and Morgan, 1981). Ferrous Fe readily oxidizes to ferric Fe at pH above 3.5 with rapid precipitation of ferric Fe oxyhydroxides. Uncatalyzed Mn^{2+} oxidation does not occur readily until pH >10 (Brezonik, 1994). Manganese oxidation can occur quicker at lower pH from 6 to 9 with autocatalysis from Mn sorption onto Mn oxide precipitates (McBride, 1994), catalysis via microorganisms (Ghiorse, 1984; Bender et al., 1994), or Mn sorption onto other solids (Davies and Morgan, 1989). Another process that limits Mn removal is the reduction of oxidized Mn in the presence of ferrous Fe. Ferrous Fe will readily reduce oxidized Mn precipitated as oxides, keeping Mn in solution (Hedin et al., 1994; Burdige et al., 1992). Due to the disadvantageous interaction between Fe and Mn, Mn removal in acid mine drainage does not occur significantly until Fe is reduced to low concentrations.

Passive Mn removal has been studied in a number of systems. Gordon (1989) and Gordon and Burr (1989) found Mn oxidation to be related to a black microbial coating found on rock surfaces. Adequate Mn removal occurred in rock bed filters with a similar black coating observed on rock surfaces (Thornton, 1995) and with gravel beds supporting a green algae-microbial mat consortium (Phillips et al., 1994). Manganese removal in both systems was purported to be biotically controlled. Biotic mediation in the algae mat may have been due to O_2 release and CO_2 uptake from the algae, maximizing Mn oxide precipitation in aerobic alkaline microenvironments (Bender et al., 1994). McMillen et al. (1994) studied the use of unsaturated vertical flow wetlands to provide conditions for Mn oxidation and precipitation and found effective Mn removal at initial Mn concentrations ranging from 0.5 to 60 mg/L. When a biocide was added, Mn removal remained high which indicated that abiotic catalysis of Mn oxidation and precipitation was the controlling factor for Mn removal. Contrary results were obtained in lake water samples with an initial Mn concentration of 2 mg/L where rate constants for Mn oxidation in poisoned samples were 10 to 100 times less than rate constants in unpoisoned samples (Johnson et al., 1995). Gordon and Burr (1989) also concluded microbial processes may play a vital role in Mn oxidation by studying Mn removal via microbial black slime on rock surfaces.

Iron and Mn have been the focus for metals removal in acid mine drainage. However, removal of trace metals in acid-mine drainage and other metal-laden wastewater will

receive more attention in the future due to impending limits on effluent concentrations via the Clean Water Act. Concentrations of Fe and Mn in acid mine drainage can range from 2 to greater than 150 mg/L, while concentrations of trace metals such as Cu, Ni, Pb, and Zn are usually less than 2 mg/L. An appropriate passive technology for trace metal removal is anaerobic wetlands since many of the trace metals of concern form metal sulfides with very low solubility. The disadvantage with surface-flow anaerobic wetlands is the anaerobic conditions are limited to the water-sediment interface with not all of the water being exposed to the reducing conditions. Improvement of treatment efficiency can occur by forcing water downward through an organic layer as done with successive alkalinity producing systems (SAPS) (Kepler and McCleary, 1994). A typical SAPS design consists of a surface layer of 1.6 to 1.9 m of water, 45 cm of compost below the water, and a 45 to 60 cm layer of limestone rock below the compost. The system provides reducing conditions and adds alkalinity to the water from sulfate reduction and limestone dissolution.

The objective of the current study was to ascertain mechanisms involved in the removal of Mn, Cu, Ni, Pb, and Zn in water treated by anaerobic and aerobic wetlands. The water used in the study was simulated acid mine drainage pretreated by an anoxic limestone drain and surface-flow aerobic wetlands where most of the Fe was removed. The anaerobic cells were hypothesized to remove the trace metals via sulfide precipitation. The aerobic cells were hypothesized to remove Mn via oxide formation.

MATERIALS AND METHODS

The study was conducted inside a greenhouse at the TVA Environmental Research Center in Muscle Shoals, AL. The cells used to simulate anaerobic and aerobic wetlands were insulated cattle feeding troughs that measured 1.1 x 0.6 x 0.6 m^3. The troughs were lined with a 40 mil plastic liner to prevent metal leakage from the galvanized steel. Each experimental unit consisted of 3 troughs placed in series with the first trough being the anaerobic cell and the second two troughs being the aerobic cells (Fig. 1). The troughs were plumbed with PVC pipe so water entered the surface of the first cell, exited at the bottom of the first cell, entered the second cell at the bottom, exited the surface of the second cell, entered the surface of the third cell, and exited on the bottom of the third cell. Surface area of one trough and one experimental unit was 0.5 and 1.5 m^2, respectively. Volume of one trough and one experimental unit was 0.3 and 0.9 m^3, respectively.

The experiment included 6 experimental units with 3 anaerobic treatments and 2 aerobic treatments. The anaerobic treatments consisted of canarygrass (CG) (*Phalaris arundinacea*), SAPS (SP) (12), and a SAPS+canarygrass combination (CG&SP). The SP treatment had a bottom 10 cm layer of crushed limestone gravel, a 30 cm mid-layer of composted chicken litter, and an 18 cm surface layer of water. The CG treatment consisted of canarygrass planted in 58 cm of river gravel. The CG&SP treatment consisted of river gravel on top of compost and limestone as set in the SP system and canarygrass planted in the gravel. Size distribution of the river gravel was 4% <6.7 mm, 66% 6.7 to 19 mm, 25% 19 to 25 mm, and 5% >25 mm. Size distribution of the limestone gravel was 4% <4.7 mm, 7% from 4.7 to 6.7 mm, 81% from 6.7 to 19 mm, and 8% from 19 to 25 mm. Size

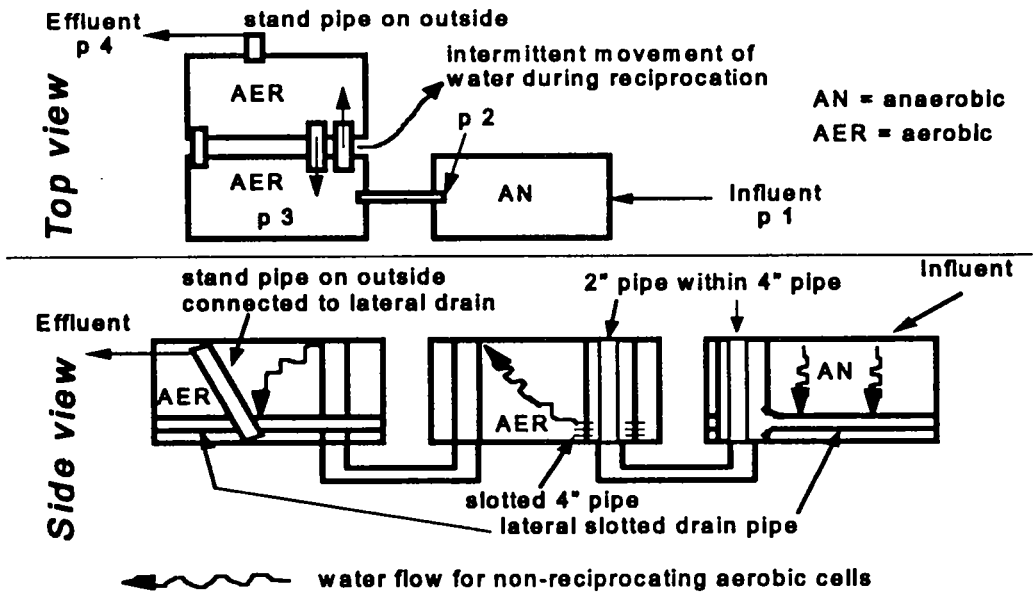

Figure 1. Diagram of one experimental unit used in the study. The four sampling positions are identified in Top view as p1, p2, p3, and p4.

distribution of the compost was 14% <0.4 mm, 3% from 0.4 to 0.6 mm, 8% from 0.6 to 1 mm, 3% from 1 to 1.2 mm, and 72% >1.2 mm.

The anaerobic treatments were replicated 2 times. Three replications of each of the two aerobic treatments were placed after a complete set of anaerobic treatments (Table 1). The two aerobic treatments consisted of reciprocating the top 20 cm of water from one cell to another (w/recip) or not reciprocating (wo/recip) (Behrends et al., 1993). The water was air-lifted by pumping air to the bottom of each of 2 paired aerobic cells through 5 cm diameter pipes. Air was intermittently pumped to each 5-cm pipe at 15 minute intervals. The pumped air moved the top 20 cm of water from one cell to another which exposed rock surface biofilms to air and provided aeration of the water.

After putting the appropriate media in the cells, 2 L of septic tank effluent were added to the composted chicken litter and on the surface of the planted gravel in order to inoculate the system with sulfate reducing bacteria. Nutrient solution was flowed through the anaerobic cells at 20 mL/min for a period of 1 month to nourish the canarygrass and sulfate-reducing bacteria. The nutrient solution contained 112, 137, 55, 12, 46, 2.3, 3.6, and 186 mg/L NH_4-N, Ca, K, P, Mg, Na, Cl, and S, respectively. Micronutrients in the solution consisted of 11, 59, 55, 17, 13, 3.2 and 296 ug/L B, Co, Mn, Mo, Zn, Cu, and Fe, respectively.

Acid mine drainage that simulated water pretreated with an aerobic wetland was passed through the experimental cells beginning on August 24, 1994 after the cells were acclimated with nutrients. August 24 was considered the initial time of the experiment and represented as day 0 in the figures. The salts $MnSO_4 \cdot H_2O$, $FeSO_4 \cdot 7H_2O$, $NiSO_4$, $CuSO_4$

Table 1. Outline of experimental treatments.

Experimental unit	Anaerobic treatment	Anaerobic rep	Aerobic treatment	Aerobic rep
1	CG	1	w/recip	1
2	SP	1	w/recip	2
3	CG&SP	1	w/recip	3
4	CG	2	wo/recip	1
5	SP	2	wo/recip	2
6	CG&SP	2	wo/recip	3

$\cdot 5H_2O$, $PbSO_4$, $ZnSO_4 \cdot 7H_2O$, $CaSO_4$, and $MgSO_4 \cdot 7H_2O$ were added to 1135 L resulting in expected concentrations of 20, 1.5, 2, 2, 2, 2, 90, 90, and 209 mg/L of Mn, Fe(II), Ni, Cu, Pb, Zn, Ca, Mg, and S, respectively. The solution was delivered to the first trough in each experimental unit at a rate of 20 ml/min. This flow rate resulted in loading rates of 0.82 and 0.12 $g/m^2/d$ for Mn and Fe, respectively, and a loading rate of 0.08 $g/m^2/d$ for the trace metals Cu, Ni, Pb, and Zn. Assuming the effluent water from TVA aerobic wetlands (Brodie, 1993) were treated by an equivalently sized wetland system for Mn removal, the Mn loading rates would range from 0.03 to 1.73 $g/m^2/d$ with an average of 0.34 $g/m^2/d$.

During May, 1995, the canarygrass was showing symptoms of N deficiency with very pale green leaves. Fertilizer was added on June 2, 1995 (day 282) to supply nutrients to canarygrass. To equalize effects of fertilizer addition on the anaerobic processes, fertilizer was added to all anaerobic treatments including the SP treatment that did not contain canarygrass. Fertilizer was added as N, P, and K as 34-0-0, 0-40-0, and reagent grade KCl at 60, 10, and 10 g, respectively, to each anaerobic cell. Fertilizer was mixed and added as a solid to the surface of each cell. In the gravel wetlands (CG and SP&CG), gravel was dug out to allow fertilizer to be added to the water. If the fertilizer was fully diluted with the water in the anaerobic wetlands, solution concentrations would be 154 mg/L N, 15 mg/L P, and 40 mg/L K.

Based on rock and compost porosity, the average pore volume in the cells was 40%, the retention time through each cell was 4.2 days and the retention time through the experimental unit was 12.5 days. Retention time through the organic layer in the anaerobic cells was approx. 2.1 days. This retention time was less than the retention times of 5 to 10 d reported to be maximum required for adequate sulfate reduction (Eger, 1992). The retention time through the limestone layer in the SAPS was approx. 0.7 d which was greater than the retention time of 0.5 d required for maximum alkalinity production (Hedin et al., 1994). The hydraulic load for the whole experimental unit was 1.9 cm/d and was within the range of hydraulic loads of 0.9 to 27 cm/d reported for wetland systems treating acid mine drainage (Hedin et al., 1994; Brodie, 1993).

Approximately once a month for a total of 11 sampling periods, water samples were collected for chemical analyses. Samples were collected in 4 locations at the influent (p1), effluent of anaerobic cells (p2), a 10 cm diameter well in the first aerobic cell (p3), and final effluent from the aerobic cell (p4) (Fig. 1). Water samples for metal analyses were acidified with 0.5 mL HNO_3 per 50 mL and refrigerated until analysis. Samples were analyzed for Fe, Mn, Ni, Cu, Pb, Zn, Ca, and Mg by inductively coupled plasma spectrophotometer (ICP). Detection

limits were 0.003, 0.003, 0.005, 0.004, 0.03, and 0.003 mg/L for Fe, Mn, Ni, Cu, Pb, and Zn, respectively. Another set of water samples were frozen until analyses for SO_4-S, C, NH_4-N, NO_3-N, and PO_4-P. Sulfate was determined by ion chromatography. Nonpurgeable organic C was determined by a Dohrmann DC 190 TOC analyzer. Ammonium, NO_3-N, and PO_4-P were determined with LACHAT flow-injection analysis. Alkalinity, pH, and sulfide were determined in water immediately after sampling. After measuring pH with a glass electrode, total alkalinity was determined via titration to pH 4.5 with NaOH using the pH electrode to monitor the endpoint. Water was sampled directly into an Orion SAOB solution at a ratio of 10 mL sample to 10 mL SAOB solution and analyzed with a sulfide ion-selective electrode.

Probes were placed into access wells to monitor dissolved oxygen (DO), temperature, electrical conductivity (EC), and redox. Dissolved oxygen and temperature were monitored with a YSI probe and meter. Electrical conductivity was determined with an Orion probe and meter. Platinum-tipped copper wire electrodes were kept in place for redox measurements. Two redox electrodes were placed into the effluent sump of each anaerobic cell 7.5 cm from the bottom (p2, Fig. 1) and one redox probe was placed in a well in the first aerobic cell 15 cm from the bottom (p3, Fig. 1). Another redox probe was placed in a well in the second aerobic cell 15 cm from the bottom near the subsurface effluent header (see Fig. 1). The redox measured from this redox probe was assumed to be the redox of the effluent water (p4). The redox probes were checked every 3 months by placing the probes in a standardized redox solution (0.1 M Fe^{2+}, 0.1 M Fe^{3+}, and 1 M H_2SO_4) that should have yielded a 432 mV reading at 22 °C using a saturated calomel reference electrode. Greater than 90% of the probes tested yielded values that were within ± 30 mV of the standard mV reading. If the probe tested fell outside of this range, it was replaced. The reference electrode used for redox determination was a saturated calomel electrode. The measured redox potentials were adjusted in reference to a standard H_2 reference electrode by adding 244 mV to the electrode readings (Stumm and Morgan, 1981).

There were six experimental units as outlined in Table 1 that were randomly laid out in the study. Data from the anaerobic and aerobic wetlands were analyzed separately to first discern differences among the three anaerobic treatments and then to discern differences among the two aerobic treatments. In an overview of the whole system (Table 2), data was averaged across the two replications for each anaerobic treatment and across the three replications for each aerobic treatment. Since the anaerobic effluent water from CG and the treatments with organic matter (SP and CG&SP) were very different, the effectiveness of the aerobic treatments was further evaluated (Table 3) for treating effluent only from anaerobic cells containing organic matter. The aerobic treatments were thus averaged across aerobic replications 2 and 3 as shown in Table 1.

Total alkalinity determined from titration also included HS^- and HPO_4^{2-} in the waters tested in addition to HCO_3^-. Therefore,

$$A_T = A_C + [HS^-] + [HPO_4^{2-}]$$

where A_T is total alkalinity and A_C in carbonate alkalinity (Butler, 1991). The unit for alkalinity in the above equation is mole/L. Rearranging and solving for $[HS^-]$ and $[HPO_4^{2-}]$ to equal total sulfide and phosphorus concentrations yields:

$$A_C = [HCO_3^-] = A_T - \frac{10^{-7.02} S_T}{10^{-7.02} + 10^{-pH} a_{HS}} - \frac{10^{-7.2} P_T}{10^{-7.2} + 10^{-pH} (a_{HPO4}/a_{H2PO4})}$$

where S_T and P_T are total sulfide and phosphate concentrations, respectively, in mole/L and $10^{-7.02}$ and $10^{-7.2}$ are proton dissociation constants for H_2S and $H_2PO_4^-$, respectively. The activity coefficients (a_i) were calculated from Davies equation using electrical conductivity to estimate ionic strength (Lindsay, 1979). The carbonate alkalinity shown as mole/L HCO_3^- in the above equation was converted to $CaCO_3$ alkalinity with units of mg/L by:

$$A_C\ (100{,}000\ \text{mg}\ CaCO_3 / 2\ \text{mole}\ HCO_3^-)$$

Predicted carbonate alkalinity produced from calcite dissolution was calculated from the increase in Ca concentration observed from the influent to the effluent of the anaerobic wetlands. Assuming the Ca increase was due solely to calcite dissolution, 2 moles of alkalinity are produced for every mole of Ca dissolved:

$$2\ H^+ + CaCO_3 \longrightarrow Ca^{2+} + 2\ HCO_3^-$$

The alkalinity was calculated in terms of $CaCO_3$ alkalinity in mg/L as:

$$\frac{(\text{Ca increase, mg/L})}{(40.1\ \text{mg}\ Ca^{2+}/\text{mmol})} \times (2\ \text{mmol}\ HCO_3^- / \text{mmol}\ Ca^{2+}) \times (100\ \text{mg}\ CaCO_3 / 2\ \text{mmol}\ HCO_3^-)$$

For sulfate reduction, 2 moles of alkalinity are produced for every mole of sulfide produced:

$$2\ CH_2O + SO_4^{2-} \longrightarrow S^{2-} + 2\ HCO_3^-$$

Predicted alkalinity from sulfate reduction was calculated in terms of $CaCO_3$ alkalinity in mg/L as:

$$\frac{(\text{S conc., mg/L})}{(32.1\ \text{mg S/mmol})} \times (2\ \text{mmol}\ HCO_3^- / \text{mmol}\ S^{2-}) \times (100\ \text{mg}\ CaCO_3 / 2\ \text{mmol}\ HCO_3^-).$$

Activities of chemical components were calculated from solution concentrations using MINEQL$^+$ (Schecher and McAvoy, 1991). The activites were used to determine the potential existence of various solid phases that would support determined metal activity levels assuming equilibrium conditions existed. For evaluation of potential Mn solid phases in the anaerobic wetlands, saturation indices were calculated for Mn solid phases as:

$$\text{Log (SI)} = \text{Log (IAP} / K_{sp})$$

where SI is the saturation index, IAP is the ion activity product in solution, and K_{sp} is the theoretical solubility product for the solid phase in question (Sposito, 1989). If Log (SI) were greater than 0, the solution was supersaturated with respect to the solid phase. If Log (SI) were less than 0, the solution was undersaturated with respect to the solid phase. In both these cases, the existence of the solid phase was not likely. If Log (SI) were equal to 0, the solution was in equilibrium with the solid phase and the existence of the solid phase in question was likely.

For evaluating the potential Cu, Ni, Pb, and Zn solid phases in the anaerobic wetlands and potential Mn solid phases in the aerobic wetlands, metal activities were plotted versus pe+pH and compared to solubility lines for known solid phases. The pe was determined as redox (mV)/59.2 (Lindsay, 1979). The

pH was added to pe to obtain pe+pH. Since the reducing or oxidizing potential of an aquatic environment is affected by both pe and pH, the pe+pH is a useful parameter to define the oxidizing potential of the water. For comparison, another scale is presented for redox in mV at the average pH levels of the waters analyzed.

The Log (CO_2, atm) was calculated using the HCO_3^- activity calculated from MINEQL$^+$ and pH as:

$$\text{Log}(CO_2, \text{atm}) = 7.82 - \text{pH} + \log(HCO_3^-).$$

Results and Discussion

Overview of Coupled Anaerobic and Aerobic Cells

Table 2 displays average water chemistry data across all sampling periods and replications for effluent waters from each anaerobic and aerobic treatment. In the anaerobic effluent waters, pH and alkalinity were greater in the SP and CG&SP treatments compared to the CG treatment (Table 1). The organic matter in the SP and CG&SP treatments was effective in producing highly anaerobic conditions as evidenced by redox less than -170 mV, DO less than 4% saturation, and sulfide concentrations from 19 to 21 mg/L in the effluent waters. The high alkalinity produced in these anaerobic waters resulted in CO_2 partial pressures that were supersaturated with respect to atmospheric CO_2 of $\log(CO_2) = -3.52$ (Table 2). As the water was treated by the aerobic cells, the CO_2 degassed from the water more effectively in the reciprocating treatment compared to the non-reciprocating treatment which resulted in lower alkalinity, lower CO_2 partial pressures, and higher pH in the effluent from the reciprocating treatment compared to the nonreciprocating treatment.

Calcium concentrations in the effluent of the anaerobic cells increased from the influent concentrations due to dissolution of calcite (Table 1). Calcite was added to the SP and CG&SP treatments as limestone. Since river gravel was the only medium used in the CG treatment, an increase in Ca concentration in this treatment is indicative of some calcite present in the river gravel. Batch titrations with HCl has shown the river gravel to have a calcium carbonate equivalence of 0.35%.

Manganese concentration was decreased approximately 50% in the anaerobic cells with further reduction to levels below 0.3 mg/L in the aerobic cells (Table 1). The SP and CG&SP treatments of the anaerobic cells were very effective in reducing concentrations of Cu, Ni, Pb, and Zn to levels near the detection limits of the ICP. The CG anaerobic treatment was less effective with concentrations ranging from 0.06 to 0.44 mg/L for Cu, Ni, and Zn.

The addition of compost in the SP and CG&SP anaerobic treatments resulted in greater NH_4-N and PO_4-P concentrations in the anaerobic effluent waters compared to the CG treatment (Table 1). The reciprocating aerobic wetland was very effective at reducing NH_4-N levels below analytical detection. However, both reciprocating and nonreciprocating aerobic treatments only reduced PO_4-P concentrations to 2.9 and 2.4 mg/L, respectively, with an average 3.6 mg/L aerobic influent concentration.

Table 2. Water chemistry of the influent and effluent of anaerobic cells and effluent of aerobic cells.

		Anaerobic effluent			Aerobic effluent	
Parameter †	*Influent*	*CG*	*SP*	*CG&SP*	*w/recip*	*wo/recip*
pH	4.9(0.5)‡	6.7(0.2)	7.2(0.1)	7.0(0.1)	7.7(0.2)	7.4(0.4)
Alkalinity	0	81(33)	142(44)	193(59)	57(18)	118(41)
log(CO_2, atm)		-1.73(0.13)	-1.96(0.26)	-1.64(0.21)	-2.88(0.19)	-2.12(0.37)
Redox		304(357)	-172(120)	-174(93)	584(136)	532(163)
DO	98(14)	11(11)	3(3)	1.8(1.8)	92(18)	30(19)
Ca	90(5)	120(15)	122(10)	133(17)	127(14)	142(14)
Mg	93(5)	96(3)	99(3)	98(4)	89(14)	92(18)
Mn	20(1)	11(8)	10(5)	10(6)	.04(.04)	.24(.24)
Cu	2.0(0.2)	.06(.06)	.009(.009)	.005(.002)	.005(.003)	.03(.03)
Ni	1.5(0.3)	0.44(0.41)	.016(.016)	.008(.008)	.005(.004)	.006(.003)
Pb	2.0(0.3)	.04(.03)	< .03	.03(.01)	< .03	< .03
Zn	2.1(0.1)	.30(.30)	.009(.009)	.01(.01)	.006(.005)	.007(.007)
Fe	1.0(1.2)	1.2(1.2)	.062(.062)	.073(.072)	.005(.005)	.009(.009)
S	< .02	.10(.10)	19(9)	21(9)	< .02	< .02
SO_4-S	207(36)	211(31)	193(36)	182(33)	209(42)	198(31)
org-C	1.7(1.0)	2.2(1.6)	5.3(1.8)	5.2(2.1)	3.5(1.8)	3.7(2.1)
EC	1.12(.04)	1.23(.06)	1.24(.04)	1.27(.08)	1.18(.17)	1.26(.07)
NH_4-N	<.02	.28(.28)	2.2(2.2)	1.6(1.6)	<.02	.8(.8)
NO_3-N	.23(.06)	.04(.04)	<.02	<.02	3.5(3.5)	2.0(2.0)
PO_4-P	<.03	.04(.04)	4.8(3.9)	6.0(5.0)	2.9(2.9)	2.4(2.4)

† Units for all parameters are in mg/L except for pH and log (CO_2), which are unitless, redox is in mV, DO is % saturation, and EC is mS/cm.

‡ Data is averaged across sampling time and replications. Values in parenthesis represent standard deviations.

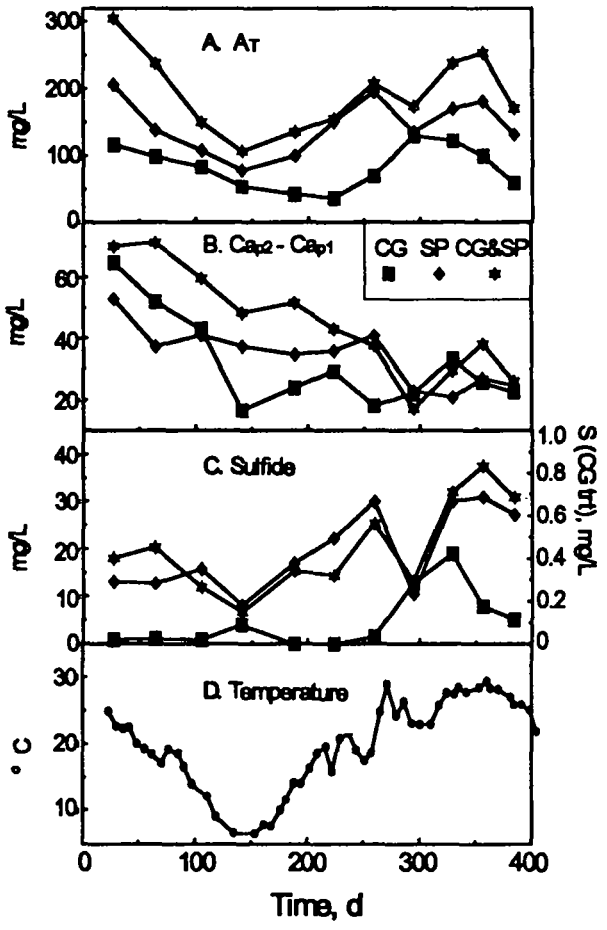

Figure 2. Total alkalinity, A_T (A), Ca increase (B), sulfide (C), and water temperature (D) changes with time in the effluent waters from the anaerobic wetland cells for the canarygrass (CG), SAPS (SP), and canarygrass+SAPS combination (CG&SP) treatments. Calcium increase is the difference in Ca concentrations in the effluent and influent waters of the anaerobic cells (Ca increase = Ca_{out} - Ca_{in}).

Anaerobic Wetland Cells

Alkalinity in the effluent waters of the anaerobic cells was correlated fairly well with water temperature (Fig. 2A). Alkalinity was greater with higher water temperatures. Alkalinity generation was greater in the anaerobic treatments containing compost (SP and CG&SP) compared to CG. The alkalinity from CG&SP was greater than SP alkalinity during the warmest times of the year at the initiation of the experiment (day 0 to day 60) and during the summer (day 320 to day 360).

A chemical parameter that has a relationship to alkalinity generation is Ca concentration difference between effluent and influent waters (Fig. 2B). If the Ca difference can be assumed to be due solely to calcite dissolution, some conclusions on calcite dissolution effectiveness can be made in these systems. Calcite dissolution was greatest at the initiation of the experiment and continually declined with time for all anaerobic treatments. Calcite dissolution declined more rapidly in the CG treatment probably due to a limited amount of calcite present in the river gravel. Calcite dissolution did not show seasonal trends. For example, the Ca difference continually declined during the warming time of the year from day 200 to day 400. The decline in calcite dissolution may have been due to calcite particles being continually coated by organic matter and inorganic precipitates.

Another chemical parameter related to alkalinity generation is sulfide concentrations as shown in Fig. 2C. Sulfide was much greater in the treatments containing compost (SP and CG&SP) compared to the treatment without compost (CG). The increase in sulfide concentrations was correlated well with increased water temperatures that must have increased productivity of sulfate reducing bacteria. At day 290 sampling, there was a drastic decline in sulfide concentration in SP and CG&SP. The systems were fertilized with N, P and K at day 282. The decline in S may have been due to sulfate-reducing bacteria unsuccessfully competing with other heterotrophic bacteria that may have flourished after adding supplemental nutrients.

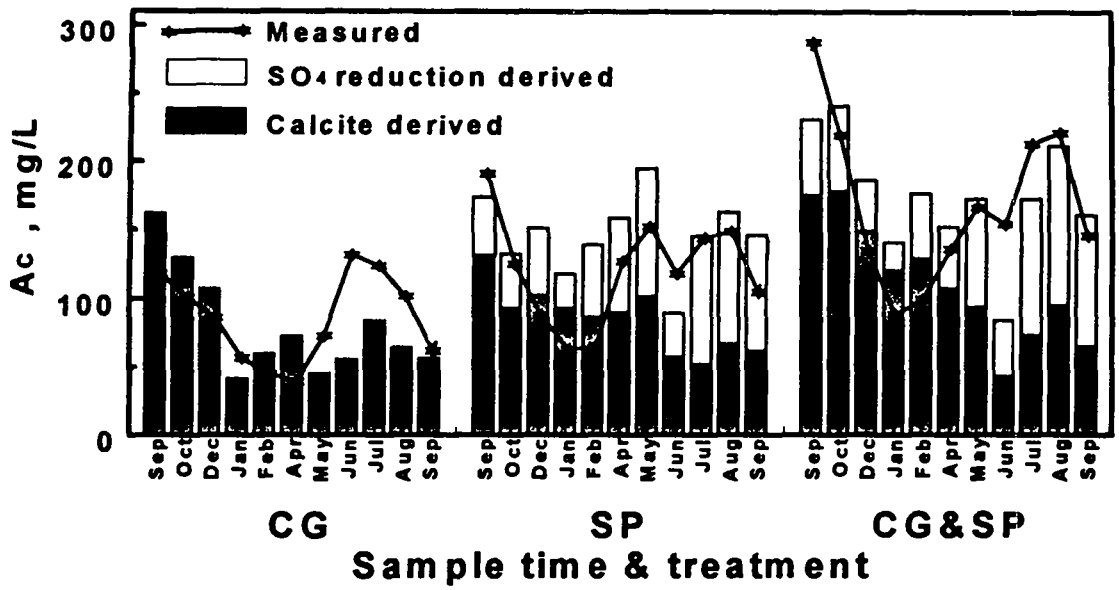

Figure 3. Calculated carbonate alkalinity (A_C) based on calcite dissolution and sulfate reduction compared to measured carbonate alkalinity in effluent waters from the anaerobic wetland cells for the canarygrass (CG), SAPS (SP), and canarygrass + SAPS combination (CG&SP) treatments.

The expected carbonate alkalinity produced from calcite dissolution and sulfate reduction was calculated and compared to carbonate alkalinity from measured total alkalinity (see Materials and Methods) (Fig. 3). Alkalinity calculated from calcite dissolution and sulfate reduction followed a trend similar to measured alkalinity with lower alkalinity during colder periods of the year and higher alkalinty during warmer periods. However, there were discrepancies between measured alkalinity and total alkalinity predicted from calcite dissolution and sulfate reduction. The overestimated alkalinity predicted from calcite dissolution and sulfate reduction can be due to an increase in Ca concentration related to chemical processes that do not release HCO_3^- into solution, such as exchangeable Ca or dissolution of non-carbonate Ca minerals such as gypsum. Another possible explanation is that pure calcite was not present so the assumption that 2 moles of HCO_3^- was released for each mole of Ca dissolved may have overestimated alkalinity calculated from Ca dissolution. Yet another explanation is that HCO_3^- released from calcite dissolution precipitated with other metals, such as Mn, leaving Ca in solution while HCO_3^- was removed.

The underestimation of alkalinity during the months from May to August in CG and from June to July in CG&SP may have been due to active canarygrass roots releasing HCO_3^- during anion uptake (Mengel and Kirkby, 1982), which would be an unaccounted source of alkalinity. The SP treatment did not contain canarygrass, but did contain algae in the surface water. Lower CO_2 and higher pH were observed in the SP treatment compared to the CG and CG&SP treatments (Table 2). Algae absorbs CO_2 during photosynthesis which increases pH (Vymazal, 1995), but these processes counterbalance

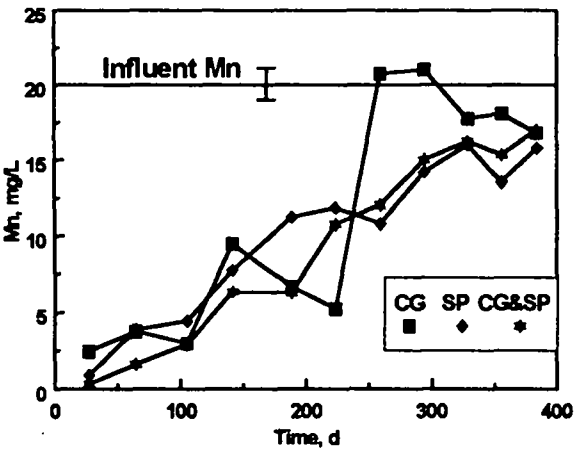

Figure 4. Manganese concentrations in effluent waters from the anaerobic wetland cells for the canarygrass (CG), SAPS (SP), and canarygrass+SAPS combination (CG&SP) treatments with time.

one another with no change in alkalinity (Butler, 1991). Therefore, large differences were not observed between measured and predicted alkalinity in SP during the summer months.

The anaerobic cells effectively removed Mn in influent waters with low effluent Mn concentrations observed at the beginning of the experiment (Fig. 4). However, as time progressed Mn removal efficiency declined as Mn concentrations approached influent concentration of 20 mg/L. Similar trends in Mn removal were observed by Wildeman et al. (1993) and Stark et al. (1995). In Wildeman et al. (1993), bench-scale studies of anaerobic systems have shown greater than 95% Mn removal from 24 to 71 days of operation. However, in a pilot-scale anaerobic wetland with influent Mn concentration of 31 mg/L, Mn removal efficiency was near 50% during the first 3 months with removal efficiencies declining to 10% from 6 to 24 months of the study. In Stark et al. (1996), results similar to Fig. 4 were obtained with Mn concentrations slowly increasing in effluent of anaerobic wetlands with 50% removal efficiency occurring at approximately 120 day.

As anaerobic wetlands are initiated there may be plentiful sorption sites on organic matter and limestone for Mn to sorb onto. Although the affinity of organic functional groups for Mn is low (McBride, 1994; Wildeman et al., 1994), some sorption of Mn onto abundant carboxyl and hydroxyl functional groups in the compost should be expected. Manganese sorption onto limestone may occur via surface displacement of Ca^{2+} for Mn^{2+} to form a solid solution or precipitation of a $MnCO_3$ solid phase at the surface (Evangelou, 1995). Once Mn sorption capacities were reached on both compost and limestone media and no other important removal mechanism existed, Mn concentrations in effluent water from anaerobic wetlands would be expected to increase as observed in Fig. 4.

To determine what Mn solid phases were controlling Mn in the water of the anaerobic cells, saturation indices were calculated and are plotted in Fig. 5. Throughout the experiment, the water was very much undersaturated with respect to MnOOH. The Mn oxide solid phases contain Mn in an oxidized state. The reducing conditions of the anaerobic wetlands keep Mn in the reduced manganous state (Mn^{2+}), making formation of Mn oxides extremely unlikely.

The anaerobic waters were undersaturated with respect to $MnCO_3$ at the beginning of the experiment. From day 220 to day 400, water from the SP and CG&SP treatments was near equilibrium to supersaturated with respect to $MnCO_3$. From day 300 to day 400, water from CG was near equilibrium to undersaturated with respect to $MnCO_3$. Similar Log (SI) values were reported for $MnCO_3$ in effluent waters from anoxic limestone

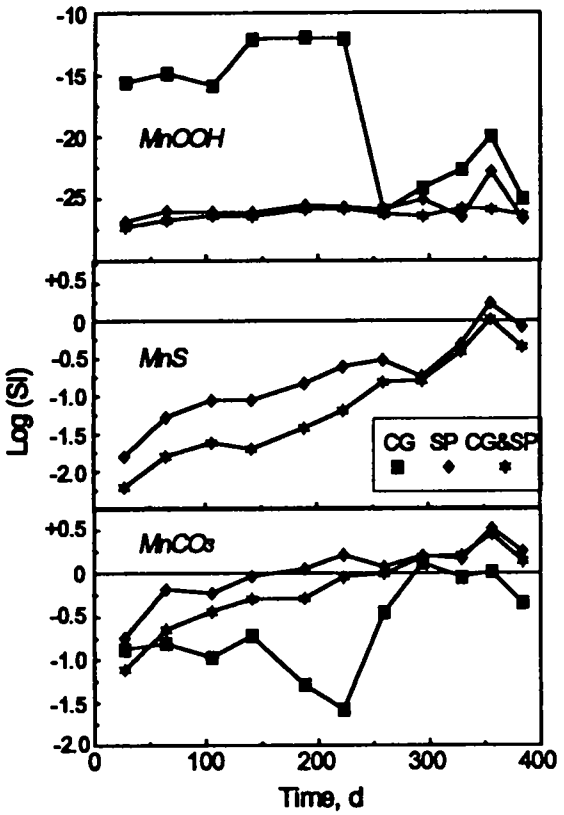

Figure 5. Logarithm of saturation indices (Log (SI)) for MnOOH(s), MnS(s), and $MnCO_3$(s) in effluent waters from the anaerobic wetland cells for the canarygrass (CG), SAPS (SP), and canarygrass+SAPS combination (CG&SP) treatments with time.

drains (Hedin and Watzlaf, 1994) where limited Mn removal in the waters occurred. The Log (SI) values indicated conditions existed for $MnCO_3$ to form and may exert some control on Mn concentration in water toward the end of the experiment. The supersaturation levels that existed may have been due to kinetic limitations on $MnCO_3$ precipitation.

The waters from the treatments with organic matter (SP and CG&SP) were undersaturated with respect to MnS for most of the study until day 350 when the solutions were near equilibrium with this mineral. Saturation indices was not calculated for the CG treatment since sulfide was not detected in most of the samples. The solubility of MnS is fairly high compared to other metal sulfides. Therefore, removal of Mn in anaerobic wetlands via MnS formation is not very effective.

The data in Fig. 5 further supports the hypothesis that Mn was being sorbed onto compost and limestone at the beginning of the experiment when the waters were undersaturated with respect to MnS and $MnCO_3$. As the sorption sites filled, Mn concentrations rose to a high enough level so MnS and $MnCO_3$ could begin to control Mn solubility. The advantage of removing Mn via precipitation as $MnCO_3$ or MnS is that the anion precipitating with Mn is in constant supply from the sulfate reduction process. This is contrary to Mn removal via sorption where the component removing Mn (sorption sites) has a finite availability. A disadvantage of Mn being removed as $MnCO_3$ or MnS is that the solubilities of the minerals are so high that they support Mn concentrations in water that are unacceptable for discharge (Fig. 4). Therefore, a subsequent oxidation of the Mn to form Mn oxides is required in an aerobic wetland.

The concentrations of Cu, Ni, Pb, and Zn in the effluent from the anaerobic cells containing organic matter (SP and SP&CG) were always near the detection limit for the ICP (Table 2). However, the Ni and Zn concentrations in the anaerobic effluents from the CG treatment increased to 1.1 to 1.2 mg/L after 120 days and subsequently decreased (Fig. 6).

The redox in the treatments with organic matter were always at very low values (approx. -200 mV) throughout the experiment, whereas the redox in the effluent stream from the CG treatment was >400 mV up to day 240 and then drastically dropped to maintain redox levels between -200 and 200 mV. The decrease in the

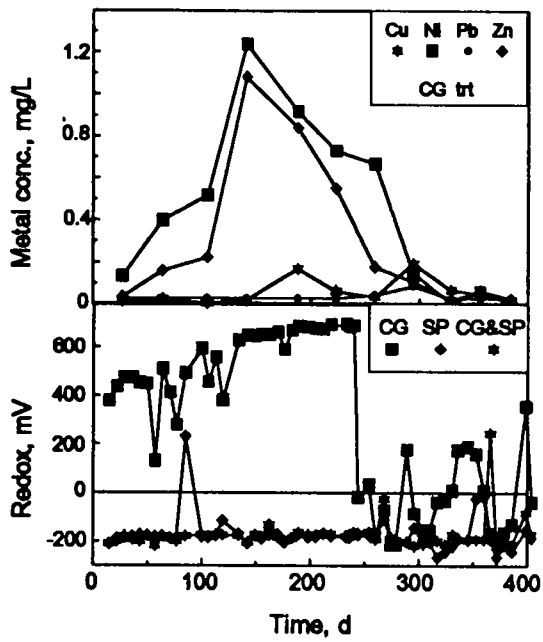

Figure 6. Concentrations of Cu, Ni, Pb, and Zn and redox in effluent waters from the anaerobic wetland cells for the canarygrass (CG), SAPS (SP), and canarygrass+SAPS combination (CG&SP) treatments with time.

redox in the CG treatment coincided with the spring growth of the canarygrass in late April 1995. As in this study, canarygrass growth has been observed to reduce redox levels to very low values in gravel bed wetlands (Zhu and Sikora, 1995; Steinberg and Coonrod, 1994). Because of this ability to reduce redox levels, canarygrass was chosen for this experiment to determine if sulfide could be produced in a plant-rock system without organic matter. Even though redox levels were low (<200 mV), there must still have been limitations to sulfate reduction due to the low sulfide concentrations observed (Fig. 2). The limitation was probably a lack of available C.

To determine what solid phases may have been controlling concentration of trace metals in the anaerobic wetland effluents, stability diagrams were constructed with log of metal activities plotted versus pe+pH (Fig. 7). The data represented in the figure is only from samples in which the metal was detected above analytical detection limits of the ICP. This represented 33, 30, 8, and 42 % of the samples collected for Cu, Ni, Pb, and Zn, respectively. The data for the treatments with organic matter (SP and CG&SP) were combined since there was very little difference between these treatments. The Cu activities were undersaturated with respect to $CuCO_3$. Except for some data from the CG treatment, there was an observable trend that Cu activities were drastically reduced with a decrease in pe+pH due to precipitation of CuS. Most of the CG data at higher pe+pH indicate Cu may be controlled by a Cu oxide solid phase. Nickel activity levels were undersaturated with respect to $NiCO_3$ and $Ni(OH)_2$ indicating some other solid phase or sorption was controlling this metal in solution. At low pe+pH, data clustered around the NiS solubility line. From the limited data available for Pb, $Pb(OH)_2$ appeared to control Pb activities at high pe+pH and PbS controlled Pb activity at low pe+pH. For Zn, solubility appeared to be controlled by $ZnCO_3$ in the CG data at high pe+pH and by ZnS at low pe+pH for all treatments. Figure 7 is only useful as a qualitative predictor of what minerals may control metal activities in solution because indirect analysis of what solid phases may be present is made from solution chemistry. Other solid phases not considered or sorption of metals onto Fe and Mn oxides (McBride, 1989) may have also controlled metal solution activities.

The similarity in the solubility diagrams of Fig. 7 is that metal sulfides precipitate and reduce metal activities to very low levels in solution at low pe+pH. The pe+pH value at which the metal sulfide solubility line begins to decline depends on the solubility of the

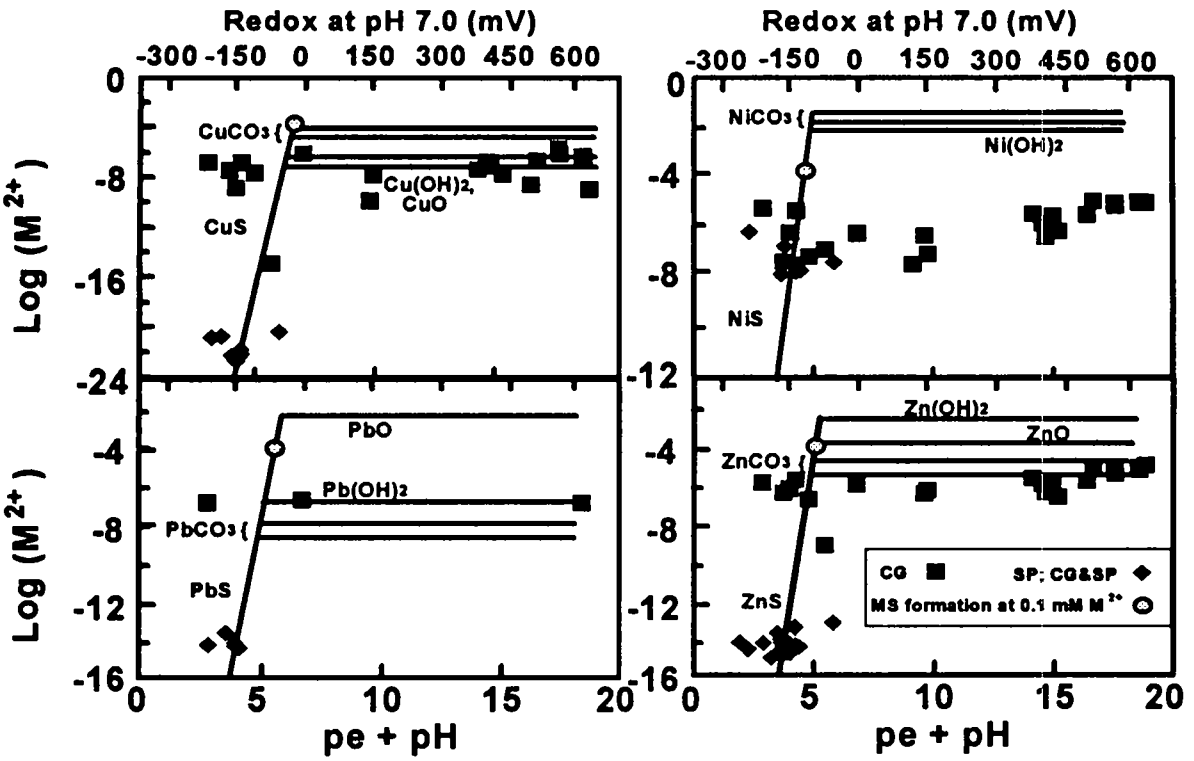

Figure 7. Metal activities as a function of pe + pH for Cu, Ni, Pb, and Zn in effluent waters from the anaerobic wetland cells for the canarygrass (CG), SAPS (SP), and canarygrass+SAPS combination (CG&SP) treatments. Symbols represent experimental data. Lines represent solubility of known solids. The two lines for $MnCO_3$ represents the range of $MnCO_3$ solubility for low measured alkalinity (upper line) and high measured alkalinity (lower line). The upper x axis is the equivalent redox value for the average pH level of 7.0.

metal sulfide. The greater the solubility, the lower the pe+pH has to be before precipitation of the metal sulfide begins. The redox levels at pH 7 required to reduce metal activities below 0.1 mM are -24, -62, -94, and -114 mV for Cu, Pb, Zn, and Ni, respectively (shown as shaded circles in Fig. 7). Although not shown in Fig. 7, the required redox at pH 7 to reduce Mn activities below 0.1 mM via MnS formation is -175 mV. Therefore, the affinity of metals to form metal sulfides decreases in the order Cu>Pb>Zn>Ni>Mn. The same trend for metal sulfide formation can be presented by comparing solubility products of the metal sulfides (Hedin et al., 1994). Both the anaerobic treatments with organic matter (SP and CG&SP) produced very low redox levels, which were adequate in precipitating Cu, Ni, Pb, and Zn as metals sulfides and reducing solution concentrations of the metals to very low levels (Table 1).

The stability diagrams for Ni and Zn concentrations in the CG treatment in Fig. 7 can be used to explain observed changes in Ni and Zn concentrations (Fig. 6). At higher pe+pH, $ZnCO_3$ appeared to control Zn solubility (Fig. 7). At the beginning of the experiment, alkalinity was high which would be expected to lower Zn concentrations

Table 3. Water chemical parameters affected by reciprocation in aerobic treatments. Only the aerobic cells treating effluent from the SP and CG&SP anaerobic treatments are considered.

Parameter †	Anaerobic effluent	First aerobic cell		Aerobic effluent	
		w/recip	wo/recip	w/recip	wo/recip
pH	7.1	7.7(0.2)‡	7.2(0.1)	7.7(0.2)	7.3(0.4)
Alkalinity	168	63(19)	149(62)	63(16)	127(46)
Log(CO_2)	-1.8	-2.85(0.10)	-1.91(0.20)	-2.85(0.13)	-2.17(0.46)
Redox	-173	544(124)	-195(87)	588(112)	509(155)
DO	2.4	94(14)	2.3(2.3)	93(14)	11(11)
Mn	10	0.31(0.88)	6.2(3.4)	0.05(0.08)	0.35(0.35)

† Units for pH and Log (CO_2,atm) are unitless, alkalinity is in mg/L, redox is in mV, DO is in % saturation, and Mn is in mg/L.

‡ Data is averaged across sampling time and replications. Values in parenthesis represent standard deviations.

due to $ZnCO_3$ precipitation. As time progressed, alkalinity declined (Fig. 1) resulting in subsequent increase in Zn concentrations (Fig. 6). The redox of the CG effluent water dramatically dropped at day 240 (Fig. 6) which resulted in a decrease in pe+pH to levels that predicted ZnS and NiS controlled (Fig. 7) and lowered Zn and Ni concentrations (Fig. 6).

Aerobic Wetland Cells

There were little differences in Mn concentrations in the effluent waters from the aerobic cells treating water from the anaerobic cells containing organic matter when comparing reciprocating and non-reciprocating treatments (Table 3). However, considerable differences were observed for these parameters in waters taken from the first aerobic cells (Table 3). The cells with reciprocation resulted in higher DO, redox, and pH levels in the first aerobic cells compared to non-reciprocating cells. The high redox and pH levels most likely resulted in lower Mn concentrations in the first reciprocating aerobic cells compared to the first nonreciprocating cells (Table 3) due to greater likelihood for Mn oxide precipitation to occur. The long retention time of 9 days in the aerobic cells was ample for adequate removal of Mn in the final effluent whether the cells were reciprocating or not.

A plot of log Mn activity vs pe+pH was constructed (Fig. 8) for predicting what solid phases may have been controlling Mn concentrations in the aerobic wetlands. The lines are predictive solubility lines for MnS, $MnCO_3$, MnOOH, Mn_2O_3, and Mn_3O_4. Without reciprocation, most of the waters in the first aerobic cells had low pe+pH values and the Mn

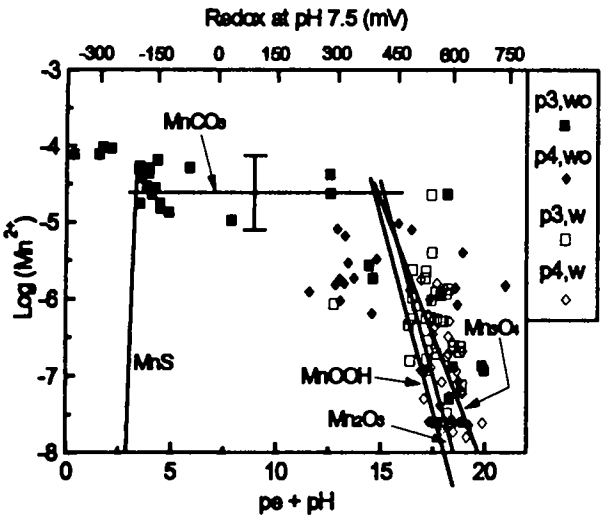

Figure 8. Manganese activities as a function of pe + pH in water sampled from p3 and p4 (see Fig. 1) in the aerobic wetland cells for treatments without reciprocation (wo) and with reciprocation (w). Symbols represent experimental data. Lines represent solubility of known solids. The error bar for $MnCO_3$ represents the range of $MnCO_3$ solubility for low measured alkalinity (upper boundary) and high measured alkalinity (lower boundary). The upper x axis is the equivalent redox value for the average pH level of 7.5.

activities appeared to be controlled by a combination of MnS and $MnCO_3$. In the effluent waters of the nonreciprocating treatment, the pe+pH levels increased and the Mn activities approached solubility with Mn oxides. The pe+pH levels were high at both sampling positions in the reciprocating treatment with Mn activities clustered around the Mn oxide solubility lines.

It is difficult to ascertain whether the mechanism for Mn oxidation was biotic or abiotic since poisoned abiotic treatments were not included in the experiment. However, a conclusion can be made that producing an environment that thermodynamically favors Mn^{2+} oxidation and precipitation as Mn oxides is the first requirement for Mn removal from solution. Such an environment is produced with pe+pH above 15 or redox above 400 mV at pH 7.5. A second requirement may be that an appropriate type and quantity of bacteria needs to be present to increase the rate of Mn oxidation. The influent waters to the aerobic systems were highly anaerobic which may have exaggerated the utility of having a reciprocating aerobic wetland to rapidly increase pe+pH to allow for rapid Mn oxidation and subsequent precipitation. For waters that already have higher pe+pH, aerating the water may not be that critical. Although aeration occurred via reciprocating water in a gravel bed in the current experiment, aeration by passing the water over a riprap may also suffice in increasing pe+pH levels to allow for rapid Mn oxidation and precipitation.

CONCLUSIONS

The data collected the first year of this study has shown decreased calcite dissolution and Mn removal efficiencies with time in anaerobic wetlands. Alkalinity generation was observed to be seasonal with greater alkalinity in the warmer time of the year. Mn removal was very effective in aerobic wetlands following anaerobic wetlands. Aeration of effluent waters from anaerobic wetlands is required for raising pe+pH to high enough levels to allow for rapid oxidation of Mn to form Mn oxide precipitates. Trace metal removal was very effective in the anaerobic treatments with organic matter due to precipitation of metal sulfides. The study is being con- ducted for another year to determine long term efficiencies of the wetland systems and to develop scaled-up design criteria.

ACKNOWLEDGMENTS

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THE USE OF OVERLAND FLOW WETLAND TREATMENT SYSTEMS[1] TO REMOVE NICKEL FROM NEUTRAL MINE DRAINAGE

by

Paul Eger, Jon Wagner, Glenn Melchert[2]

Abstract. In 1992, two overland flow wetland treatment systems were built in existing natural wetlands in northeastern Minnesota to remove copper, nickel, cobalt and zinc from neutral mine drainage. Typical input metal concentrations ranged from 2 - 5 mg/L for nickel, to less than 0.1 mg/L for copper, cobalt and zinc. Flow rates were on the order of 75 L/min for both systems. The treatment systems covered 4200 m^2 and 7000 m^2 and contained a series of soil berms installed across the wetland, and about a 30 cm layer of a mixture of peat and peat screenings (a waste material generated during the processing of horticultural peat). Although these systems have been successful in removing about 70 - 90% of the input metals, output nickel concentrations in one of the wetlands exceeded the discharge standard by as much as a factor of four. Average flow rates were greater than the design value by a factor of two, and the wetland was unable to adequately treat this flow volume. In 1993 and 1994, changes made to reduce the hydraulic gradient and minimize channeling improved performance, but nickel concentrations still exceeded permit requirements during periods of high flow and during the fall as temperature decreased. In 1995, the mining company constructed an additional 10,000 m^2 of wetland to provide additional treatment. After the expansion, the discharge was in compliance with permit requirements, until high flows and decreasing temperatures in the fall caused nickel concentrations to exceed standards.

Additional keywords: copper, cobalt, zinc, passive treatment, stockpile runoff

Introduction

Wetlands have been used to address a variety of water quality problems, including those arising from agricultural, municipal and industrial sources (Hammer 1989, Moshiri 1993). Wetlands have also been successful in treating coal mine drainage, and are being examined for their ability to treat metal mine drainage (Hedin and Nairn 1993, Eger et al. 1993, Wildeman et al. 1993). The use of wetlands to treat mine drainage is an attractive alternative to more conventional treatment methods. Wetlands are less costly to build than the conventional treatment systems, use processes which naturally occur in wetlands to remove metals from the water (e.g. adsorption, filtration), and offer a system that ideally should operate with little to no maintenance for extended periods of time.

LTV Steel Mining Company operates several taconite mines in northeastern Minnesota. At their Dunka Mine, located

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near Babbitt, surface drainage with elevated copper, nickel, cobalt, and zinc is generated as water infiltrates stockpiles of mineralized Duluth Complex (an igneous intrusive rock formation which contains disseminated copper and nickel sulfides). In the mid-1980's, LTV began an extensive program to evaluate various options for mitigating the problems at this mine. The company's preferred option was a combination of passive alternatives which would reduce flow emanating from their stockpiles, and which would use wetland treatment to remove metals from the resulting drainage. In 1988, four overland flow test cells were built to investigate methods to optimize metal removal and to provide design data for the ultimate implementation of wetland treatment at this facility. Based on the results of this study, two full-scale wetland treatment systems were built in the winter of 1992. This paper discusses the performance of those two systems.

Methods

General Construction

Each of the two treatment systems was built in existing wetlands. Construction began in the winter of 1992. The wetland systems were designed by STS Consultants, Ltd., and built by LTV Steel Mining Company (Frostman 1992).

Both areas were originally a combination of emergent (wet meadow) and scrub-shrub type wetlands, and the majority of the woody vegetation, which consisted primarily of alder, was removed from the site. The basic design for each system included the construction of a series of soil berms, which were built to control water levels and to maximize contact between the drainage and the substrate (Figure 1). Soil berms were built with glacial till (sandy silt) available from a surface overburden stockpile on the property. After the berms were constructed, a one-foot layer of a mixture of local peat and peat screenings was applied. The screenings are a waste material generated during the processing of horticultural peat and consist mostly of wood fragments and long peat fibers. This material was selected to increase the permeability of the peat to at least 10^{-3} cm/sec and to provide available organic carbon. In the spring of 1992, the berms were hand-seeded with Japanese Millet, while the open water areas were seeded with cattails. To obtain the cattail seeds, cattail heads were placed in a container of water with a small amount of liquid soap and several large bolts, and then the mixture was agitated until the heads broke and the seeds were dispersed. The slurry was then broadcast over the wetland.

W2D/3D system

This system covers 4200 m^2, contains 6 berms, and treats the drainage from two seepages which are associated with waste rock stockpile 8031 (Figure 2). One of the seeps is diffuse with an undefined channel (Site W2D), while only limited flow data exists for the other seep (Site W3D). The average flow from the stockpile has been estimated to be on the order of 75 L/min. From 1992-94, the input to the wetland was estimated to have an average pH of 7, with mean metal concentrations of 1.92 mg/L nickel, 0.05 mg/L copper, 0.05 mg/L zinc and 0.02 mg/L cobalt (Table 1).

W1D system

The majority of the flow to this system originates from the base of the 8018 stockpile, although additional seepage from the 8031 stockpile also drains to this area (Figure 2). V-notch weirs were installed to provide continuous measurement of the input and output flows. Annual average flows, from 1986-94, ranged from 75 - 150 L/min, with peak flows exceeding 750 L/min. Water quality samples of the inflow and outflow were colleted twice per month during the period of flow (generally March - December). Samples were collected within the system about once per month (Figure 2). From 1992-94, the input to the

Table 1. 1992-94 water quality data for the W1D and W2D/3D wetland treatment systems.

	W2D/3D input	W2D/3D output	W2D/3D standards	W1D input	W1D output	W1D standards
pH	7.0	7.0	6.5-8.5	7.1	7.1	6.5-8.5
Copper	0.050	0.006	0.023	0.068	0.010	0.023
Nickel	1.92	0.124	0.213	3.94	0.38	0.484*
Cobalt	0.017	<0.001	0.050	0.032	0.010	0.050
Zinc	0.054	<0.01	0.343	0.051	0.012	0.343

All Concentrations are in mg/L except pH, which is in standard units.
* Nickel standard increased from 0.213 in 1995.

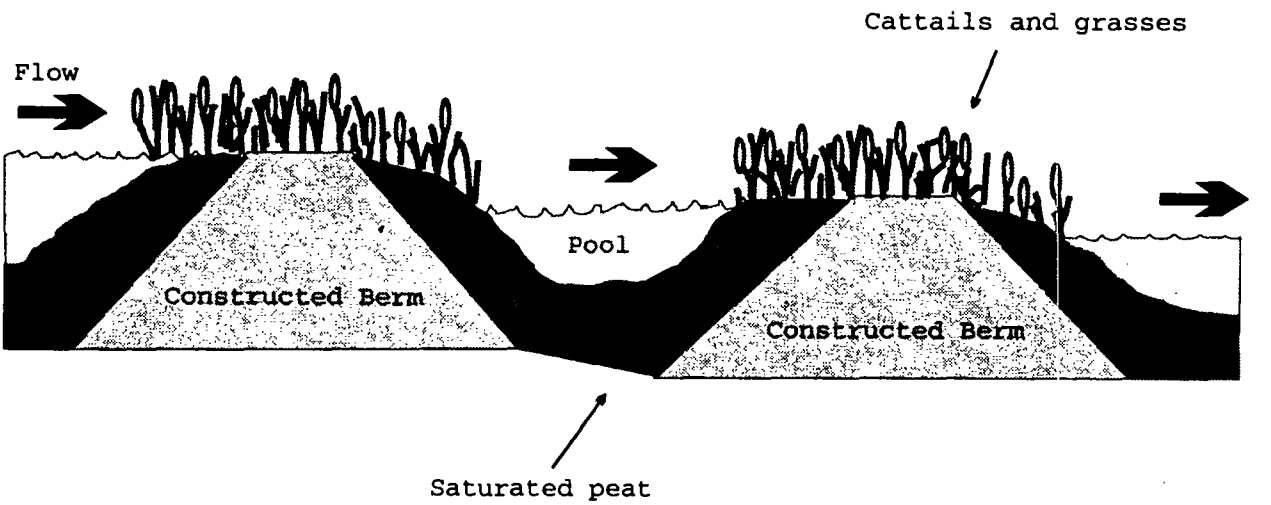

Figure 1. Cross section of typical berms (schematic).

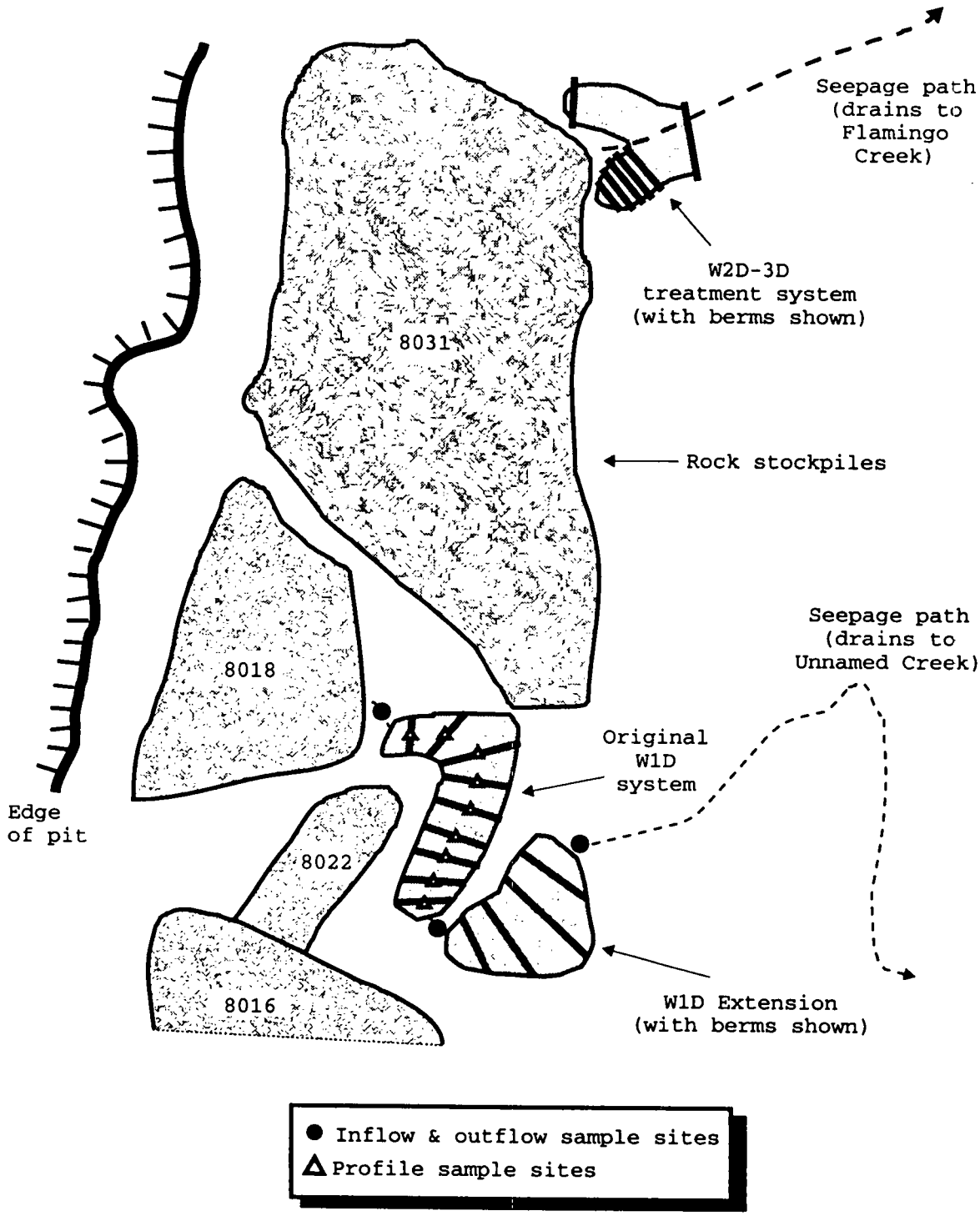

Figure 2. Dunka Pit site map showing locations of the W1D and W2D/3D wetland treatment systems and surrounding waste rock stockpiles.

wetland had an average pH of 7.1, and contained 3.94 mg/L nickel, 0.07 mg/L copper, 0.05 mg/L zinc and 0.03 mg/L cobalt (Table 1).

The original W1D treatment system, constructed in the spring of 1992, covered 7000 m^2 and contained a series of 9 berms. In 1993 and 1994, changes were made to the system to disperse flow, minimize channeling and improve contact between the drainage and the substrate. In 1995, the system was expanded by 10,000 m^2, and included an alternating series of overflow and underflow berms (Figure 3). Prior to construction of these berms, the original organic soils were removed; the berms were then built on the mineral soil base and compacted to minimize any future settling.

Results

This paper will focus on the results from the W1D system. This site has higher flows, higher concentrations of trace metals, and a steeper slope than the W2D/3D system. The W2D/3D system has been in compliance with permit requirements since it was constructed in 1992 (Table 1).

Limited data collected prior to construction indicated that the metal concentrations were reduced to acceptable levels in the natural wetland even before the system was constructed (Lapakko and Eger 1987). The construction of the system and the addition of new substrate has made the system more efficient and increased the length of time the system should be capable of providing treatment.

W1D Flow

For the period 1992-94, the input flow rates were similar to the long term average for the site, with average annual flows of 110 - 150 L/min. Daily input flows ranged from 0 in the winter to around 750 L/min during periods of heavy precipitation. In general, outflow was greater than or equal to inflow, except during those periods of the summer when precipitation was low.

Water quality

Overall, the wetland was effective in reducing metal concentrations. All metal concentrations decreased, ranging from 70% for cobalt to 90% for nickel. Outflow levels of copper, cobalt, and zinc consistently met water quality standards (Table 1). Despite an overall 90% decrease in concentration, nickel exceeded the water quality standard, particularly during high flow periods in the summer and when temperatures decreased in the fall (Figure 4). Nickel concentrations to less than 0.2 mg/L after the size of the system was increased in the spring of 1995. Outflow concentrations were in compliance until October, when concentrations exceeded the effluent limit of 0.484 mg/L (Figure 4).

Mass Removal

Overall metal mass removal in the wetland, from 1992-94, has ranged from 4.5 kg for zinc to 453 kg for nickel. These correspond to areal rates of removal ranging from 1 to 96 $mg/m^2/day$ (Table 2).

Discussion

Trace metal removal in a wetland is influenced by a large variety of physical, chemical and biological processes (Hammer 1989). Physical processes, such as filtration and sedimentation, are important in removing particulate metals, while it is the chemical and biological processes that provide for the removal of dissolved metals. The majority of trace metal removal in wetland treatment systems is associated with the substrate (Wildeman 1993, Gersberg 1984). Data from the test cells constructed at the Dunka mine, indicated that over 99% of the metal removal was associated with the peat, with only about 1% being removed by the vegetation (Eger et al. 1994).

Table 2. Mass removal and areal removal rates, W1D wetland treatment system 1992-94.

	Mass Removed (kg)	Areal Removal Rate * (mg/ m^2 /day)
Copper	6.1	1.3
Nickel	453	96
Cobalt	6.7	1.4
Zinc	4.5	1.0

* Calculated by dividing the total mass removed by the area of the wetland and the number of days of flow

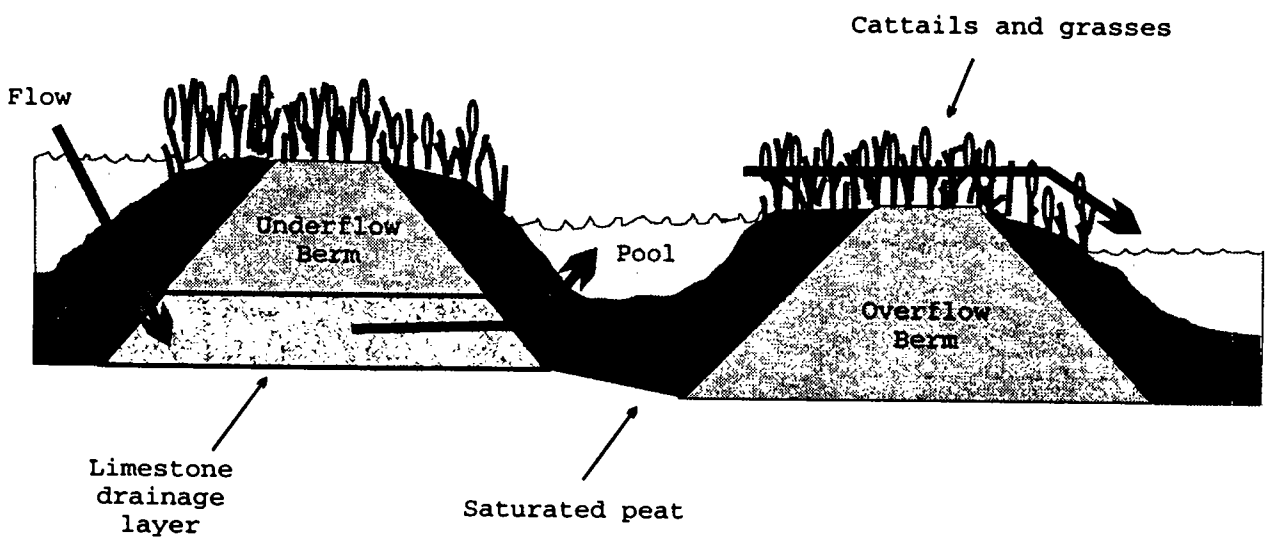

Figure 3. Cross section of an underflow/overflow berm system (schematic).

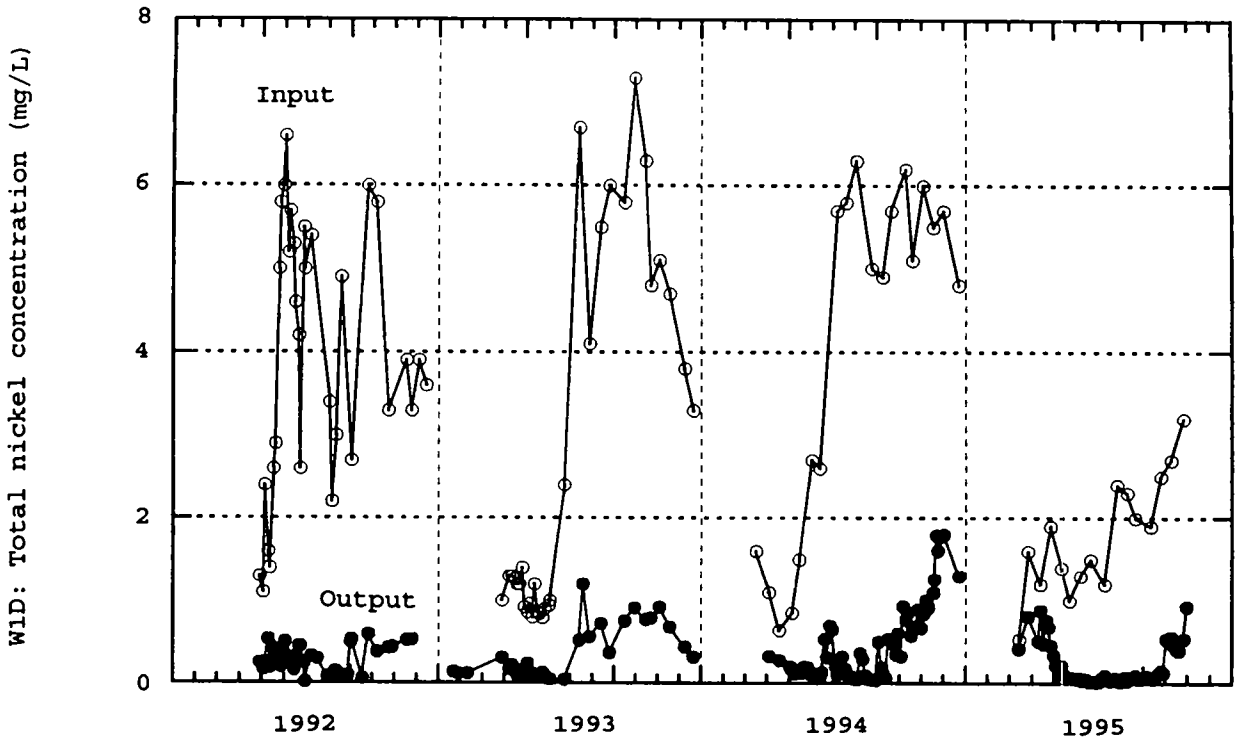

Figure 4. W1D input and output nickel concentrations, 1992-1995.

(Vegetation can enhance overall removal by dispersing flow, increasing the hydraulic conductivity of the organic substrate, and providing a source of new organic material.) Therefore, any full-scale system should be designed to maximize the contact of the drainage with the substrate. Metal removal in the test cells with a water depth of 5 cm was significantly greater than in those where the water depth was 15 cm (Eger et al. 1993). Removal in the test cells also increased as residence time increased, with a minimum of two days needed to achieve optimum removal.

Important factors in the design of a wetland treatment system to remove metals include detailed characterization of the drainage, the effluent standards that must be met, and performance data for the type of wetland to be constructed (Eger and Melchert 1992). Two of the most important design parameters are the residence time and the rate of metal removal in the wetland. Both of these factors are needed to determine the appropriate size of the wetland system.

Using the results from the test cells, an initial size for the W1D system was determined (Eger and Melchert 1992). The required treatment area was calculated based on: 1) the minimum required residence time needed to achieve compliance with permit standards and 2) the measured rate of metal removal expressed per unit area (Table 3). Since the area of the wetland must be of sufficient size compliance with permit standards to satisfy both the residence time and metal removal criteria, the

Table 3. Comparison of the actual constructed wetland size to the site required based on initial design calculations, for the W1D and W2D/3D systems.

	W2D/3D Design Values	W2D/3D[a] Actual Values 1992-94	W1D Design Values[b]	W1D Actual Values 1992-94
pH	7.0	7.0	7.1	7.1
Nickel (mg/L)	2	1.92	5.4	3.9
Flow (L/min)	75[c]	75[c]	75[d]	150
Required Size (m^2)	3700	4200	6300(12,600)[e]	7000(17,000)[f]
• Based on Flow	3700	NAp	3700 (7400)[e]	NAp
• Based on Metal loading	2500	NAp	6300 (12,600)	NAp

a: Concentration values are based on samples from site W3D
b: Average Values, 1980-91.
c: Estimated.
d: Average, 1990-91.
e: Value in parentheses based on long term flow value, 1986-1991
f: Wetland site was expanded to 17,000 m^2 in the spring of 1995
NAp: Not applicable

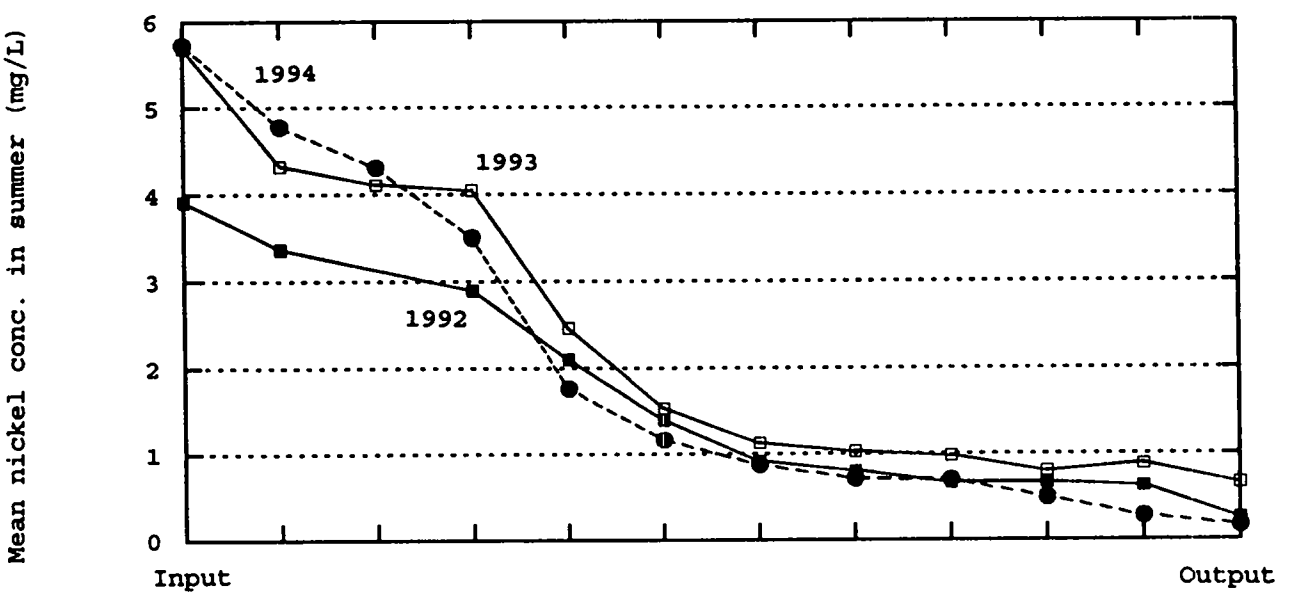

Figure 5. Nickel concentrations within the W1D wetland treatment system during the summer periods of 1992-1994. The values shown for each site are the means of the available summer data.

larger of the two calculated areas should be used as the basis for the design. Assuming an average input flow of 150 L/min (long term average), a residence time of two days, water depth of 5 cm, and good flow dispersion, the wetland size based on residence time was 7400 m^2, and 12,600 m^2 based on the rate of metal removal. Since these calculations use only the average flow, additional area or storage should be included if higher flows are to be treated adequately. Table 3 compares the size of the original W1D system and the W2D/3D system with the initial estimates made from the test cell data.

The initial size of the W1D system was substantially smaller than the area calculated from the test cell data, while the size of the W2D/3D system exceeded the calculated area. There were two reasons for the reduced size of the original W1D system. Since the company was modifying a natural wetland to provide treatment, the Minnesota Pollution Control Agency wanted to minimize the extent of wetland disturbance. The company had also made some changes to the watershed around the stockpiles. They believed that the average flow for the two years immediately preceding construction, although lower than the long term average, was representative of future flow and they based their design on this assumption. If the average input flow had been 75 L/min, the area would have satisfied the criteria developed from the test cells.

The inability of the W1D wetland to produce water which consistently satisfies the effluent requirements is primarily due to its small size and inadequate flow dispersion. Performance is particularly poor during periods of high flow. Flow channels have developed in the wetland, particularly in the lower half of the system, decreasing the contact between the drainage and the substrate. The average measured flow during 1992 to 1994 was essentially the same as the long-term average, and as a result the wetland was undersized by about a factor of two. Water samples collected from within the wetland indicated that metal removal was particularly low in those areas with significant channelization (Figure 5). Dye studies confirmed the lower contact and shorter residence time in this section (McCarthy et al. 1994).

Once the system was enlarged, concentrations decreased to levels below standards until October 1995, when concentrations increased dramatically and exceeded discharge limits (Figure 4). Unusually large rains (13 cm) at the end of September dramatically increased inflow at a time when wetland performance typically decreases due to decreasing temperatures. Additional data will be collected and analyzed to better determine the cause of the lack of treatment efficiency during this time.

Conclusions

Wetland treatment has been successful in reducing trace metal concentrations in stockpile drainage by over 90%. Design criteria derived from test cells provided a reasonable estimate of required wetland size. Measured rates of metal removed expressed per unit area of wetland ranged from 1 $mg/m^2/day$ for zinc to 96 $mg/m^2/day$ for nickel. Enlarging the W1D system improved metal removal and generally produced effluent which met water quality standards. Additional work is planned to investigate the poor performance of the system in the fall of 1995, metal removal within the wetland, and the total metal removal capacity of the wetland.

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TREATMENT OF METAL CONTAMINATED GROUNDWATER IN PASSIVE SYSTEMS: A DEMONSTRATION STUDY[1]

by

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Abstract. A pilot scale field test on passive treatment systems for metal and sulfate removal from ground water and surface runoff, is being conducted in Butte, Montana. This study is a collaborative effort between Montana Tech and the Atlantic Richfield Company (ARCO) to find solutions for treating metal contaminated water (e.g. Cu, Zn, Cd, and Fe). The field demonstration systems include anaerobic horizontal flow gravel bed wetlands with plants, horizontal flow gravel/organic substrate wetlands with plants, and vertical upflow gravel/organic substrate system without plants. The experimental design for sulfate and metal removal will be presented as well as the hydraulic design criteria used. Results from this test will be used to estimate system economics, long term performance and the life expectancy for these systems. Studies on mesoscale passive treatment systems such as 1.5 m high vertical upflow gravel/organic substrate columns are also presented. The parameters describing system performance are: hydraulic conductivity, volumetric sulfate-reduction rate, and metal immobilization rate.

Add. key words: Constructed wetlands, metal remediation, sulfate-reducing bacteria, field study.

Introduction

Acid mine drainage affects natural waters by decreasing water quality, stream aesthetics, and and the productivity of fisheries and other aquatic biota.

Natural and constructed wetlands have been used for many decades mainly in Europe, as a low cost, low maintenance waste water treatment system. In recent years constructed wetland systems have been adapted to treat AMD in the U.S. Passive anaerobic treatment of AMD was successful in immobilizing metals and increasing pH of the water (Wildeman, 1992).

One of the aims for the remediation effort in Butte, Montana (USA) is the emplacement of wetlands for the treatment of storm water runoff and near surface groundwater from the Metro Storm Drain (MSD) (Figure 1). Suspended and dissolved metals are being transported with the MSD water into Silver Bow Creek and eventually into the Clark Fork River. This research project is part of the ARCO remediation activities in the Clark Fork River Basin, in Southwestern Montana.

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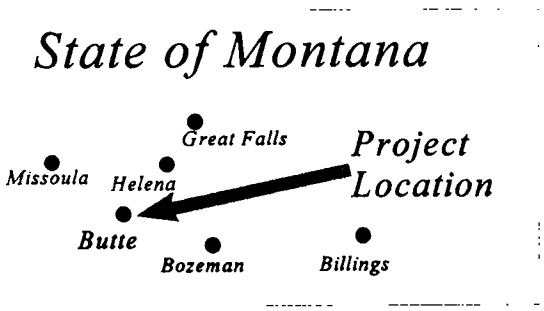


Figure 1. Location of the passive treatment demonstration site.

GOALS AND OBJECTIVES OF THE DEMONSTRATION STUDY

The overall goal of this demonstration study is to demonstrate the effectiveness of passive treatment systems for metal contaminated groundwater in Montana's alpine climate. Within this demonstration study the capabilities of three different passive treatment system designs are being evaluated based on hydraulic, chemical, microbial, and biological performance. An excessive monitoring system is in place to monitor hydraulic conductivity, water chemistry, and solids composition within the cells as well as in influent and effluent for each of the systems. Mechanistic and empirical models are being developed to increase our understanding of the fundamental processes occurring within these systems, including system performance, economy, and life expectancy. This information is essential for up scaling to a larger full scale operation.

MECHANISMS FOR METAL REMOVAL IN ANAEROBIC WETLANDS

Abiotic Processes

A pH increase in the aqueous phase due to the acid neutralizing capacity of the substrate can cause a consequent precipitation of dissolved metals and nonmetals. These pH induced precipitations can also indirectly cause other dissolved species to be removed from the aqueous phase. All of these pH initiated processes depend upon the acid neutralizing capacity, i.e. the alkalinity, of the substrate. If the alkalinity of the substrate will not be sustained, pH induced processes are short term.

Biotic Processes

Biotic initiated processes in anaerobic wetlands are the only long term self sustaining processes that will have a significant effect upon the effectiveness of the wetlands in treating acid metal and sulfate containing waters (Bolis *et al.* 1992). Sulfate reducing bacteria (SRB) will become active and produce hydrogen sulfide when exposed to a low redox potential (<-100 mV) and the presence of other required nutrients, such as short chain organic carbon compounds, nitrogen, phosphorous, and other required trace elements (Widdel, 1988). The presence of sulfide in anaerobic environments results in the formation of the corresponding metal sulfides which are extremely insoluble (Table 1) (Lawrence and McCarty, 1965). Many free metal ions can be inhibiting or toxic to the microbial consortium, whereas precipitated metal sulfides do not have a negative effect on microbial and acivity (Mueller and Steiner, 1992). These metal sulfide precipitates remain stable in anaerobic environments.

In most constructed wetlands for AMD a temperature between 4 and 20 °C is expected. Hence, the rate of microbial conversion will be slow and hydraulic residence times should be in the order of days to achieve sufficient microbial sulfate reduction from a complex substrate.

The microbial processes occurring in a natural or a constructed wetland are summarized in Figure 2. Complex organic carbon sources such as composts, or wood waste products are inexpensive and common energy sources. The carbon and energy inputs into the wetland system are present either from an external source (e.g., compost) or from plants within the system. This complex organic carbon is decomposed and converted microbially into suitable carbohydrates for fermentative organisms. Fermentative organisms

Table 1. Solubility of Heavy Metal Sulfides at 18 °C (Lawrence and McCarty, 1965).

Sulfide Salt	Solubility prod.	Solubility (mg/l)
CuS	$8.5\ 10^{-45}$	$9\ 10^{-18}$
Cu_2S	$2.0\ 10^{-47}$	$3\ 10^{-11}$
PbS	$3.4\ 10^{-28}$	$4\ 10^{-9}$
ZnS	$1.2\ 10^{-23}$	$3\ 10^{-7}$
FeS	$3.7\ 10^{-19}$	$5\ 10^{-5}$

further degrade these compounds into shorter chain volatile fatty acids and hydrogen that can be utilized by sulfate reducing bacteria. In the absence of sulfate or in the presence of inhibitors specific for sulfate reduction, methane could be produced (Loveley *et al.* 1982). Hence, a high sulfate concentration is desired to effectively suppress methane production.

System Design

Surge pond

The following is descibing design of systems presented in Figure 3 which is currently under construction. For most applications a surge pond or equalization pond is required to equalize water flow and water chemistry in the inlet water to the treatment systems. Another important function of the surge pond is solid settling. Since only minor fluctuations in water chemistry were observed for the MSD water and only a fraction of the total water flow is being treated, the main design criteria for the surge pond were its settling characteristics and its solid accumulation rate. Additionally the pond must hold a sufficient supply of MSD water to supply water to the treatment systems during a dry period.

Sediment settling tests in a 1.2 m high settling column were performed. After 30 hours a total suspended solids (TSS) reduction of 90% was observed. Multiplying by a safety factor of 3 and adjusting to a temperature of 2°C for increased water viscosity the resulting hydraulic retention time was 6.4 days.

Anaerobic treatment systems

The paradigm for successful treatment in a passive system includes three major components. The flow characteristics must be maintained within prescribed boundaries throughout the life of the system, the specific microbiological environment must be maintained, and heavy metals must be immobilized and permanently removed from the water phase. If any of these requirements are not met the system will be non functional with regard to effective metal removal.

The total required sulfate reduction may be calculated from the following based on metals and acid loading of the water (Eger, 1992):

$$M_{SO4} = \sum [Me^{2+}] + 1.5 \sum [Me^{3+}] + 0.5\,10^{-pH}$$

The volume of the constructed wetland has to provide sufficient sulfate removal to immobilize the metal loading and neutralize the acid loading. The required sulfate reduction rate (SRR) is dependent on flow rate and the total required sulfate reduction for metal removal in the contaminated ground water.

$$SRR = F \times M_{SO4}$$

The volumetric sulfate reduction rate (VSSR) is driven by the organic fraction of the system and can be calculated after the sulfate reduction potential (SRP) for a chosen substrate has been experimentally determined.

$$VSRR = SRP \times f_{co}$$

The total wetland volume (V_t) can then be calculated:

$$V_t = \frac{SRR}{VSRR}$$

Where:

F: flow rate [m^3/d];

f_{Co}: volumetric fraction of compost in the system [-];

SRP: sulfate reduction potential per volume substrate [$mole/m^3/d$];

M_{SO4}: stoichiometric amount of sulfate required to be reduced for complete metal immobilization and acid neutralization [mol/ m^3];
SRR: sulfate reduction rate for total metal immobilization and acid neutralization [mol/d];
VSRR: volumetric sulfate reduction rate per total system volume [mol/m3/d];
V_t: total wetland volume including inert gravel [m^3];

The effectiveness of the sequestering will impact the operational lifetime of the wetland. Information will be obtained by taking intact core samples and analysing the rate of metal accumulation in the systems, the form of the sequestered metals, and their stability.

Hydraulic considerations

The hydraulic residence time (HRT) of a gravel bed system can be calculated when the porosity (P) of the media is known:

$$HRT = \frac{V_t \times P}{F}$$

Hydraulic conductivity (K) is a critical parameter for a flow through a porous media and for the design of any wetland system. Most failures in the field have occurred because of inadequate knowledge of the short term and long term consequences of flow characteristics. K can be determined from Darcy's Law:

$$F = A_c \times K \times \frac{dh}{dl}$$

Where:
HRT: hydraulic residence time [d];
P: porosity [-]
A_c: crossectional area [m^2]
dh/dl: hydraulic gradient [-]
K: hydraulic conductivity [m/d]

There are several limits that will constrain the engineering choices that will be available. There will be a minimum allowable hydraulic conductivity, or alternatively for a fixed flow, there will be a maximum allowed hydraulic loading to the system to avoid fracturing, channeling, or short-circuiting around the wetland. As the fraction of organic material approaches zero (100 % gravel system) the fuel for the biological activity is eliminated, and this may adversely affect the microbial performance and the resulting metal removal efficiency.

Test Cell Dimensions and Layout

The tested passive treatment systems are two anaerobic gravel substrate wetlands sustaining horizontal flow of MSD water (Type 1 and 1A), an anaerobic organic compost - limestone gravel substrate wetland sustaining horizontal flow of MSD water (Type 2) and an anaerobic organic compost - limestone gravel substrate passive system sustaining upward flow of MSD water (Type 3) (Figure 3). The Type 1A wetland will have identical substrate as Type 1 but will be sized 50% smaller by volume. Each of the test systems are receiving MSD water from the surge pond. The effluent from each of these primary treatment systems passes through an aerobic wetland, designed to further, reoxidize and to remove iron and excess organic materials that have leached out from the compost systems. The surge pond effluent is distributed via underground pipes into each of the anaerobic test cells at a flow rate of 27.2 m^3d^{-1} to each unit. The effluents of cell 1 and 1A are combined and treated in an aerobic wetland. The effluents of cell 2 and 3 are combined and treated in a separate aerobic wetland. The effluents from the two aerobic wetlands are combined and discharged back into the MSD channel. Cell dimensions, hydraulic residence time, substrate/gravel composition, and the hydraulic conductivities are summarized in Table 2.

Complex Organic Substrates

Fermentative Organisms

Volatile Fatty Acids + HCO_3^- + H_2

Sulfate Reducing Bacteria ← SO_4^{-2}

$[HCO_3^-]$; $[OH^-]$; $[H_2S]$; $[CO_2]$

- In the absence of sulfate, methane is produced

<u>Metal Sulfide Precipitation</u>

$H_2S + Me^{2+}$ $MeS + 2H^+$

Figure 2. *Metabolic Pathways in anaerobic wetland systems.*

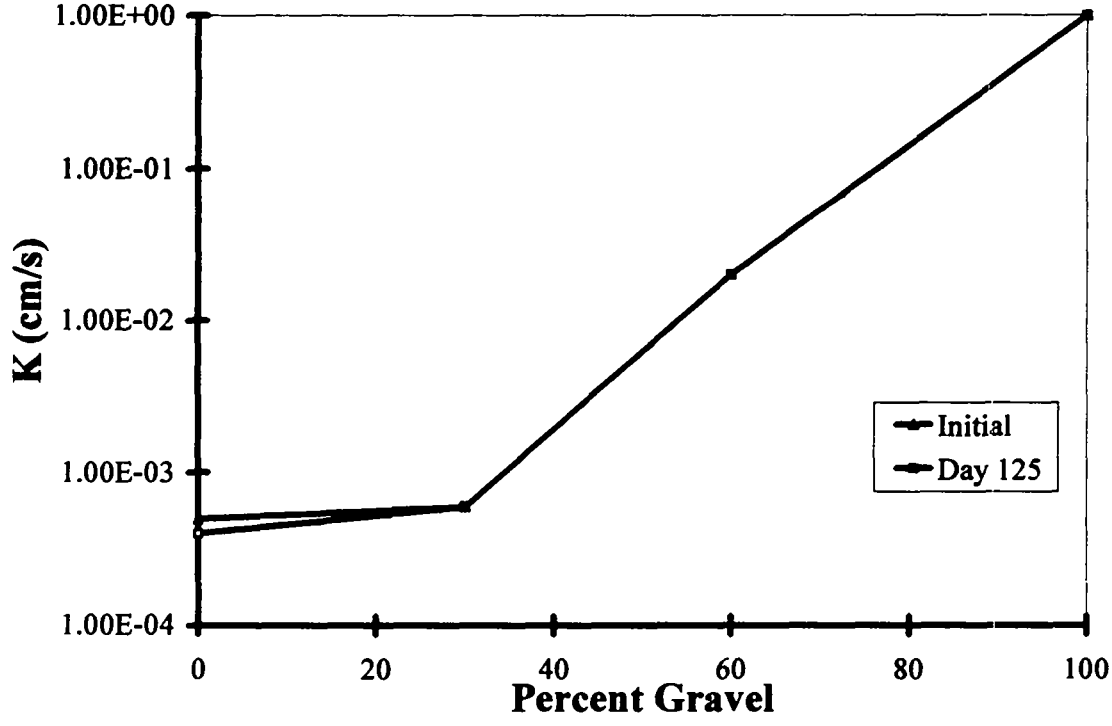


Figure 4. *Changes in hydraulic conductivity over time.*

Table 2. Design parameters for the anaerobic and aerobic test cells.

Unit Cell#	HRT [d]	flow [m³/d]	lxwxd [mxmxm]	vol m³	porosity [-]	compost/gravel [%/%]	hydraulic conductivity [cm/s]
Pond	7.0	154	20x36x1.5	1080	-	-	-
An 1	10.0	27.2	31x31x0.75	721	0.38	0/100	1-10
An 1a	5.0	27.2	22x22x0.75	370	0.38	0/100	1-10
An 2	6.0	27.2	14x27x1.2	453	0.36	20/80	0.1-0.5
An 3	1.5	27.2	9x9x1.5	122	0.34	50/50	0.01-0.06
A 1	2.8	54.5	28x8x0.7	157	NT	soil mix	NT
A 2	8.0	54.5	49x13x0.7	446	NT	soil mix	NT

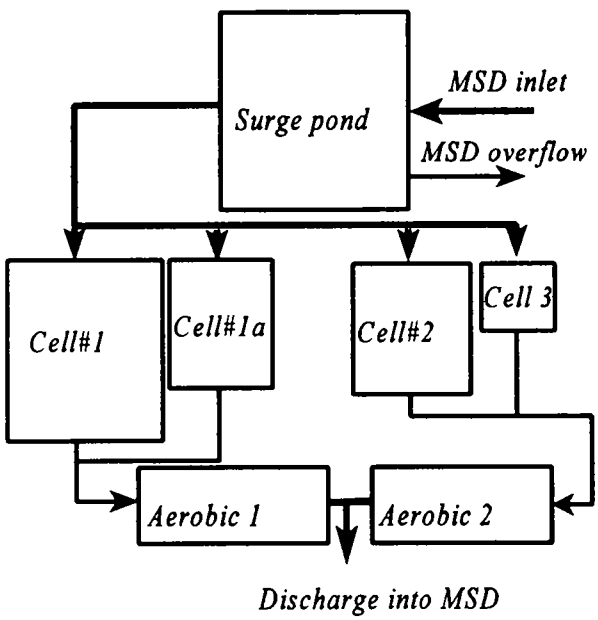


Figure 3. Schematic layout of the demonstration system.

Laboratory Results with Anaerobic Upflow Columns

Hydraulic Performance

Four 1.5 m high and 25 cm diameter upflow columns were operated over 6 months to test for hydraulic performance, sulfate reduction capacity, and metal removal efficiency. These tests columns were identical in height to the actual field system. Different ratios of compost to gravel were placed in each of the columns (Table 3). The hydraulic residence time was set at 1.5 days in all four columns. The hydraulic conductivity decreased significantly with a higher compost content in the columns. Table 3 indicates the

Table 3. Hydraulic conductivity values (K) measured in various gravel/compost mixt substrate.

C#	RG [%]	CG [%]	compost [%]	K [cm/s]
1	0	0	100	$7.8\ 10^{-4}$
2	10	20	70	$7.4\ 10^{-4}$
3	40	20	40	3.210^{-2}
4	80	20	0	1.0

dependence of hydraulic conductivity on substrate composition. The hydraulic conductivity was a function of substrate composition ranging from 100% compost to 100% gravel. A higher gravel fraction increased the hydraulic conductivity, causing the hydraulic conductivity value to asymptotically approach the performance of a pure gravel system as the pore size created by the gravel dominated the flow regime. Changes in hydraulic conductivity during 4 months of operation were not statistically significant for any of the tested

substrates (Figure 4).

Table 4. Chemical Composition of the Organic Compost Used in this Study

Constituent	value
pH (1:4 DI water)	6.7
alkalinity	23 mg $CaCO_3$/g dry weight
moisture	57 % by weight
organic content	80 % by dry weight
total organic carbon	46 % by dry weight
total Kjeldahl nitrogen	1.6 % by dry weight
SIN*	1713 ppm per dry weight
C/N - ratio	29.7
phosphate-P	276 ppm per dry weight
Cu	28 ppm per dry weight
Cd	<1 ppm per dry weight
Zn	114 ppm per dry weigh
Fe	272 ppm per dry compost

SIN: soluble inorganic nitrogen ($NH_4^+ + NO_3^- + NO_2^-$).

Chemical and microbial performance

Organic Substrate. To screen for a source of organic substrate to be used in an anaerobic passive treatment system for AMD the following criteria were applied: The total organic carbon content in the substrate should be high, the metal content should be low (most of the metals present will ultimately contribute to the overall metal load into the system), the buffering capacity should be high to raise the pH of the AMD, and other required nutrients for SRB growth and metabolism should be present at the required concentration ratios, e.g. carbon:nitrogen:phosphorous at approximately 100:10:1 by weight (Widdel, 1988). However, leaching of nutrients from the organic substrate should be relatively slow in order to avoid discharging nutrients and to supply a long lasting nutrient source for SRB and subsequent metal removal. The organic substrate used was a mixture of composted sewage sludge and wood chips.The chemical composition of this compost is presented in Table 4. Within approximately 10 residence times, the chemical oxygen demand (COD) dropped to a level of <100 ppm (eq. to a BOD of 40 ppm).

Sulfate reduction and Metal removal. A significant reduction in sulfate was observed after approximately 10 residence times in the upflow columns (Figure 5). On average a reduction rate of 100 to 200 mmol Sulfate per m^3 compost and day were observed. For comparison, a sulfate removal rate of 300 mmoles sulfate per cubic meter of compost per day by an active SRB consortium was observed by Wildeman (1993).

Metal removal was observed as soon as operation started. The initial removal of metals was most likely due to adsorption onto particulate organic matter (Angelidis and Gibbs, 1989). Metal concentrations in the effluent of the various gravel/compost columns were similar for all columns containing compost, and the metal removal efficiencies were on average 99 % for zinc, 90% for copper and 90% for cadmium (Table 5). Copper and cadmium were present at much smaller concentrations in the MSD water and the removal rate was defined by the analytical detection limit. Zinc concentration measurements

Table 5. MSD water chemistry before and after treatment in anaerobic vertical upflow columns

Constituent	MSD water (Influent)	Column effluent	Units	treatment efficiency %
Cu	80-900	10-70	µg/l	88-99
Cd	40-120	10-40	µg/l	75-92
Zn	8000-11000	90-250	µg/l	97-99
Mn	8500-9000	8000-9500	µg/l	0
Fe	40-100	900-1000	µg/l	0
pH	6.6-6.8	6.8-7.0	-	-
Sulfate	127-270	119-255	mg SO4-S/l	ND
TSS	5-2500	NT	mg/l	-

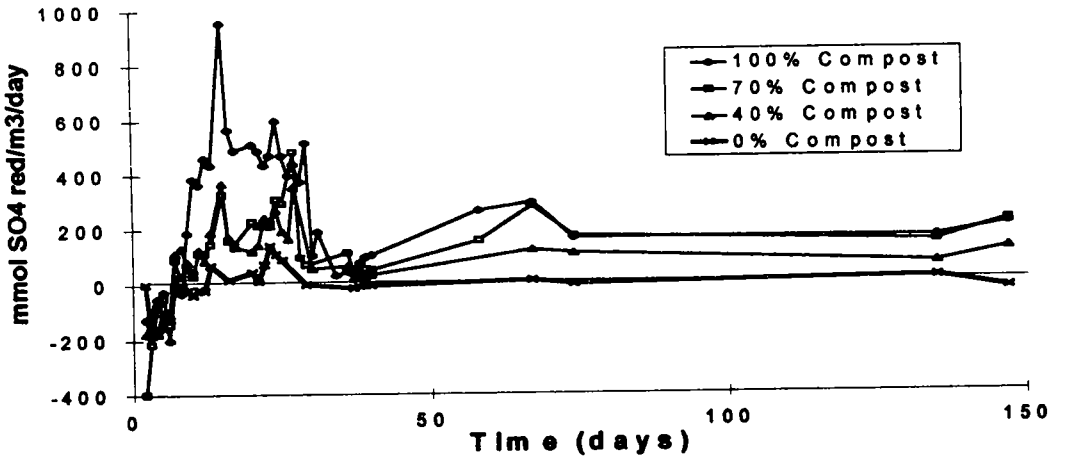
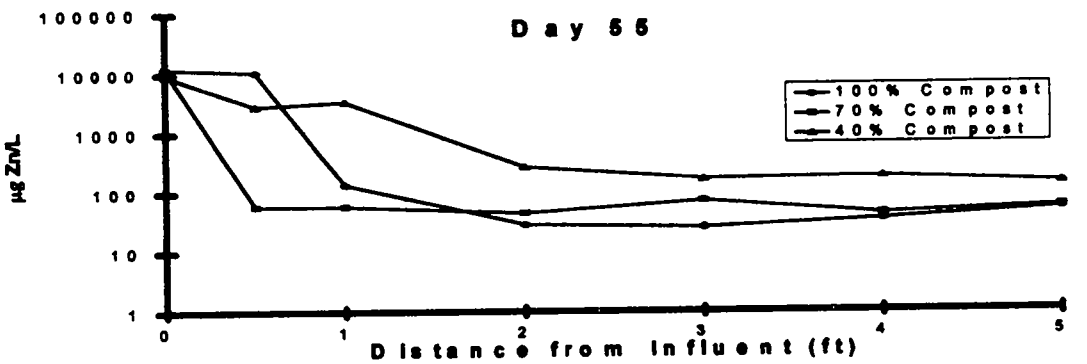
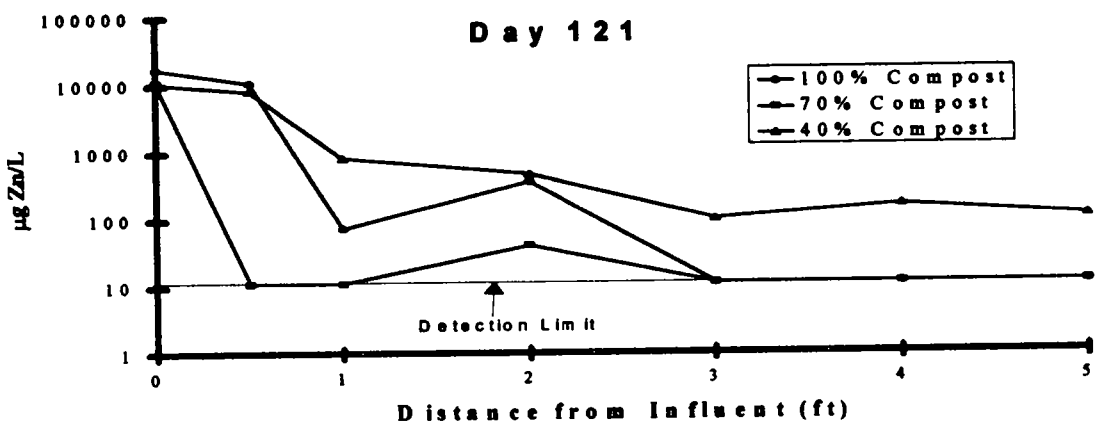


Figure 5. Sulfate removal rate was plotted over time. Significant sulfate reduction was observed in all columns with organic content.



(a)



(b)

Figure 6. Zinc concentrations over the lengths of the columns at day 55 (a), and day 121 (b). Zinc was removed faster in the columns with higher organic compost fractions.

were taken at various location of the column after 3 month of continuous operation. A shorter column length was necessary to remove zinc for the columns with a higher organic compost fraction (Figure 6). Little or no metal removal was observed in the 100% gravel columns (data not presented).

Conclusions

Laboratory experiments were used to design field test systems for metal removal from contaminated ground and surface water in Butte, Montana. The design calculations were based on the metal and acidity load of the MSD water, the potential sulfate reduction rate, the hydraulic conductivity for the chosen substrate (as observed in laboratory experiments), and on the total flow rate. The field system is being brought on line in May of 1996.

Successful zinc, copper, and cadmium removal were achieved in 1.5 m high upflow mixed substrate columns. The long term removal of these metals were based on the microbial sulfate reduction within the column substrate.

The hydraulic conductivity of the substrates controlled the flow regime of the upflow columns. Hydraulic conductivity was increased successfully by increasing the gravel/compost ratio.

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SESSION 10: MINED LAND REFORESTATION

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AMERICAN SOCIETY • SURFACE MINING and RECLAMATION •

THE USE OF LEGUMINOUS TREES IN RECLAMATION OF TROPICAL MINED SOILS[1]

by

Luiz E. Dias[2]; Avílio A. Franco[3]; Eduardo F.C. Campello[3] and Sérgio M. de Faria[3]

Abstract

Characterized as vigorous pioneer species which produce large amounts of biomass, forest leguminous trees have shown promise in studies of degraded soil rehabilitation. When associated with atmospheric nitrogen-fixing bacteria and mycorrhizal fungi, these species show superior utilization of nutrients and growth under adverse soil conditions. The objective of this paper is to present the principal results obtained from the Research Program for Use of Nodulated and Mycorrhizal Tree Legume Species in Rehabilitation of Degraded Areas, developed by National Research Center for Agrobiology-CNPAB/EMBRAPA in conjunction with the Federal University of Viçosa, Brazil. The program has been developed in four subprograms: 1. Field surveys to collect and identify native leguminous tree species with potential for land reclamation; 2. Selection to identify more efficient nitrogen-fixing bacteria for each potential species; 3. Greenhouse experiments to evaluate nutritional requirements and capacity to grow in high density soils; and 4. Field experiments in lands degraded by mining.

Additional Key Words: leguminous trees; nitrogen fixation; bauxite mining; land reclamation

Introduction

Brazil is the world's leading producer of iron ore and a major exporter of aluminum, gold, niobium and tin. Totaling U.S.$6.6 billion, its mining output constitutes almost a third of Latin America's mineral-derived revenue (Project Survey 1993). A large part of these minerals, such as iron, gold, tin, bauxite, manganese, copper and kaolin, lies under biologically diverse forests in the Amazon Region, (Griffith et al. 1996).

Human activities such as surface mining have resulted in great disturbance to original soils. Among the different aspects of degradation, loss of soil organic matter is one of the most important (Franco et al. 1995a). The absence of organic matter contributes to low nutrients levels, low water holding capacity, and compaction problems. These situations may cause great difficulties when attempting to rehabilitate these areas. Nitrogen, sulphur and phosphorus are the main factors affecting soil fertility if organic material is lacking. Therefore both biological

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nitrogen fixation and mycorrhizal associations play very important roles in plant growth in such situations (Franco et al. 1991).

Characterized as vigorous pioneer species which produce large amounts of biomass, forest legumes have shown exceptional qualities in studies of rehabilitation of degraded soils (Franco et al. 1994, Dias et al. 1995 and Franco et al 1995a). Estimates from Brazil's arid "caatinga" region indicate that the "sabiá" tree (*Mimosa caesalpinifolia*) returns 5.8 $t.ha^{-1}.year^{-1}$ of vegetative material to the soil (Suassuna, 1982). Others researchs carried out in Central America with *Erythrina* (Glover & Beer, 1986) and in India with *Leucaena leucocephala* (Sandhu et al. 1990) showed contributions from 13 and 10 $t.ha^{-1}.year^{-1}$, respectively.

When associated with atmospheric nitrogen-fixing bacteria and mycorrhizal fungi, forest legumes present superior utilization of nutrients and growth under adverse soil conditions. For the last thirteen years, several large surveys of nodulation for native legume trees have been conducted in many acidic soils of Brazil (Magalhães et al. 1982, Faria et al. 1984, 1987, 1989, Moreira et al. 1992, 1993). Several experiments have been conducted to screen rhizobial strains for the most promising species (Franco & Silva, 1985 and Faria & Franco, 1993). Tropical legume trees have shown great dependence on VA-mycorrhizae (Mendes Filho, 1985 and Monteiro, 1990). Therefore, after inoculation with these microorganism the leguminous trees could absorb more water and nutrients such as phosphorus and zinc (Lambert et al. 1979).

The objective of this paper is to present the principal results obtained from the Research Program for Use of Nodulated and Mycorrhizal Leguminous Tree Species in Rehabilitation of Degraded Areas, developed by the National Research Center for Agrobiology-CNPAB-EMBRAPA in conjunction with the Federal University of Viçosa, Brazil.

Research Program for use of Nodulated and Mycorrhizal Leguminous Tree Species in Rehabilitation of Degraded Areas

This research program basically is divided into four subprograms:

1. Field surveys to collect and identify native species of tree legumes with potential for land reclamation;
2. Selection to identify more efficient nitrogen-fixing bacteria for each potential species;
3. Greenhouse experiments to evaluate nutritional requirements and capacity to grow in high density soils;
4. Field experiments in lands degraded by mining.

1. Collection and identification of native tree legumes.

In the last twelve years, researchers of CNPAB/EMBRAPA have examined more than 600 legume species for nodulation ability (Faria & Franco, 1993). The surveys were done in differents parts of the country, especially in the "cerrado", and other phytogeographic regions in the North, Northeast and Southeast. The surveys consist in observation of young plants to

identify the presence or absence of nodules. Later, the nodules are evaluated by acetylene reduction to verify activity.

Nodulation was observed in 392 of 688 species that were studied. The species belong to different subfamilies. Nodulation was most prevalent in the *Mimosoideae* (147 of 192 species) and *Papillionoideae* (158 of 219 species) and least prevalent in the *Caesalpinioideae* (38 of 183 species) and other subfamilies (49 of 94 species). This was the first report of nodulation for 172 species in 17 genera, and no-nodulated for 118 species in 18 genera.

Nodules obtained by the surveys were isolated, purified and, after liofilization, 1417 strains of rhizobia were added to CNPAB's culture collection. At present, the strains bank at CNPAB/EMBRAPA contains over 1,800 strains.

2. Selection to identify more efficient nitrogen fixing bacteria for each potential species.

For species with more economical and ecological potential, greenhouse experiments have been conducted by CNPAB/EMBRAPA using "Leonard" jars to select more efficient strains of rhizobia. Experiments with more promising strains were carried out in soil conditions. Efficient strains were selected for more than 45 leguminous tree species including *Acacia mangium, A. auriculiformis, A. holosericea, A. angustissima, Albizia guachapele, A. saman, Paraserianthes falcataria, Gliricidia sepium, Leucaena leucochepala, Mimosa caesalpiniifolia, M. scabrella, Enterolobium contortisiliquum, Stryphnodendrum guianensis* and *Prosopis juliflora.*

Many of these species have shown good growth in adverse substrates such as acid and high density soils, which are common in lands degraded by mining.

3. Greenhouse experiments to evaluate nutritional requirements and capacity to grow in high density soils.

Promising species were evaluated for mineral nutrition requirements and capacity for growth in high density soils. These experiments provided important information for choosing species with the best potential to grow in each kind of substrate, and indicated the minimum necessary requirements for good establishment. The main results, obtained from the experiments carried out with *Acacia mangium, A. holosericea, Sclerolobium paniculatum* and *Mimosa tenuiflora* in the Soils Department of the Federal University of Viçosa, follow.

3.1 Response to liming and phosphorus fertilization.

Absence of topsoil is a common condition for surface-mined soil. In the tropics, the lack of topsoil exposes soil horizons with high acidity and phosphorus adsorption capacity. Different studies evaluated the plants response to liming and phosphorus fertilization in the B horizon of

acidic soil (pH = 4.6; P and K by Mehlich-1 = 0.4 and 5.0 mg/dm^3, respectively; Ca^{2+}, Mg^{2+} and Al^{3+} by KCl 0,01 Mol/L = 0.04, 0.02 and 0.9 cmol$_c$/dm^3, respectively).

Only *M. tenuiflora* responded to lime application (Paredes F. et al. 1995). This result confirms field observations that this species is capable of growing in soils with approximately neutral pH. For the other species, liming did not significantly affect dry matter production, height or stem diameter (Dias et al., 1990; Dias et al. 1991a and Balieiro et al. 1995). The lack of response to lime application in soil with low exchangeable calcium shows the low demand for this nutrient and the capacity of the other species to grow in poor substrates. In other greenhouse experiments with a nutrient solution, the low Ca requirement by *A. mangium* was also shown by high dry matter production over the long term despite deprivation of Ca after an initial short contact period with calcium in the original planting solution (Dias et al. 1994a).

All species responded to phosphorus fertilization. The obtained regression equations for dry matter production as a function of the different doses of applied P are shown in Table 1. These data agree with the thesis that leguminous species when fixing nitrogen by symbiotic association have high P uptake to supply the high demand of ATP by the fixation process (Siqueira & Franco, 1988).

The critical levels of P for the different species are found in Table 1. These values ranged from 18 to 32 mg/dm^3 of available Mehlich-1 P and confirm the high demand for this nutrient by fast-growing tree seedlings.

3.2 Response to sulfur and potassium fertilization

Only *M. tenuiflora* (Paredes F. et al. 1995) responded to sulfur fertilization (sulfur rates varyed from 0 to 120 mg/dm^3) which characterized this species as more demanding of soil fertility. The lack of response for other species may be related to the level of organic matter (2.8 %) of the subsoil used and the action of the liming on providing organic-S mineralization (Dias et al. 1991b and Dias et al. 1992).

Although the S requirement is low, in solution deprivation tests S was the second most important limiting factor, after N, in dry matter production for *A. mangium*, (Dias et al. 1994a).

Given the responses of *Acacia holosericea* (Balieiro et al. 1995) and *Mimosa tenuiflora* (Paredes F. et al. 1995) to potassium fertilization, it was not possible obtain good models by regression analysis. Table 2 shows the obtained equations, which indicate the negative response of *A. mangium* and positive response of *S. paniculatum.*

3.3 Capacity to grow in high density soils

Intense movment of machines over mined susbtrate can induce soil compaction which causes nutritional and water availability problems. Studies on the capacity of the plants to grow in high density soils therefore have great importance.

Fernandez et al. (1994) studied the capacity of seven leguminous woody plants to grow in columns containing compacted subsurface soil samples. The study used 0.91 and 1.14 g/cm^3 of bulk density to fill PVC columns.

Of the seven species evaluated, *Mimosa tenuiflora* and *Leucaena leucocephala* were the most sensitive to high substrate density. Observed effects were reduction in height and dry matter production at 140 days. The other five species (*Enterolobium contortisiliquum, Acacia holocericea, Mimosa caesalpinifolia, Acacia mangium and Acacia auriculiformis*) showed good tolerance to compaction. An interesting effect of compaction was the reduction of root dry matter for *A. mangium* (Figure 1). This species grew better than the others in the different mine substrates, including compacted ones. The exceptional ability of this species to produce high amounts of biomass even in harsh ambiental conditions is shown in Figure 2. *A. mangium* in compacted substrate produced the same amounts of foliar and stem dry matter as in non-compacted tests. The same behavior was observed in field conditions when *A. mangium* was grown in compacted bauxite residue ponds and developed shallow but extensive lateral root systems (Franco et al. 1995b).

4. Field experiments in lands degraded by mining.

Dias et al. (1994a) studied the recuperative capacity of nine-year old *Eucalyptus pellita* versus *Acacia mangium* trees planted on soils degraded by bauxite mining in Porto Trombetas, Pará State, Brazil. Each species was planted in plots wich conteined 36 plants each (3 x 2 m). Litter (layers L and H) and soils (depths 0 to 2.5 cm, 2.5 to 7.5 cm and 7.5 to 20 cm) were sampled at the onset of the rainy season (November 1992), midway through the dry season (June 1993) and at the end of the rainy season (April 1994).

Greatest accumulation of organic material occurred on *A. mangium* plots at 7.5 cm depth and for all three sampling times. Similar behavior occurred for soil sum of bases and effective cation exchange capacity (CEC). Plots with *A. mangium* had more litter with smaller C/N ratio and greater quantites of P, K, Mg and N. These results reinforce the importance of including leguminous trees in land rehabilitation programs.

We also conducted two other experiments at Porto Trombetas, Pará State, Brazil (Franco et al. 1994, Franco et al. 1995a and Franco et al. 1995b). For both we used concentrated sulphuric acid to scarify seeds as required, and nodulating species were inoculated with effective rhizobial strains. All species were also inoculated both with inoculum of *Gigaspora margarita* and with soil plus roots of *Brachiaria decumbes* collected in the region to provide inoculum of introduced and local vesicular-arbuscular mycorrhiza fungi.

The first test was conducted in subsoil, without topsoil, exposed by bauxite mining (pH = 4.6; P and K by Mehlich-1 = 0.1 and 9.0 mg/dm^3, respectively; Ca^{2+}+Mg^{2+} and Al^{3+} by KCl 0,01 Mol/L = 2.0 and 0.3 cmol$_c$/dm^3, respectively). We tested five nodulating (nod+) leguminous species (*Sclerolobium paniculatum, Acacia mangium, A. auriculiformis, Enterolobium contortisiliquum, Strypnodendrum guianensis*); two non-nodulating (nod-) leguminous species (*Cassia leiandra and Senna siamea*) and three pioneer but non-legume (no-leg) species (*Didimopanax morototoni*, *Byrsonima crassicarpa* and *Goupia glaba*). One year

after planting, *A. mangium* and *A. auriculiformis* had grown more than any other species. Among the nod- and no-leg species *Didimopanax morototoni* and *Byrsonima crassicarpa* showed reasonable but less growth than the nodulating legume (Table 3).

The second experiment was planted on bauxite mining residue (pH = 4.6; P and K by Mehlich-1 = 0.0 and 3.0 mg/dm^3, respectively; Ca^{2+} + Mg^{2+} and Al^{3+} by KCl 0,01 Mol/L = 0.4 and 0.2 cmol$_c$/dm^3, respectively). We tested nine nod+ species (1. *A. mangium,* 2. *A. angustissima,* 3. *A. holocericea,* 4. *Albizia saman,* 5. *A. guachapelle,* 6. *Enterolobium contortisiliquum,* 7. *Clitoria fairchiladiana,* 8. *Sclerolobium paniculatum and* 9. *Stryphnodendrum guianensis*); five nod- *species (*10. *Senna siamea,* 11. *Parkia pendula,* 12. *Dinizia excelsa,* 13. *Cassia leiandra* and 14. *Adenanthera pavonina*) and two no-leg species (15. *Cecropia sp,* and 16. *Eucalyptus terenticornis*). At planting time two of the four replications received two liters of compost (cattle manure waste) in the planting hole each and two none.

Plant heights measured at 9, 13 and 22 months after transplanting showed similar trends except that *S. paniculatum* and *S. guianensis* were growing best despite a slow start, while *A. angustissima* showed the opposite behaviour. For the nod+ species, the differences observed with compost added at planting decreased up to the last evaluation. *Acacia mangium* plants showed values above 7 m of height at last measurement. Figure 3 shows total biomass recovered in the plants 22 months after transplanting. *Acacia mangium and A. holocericea* showed highest value for shoot dry matter. These results comfirm the great potential of leguminous trees in reclamation of tropical mined lands.

Conclusions

Many leguminous trees fix atmospheric nitrogen when they occur in symbiosis with rhizobia and form mycorrhizal associations which improve efficiency of absorption of water and mineral nutrients from the soil. Such improved efficiency especially affects phosphorus, the most important and frequently limiting nutrient in tropical soils. These species, when associated with microsymbionts, are suitable for revegetation of mined tropical lands when the deficiencies of major nutrients (other than nitrogen, unnecessary because of fixation capability) have been corrected. The addition of a large amount of organic matter (litter) with low C/N ratio besides protects the soil from the direct impact of raindrops and erosion, accelerates the cycling of soil nutrients, and favors the return of life to the soil.

The slow growth of native species means, however, that we must identify native nodulating species that can provide better ecological conditions for soil rehabilitation. Griffith et al. (1994) emphasized the importance of using an ecological approach in mined land reclamation, and indicated that one way to do so is to use nodulated and mycorrhizal legume trees.

The current focus of our research program is to understand the natural evolution of areas undergoing rehabilitation so as to identify opportune moments to intervene in the revegetation process and thereby guarantee its sustainability.

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Table 1. Regression equations adjusted for dry matter production for selected species, in function of levels of P applied to substrate and critical level values in soil, so as to obtain 90% of maximum production

Species	Dosage	Model	R^2	Crit. Level
	mg/dm^3	g/pot		mg/dm^3
A. mangium	0 - 570	$Y = 7.3714 + 0.0139\ P$	0.850	1/
A. holosericea	0 - 480	$Y = 0.466 + 0.036\ P - 0.00006\ P^2$	0.968	18.55
M. tenuiflora	0 - 480	$Y = 0.1952 + 0.0119\ P - 0.00002\ P^2$	0.947	31.95
S. paniculatum	0 -570	$Y = 3.717 + 0.0513\ P - 0.00006\ P^2$	0.944	26.10

1/ The linear response to P applied did not allow obtaining the critical level

Table 2. Regression equations adjusted for dry matter production for selected species, in function of levels of K applied to substrate and critical level values in soil, so as to obtain 90% of maximum production

Species	Dosage	Model	R^2	Crit. Level
	mg/dm^3	g/pot		mg/dm^3
A. mangium	0 - 200	$Y = 15.041 - 0.561\ X^{0.5} + 0.026\ X$	0.985	1/
S. paniculatum	0 -300	$Y = 9.920 + 0.046\ X - 0.0001\ X^2$	0.909	27.4
M. tenuiflora	0 - 160	$Y = 1.563 + 0.053\ X^{0.5} - 0.004\ X$	0.886	16.61

1/ The negative response to applied K did not allow obtaining the critical level.

Table 3. Height at 13 months for 10 species cultivated in subsoil (topsoil absent) exposed by bauxite mining

Species	height
	--- cm ---
Sclerolobium paniculatum	56.9
Acacia mangium	258.8
Acacia auriculiformis	163.9
Enterolobium contortisiliquum[1]	76.1
Strypnodendrum guianensis	83.4
Cassia leiandra[1]	78.9
Senna siamea	97.4
Didimopanax morototoni	14.8
Byrsonima crassicarpa	75.2
Goupia glaba	33.1

1/ Plants browsed by deer

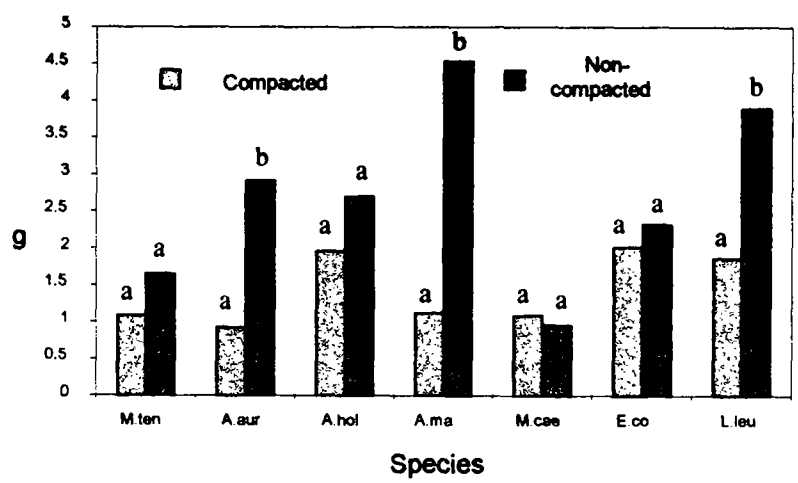

Figure 1. Root dry matter for seven forest leguminous plants grown in compacted and non-compacted soil. M.ten = *Mimosa tenuiflora,* A. aur = *Acacia auriculiformis,* A. hol = *Acacia holocericea,* A. man = *Acacia mangium,* M. cae = *Mimosa caesalpinifolia,* E. co = *Enterolobium contotisiliquum,* and L. leu = *Leucaena leucocephala.* The letters indicate differences between compacted and non-compacted soil within a species.

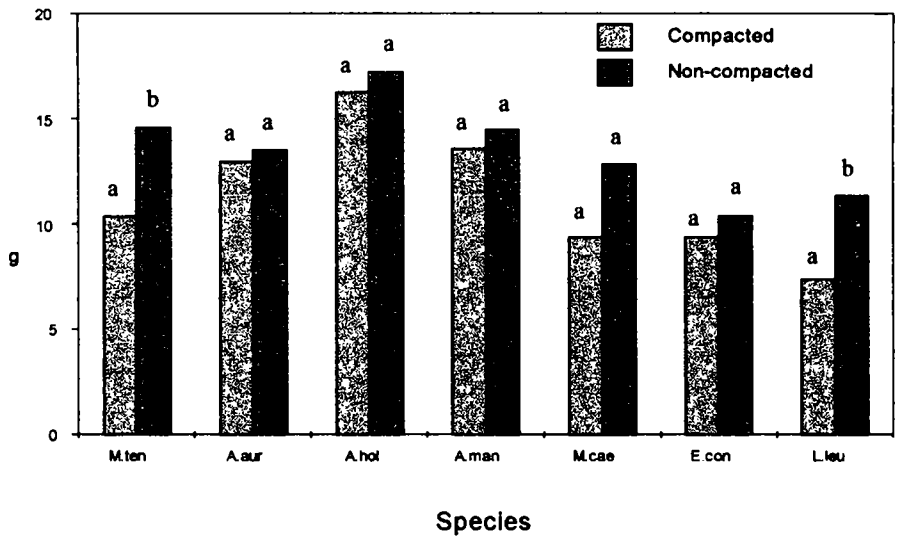

Figure 2. Total dry matter of seven forest leguminous species grown in compacted and non-compacted soil. M.ten = *Mimosa tenuiflora,* A. aur = *Acacia auriculiformis,* A. hol = *Acacia holocericea,* A. man = *Acacia mangium,* M. cae = *Mimosa caesalpinifolia,* E. co = *Enterolobium contotisiliquum,* and L. leu = *Leucaena leucocephala.* The letters indicate differences between compacted and non-compacted soil within a species.

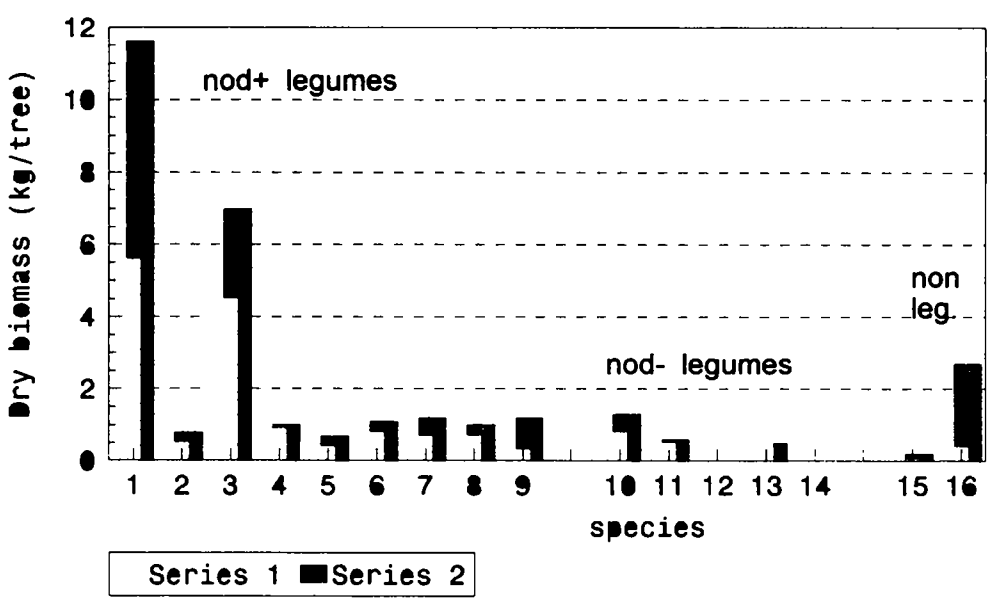

Figure 3. Biomass of 16 forest leguminous species production,22 months after transplanting to the field. Series 1 without compost and series 2 with compost.

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ORGANIC AMENDMENT EFFECTS ON NITROGEN AND CARBON MINERALIZATION IN AN APPALACHIAN MINESOIL[1,2]

R.D. Faulconer, J.A. Burger, S.H. Schoenholtz, and R.E. Kreh [3]

Abstract: The use of blasted rock overburden as a topsoil substitute during surface-mined land reclamation is practiced in areas with thin, unrecoverable topsoil. The long-term productivity of topsoil substitutes has often been difficult to maintain under forage and row crops. The objective of this project was to evaluate the effectiveness of an unamended topsoil substitute as a tree growth medium compared to both topsoil and organic matter amended minesoils based on the accumulation and mineralization of C and N pools. A factorial experiment was established in 1987; treatments (5 cm of a Jefferson series topsoil, 8 cm of whole-tree woodchips, and an unamended control) were assigned to lysimeters filled with blasted overburden. All lysimeters were planted with a tree-compatible ground cover of grasses and legumes and 10 pitch pine *x* loblolly pine hybrid seedlings (*Pinus rigida* L. *x P. taeda* L.). Net accumulated Total Organic C in 1989 was 4.4, 3.7, and 9.2 g kg^{-1} for the control, topsoil, and woodchip treatments, respectively; in 1995, concentrations were 12.7, 16.0, and 18.2 g kg^{-1} . Aerobic N mineralization potential in 1988 was 31, 63, and 56 mg kg^{-1} for the control, topsoil, and woodchip treatments and increased to 112, 157, and 118 mg kg^{-1} by 1995. These and other results show that within 8 years, N and C accumulation and cycling in the unamended control have reached levels comparable to the amended minesoils. This suggests the potential for reclamation cost savings through the use of unamended topsoil substitutes without compromising long-term forest productivity.

Additional Key Words: Topsoil substitute; Aerobic nitrogen mineralization; Reclamation forestry.

Introduction

Coal operators are required by law (SMCRA: P.L. 95-87) to reclaim surface-mined lands to a level of productivity equal to or greater than the productivity of the land before mining occurred. The coal operator is responsible for meeting all legal reclamation requirements and must post performance bonds that are released in stages as requirements are met. When the coal operator leases the mined area from a separate landowner, a conflict can occur between the short-term goals of the coal operator (to perform the most cost effective reclamation that meets legal obligations) and the long-term goals of the landowner (which may be to achieve and maintain productivity necessary for forest management). The post-mining land use chosen as a reclamation goal has often been hay lands or pasture lands because a grass cover crop is relatively easy to establish. These hay or pasture lands often represent a missed economic opportunity for the landowner because the long-term productivity of reclaimed lands is difficult to maintain under nutrient-intensive forage crops. Failing hay and pasture lands in remote, rugged areas often create erosion hazards and liabilities and will eventually revert to native forest cover. However, this forest will probably be slow growing and unproductive unless forestry is specifically planned for before and during reclamation. Re-establishment of trees can improve minesoil properties, provide food and habitat for wildlife, and provide an additional source of income through timber harvests or tree crops. Reclamation

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[2] Publication in this proceedings does not prevent authors from publishing their manuscripts, whole or in part, in other publication outlets.

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forestry can provide long-term benefits for landowners and environmental and aesthetic benefits for local communities. For reforestation to be a viable alternative for landowners and coal operators, reclaimed mine land must be capable of supporting productive forests and reclamation forestry must be cost effective. Consequently, a great deal of research has focused on the forest productivity of minesoils created during the reclamation process (Moss et al. 1989; Torbert et al. 1988, Bengtson and Mays 1978).

Long-term growth and maintenance of forest vegetation in large part depends on accumulating pools of C and N and establishment of nutrient cycles associated with these pools. Freshly reclaimed minesoils contain little or no organic matter and little C or N in plant-available forms (Bradshaw et al. 1982). Fertilizer applied at the time of seeding provides enough N and other nutrients to aid in the establishment of a plant community; however, this nutrient supply is transitory and high nutrient demanding forages often show deficiency symptoms after only a few years (Reeder and Sabey 1987; Woodmansee et al. 1978). Traditionally, the approach to establishing pools of C and N has been to replace native topsoil, when available. Since Appalachian topsoils are frequently thin and difficult to recover, there has been some interest in amending minesoils with organic materials such as sawdust or sewage sludge (Moss et al. 1989; Roberts et al. 1988). In contrast to these methods which involve the placement of a substrate which already includes C and N, another option is to stimulate the establishment of a soil/plant system which builds the C and N pools by natural processes. The gradually increasing resource needs of trees more closely matches the supply of C and N from gradually increasing pools. It is hypothesized that the C and N cycles can develop naturally if coal operators will select an appropriate spoil type, place it in an uncompacted fashion, establish a tree-compatible ground cover of grasses and legumes, and plant or seed selected tree species. Under the right conditions, this last approach saves the cost of topsoil or organic matter amendments and creates a growth medium where C and N accumulate and cycle and trees and ground cover survive and grow.

The objective of this research project was to evaluate the effectiveness of an unamended topsoil substitute as a growth medium compared to both topsoil and organic matter amended minesoils and N fertilized minesoils. This evaluation was based on the accumulation of C and N pools and the mineralization capacity of these pools over the eight years since the minesoils were created.

Materials and Methods

This project was initiated at the Reynolds Homestead Forest Resources Research Center in Critz, Virginia in 1987 and was fully described by Schoenholtz (1990). Large concrete lysimeters (2.6 m^3) were used as replications in a 2 by 3 factorial experiment testing the effects of two levels of N fertilization and three levels of organic amendment on C and N cycling in minesoil.

In July, 1987, eighteen concrete lysimeters were filled with a pre-weighed amount of fresh mine spoil associated with the Middle Wise Formation obtained from Wise County, Virginia. The mine spoil consisted mainly of sandstone and siltstone with traces of shale. An attempt was made to fill each lysimeter with the same percentage of the different size fragments present ranging from sand-sized particles to large rocks. Each lysimeter has a surface area of 2.9 m^2 and a depth of 0.8 m.

Six of the lysimeters received 5 cm of topsoil found in the area where the spoil was removed. The topsoil consisted of the A horizon from a Jefferson series soil (loamy, mixed, mesic Typic Hapludult) also from Wise County, Virginia. Six lysimeters received 8 cm of fresh, yellow-poplar (*Liriodendron tulipifera* L.), whole-tree wood chips; and the remaining six lysimeters received no organic amendments. Within the three treatments, half received 100 kg ha^{-1} N applied as NH_4NO_3 and half got no N fertilization. Results from the first three years of this experiment show that the fertilizer treatment effects on N and C parameters ceased to be important after the first year (Schoenholtz 1992). All lysimeters were initially fertilized with 100 kg ha^{-1} P and 60 kg ha^{-1} K. The wood chips had a C/N ratio of 261:1 and were used to provide a source of organic N and a C source for heterotrophic bacteria. The topsoil had a C/N ratio of 17:1 and was used to evaluate the potential of topsoil as a source of organic N and as a microbial innoculum to promote N cycling. The wood chips and topsoil were tilled into the top 25 cm of the mine spoil prior to seeding and tree planting.

In July 1987, a tree-compatible mixture of the following grass and legume species was sown in each lysimeter: foxtail millet [*Setaria italica* (L.) Beauv.], perennial ryegrass (*Lolium perenne* L.), annual ryegrass (*Lolium multiflorum* Lam.), redtop (*Agrostis gigantea* Roth), birdsfoot trefoil (*Lotus corniculatus* L. var. *corniculatus*), Korean lespedeza (*Lespedeza stipulacea* Maxim.), and 'Appalow' sericea lespedeza [*Lespedeza cuneata* (Dum.-Cours.) G. Don]. Ten 1-0 bareroot pitch *x* loblolly hybrid pine (*Pinus rigida* L. *x* *P. taeda* L.) seedlings were planted in each lysimeter in March 1988. The legumes accounted for over 80 percent of the ground cover after three years, and tree survival was 60 percent in the topsoil treatment, 83 percent in the control, and 98 percent in the woodchip treatment (Schoenholtz 1992).

Soil Analyses

A composite soil sample was collected from the 0 to 10 cm layer of each tank using a 2 cm push tube and used for the following analyses.

- Total organic carbon (TOC) was determined by dry combustion (Nelson and Sommers 1982).
- Total Kjeldahl nitrogen (TKN) was determined by a modified micro-Kjeldahl digestion procedure (Bremner and Mulvaney 1982).
- N mineralization potential (N_o) was determined from composite samples using the aerobic incubation procedure of Stanford and Smith (1972) as modified by Burger and Pritchett (1984).
- An in-situ buried-bag procedure was used to determine the monthly rate of net N mineralization. Net mineralized N was calculated as the difference in 2M KCl-extractable N between unincubated and incubated samples.
- In-situ CO_2 was monitored over a 24 hour period using the alkali trapping method described by Anderson (1982).

Foliar N concentration was determined by Kjeldahl digestion of dried and ground samples (Bremner and Mulvaney 1982). All soil and foliar N concentrations were determined colorimetrically using a Technicon Autoanalyzer II.

Since there were no significant differences between N fertilizer treatments on vegetation and soil variables by 1989, the treatment was disregarded and data were analyzed using a one-factor analysis of variance with three levels of amendment (control, topsoil, and wood chips) and six replications. Duncan's Multiple Range Test (α level of 0.05) was used to test for differences among treatment means. Non-linear, least-squares regression (NLLSR) was used to estimate N_o from aerobic incubation data. NLLSR was also used to determine the accumulation rates (k) of TOC and TKN.

Results and Discussion

C and N Accumulation

The values for TOC and TKN that follow are amounts that have accumulated since the start of the project. All three treatments had accumulated between 1.2 to 1.8 percent TOC or roughly 2 to 3 percent organic matter after eight years (Figure 1). The woodchip treatment had the highest TOC over 8 years, though not significantly different from the topsoil treatment in the eighth year. The control treatment TOC was not significantly different from the topsoil treatment except in the eighth year (Figure 1). The TOC accumulation rate for the woodchip treatment (0.033 $month^{-1}$) was significantly higher than both the topsoil (0.008 $month^{-1}$) and control (0.013 $month^{-1}$) treatments which were not significantly different from each other.

TKN was highest in the topsoil treatment over 8 years (Figure 2). The TKN values for the woodchip and control treatments were not significantly different from each other after 8 years. The TKN accumulation rates were not significantly different among the three treatments, but the rate of TKN accumulation in the woodchip and control treatment appears to have slowed by year 8. This may be attributable to a difference in the number of trees per tank which, in turn, affected shading and ground cover. Seedling survivability was an initial part of this research, and, as a result, seedlings that died were not replaced. After 8 years, there were an average of 9.7 trees per tank in the woodchip treatment, 7.3 trees per tank in the control, and 5.2 trees per tank in the topsoil treatment. The higher number of trees in the woodchip and control treatments caused total ground cover, including the legume component, to decrease after the third growing season. Therefore, the decrease in the rate of

TKN accumulation that occurred after the third growing season in the woodchip and control treatments may be largely the result of tree-ground cover interactions.

The TOC and TKN concentrations were converted to a weight-per-unit-area basis using the fine-soil bulk density and the fine-earth fraction of the 0 to 10 cm layer (Table 1). After 8 years, TOC in the control and woodchip treatments was not significantly

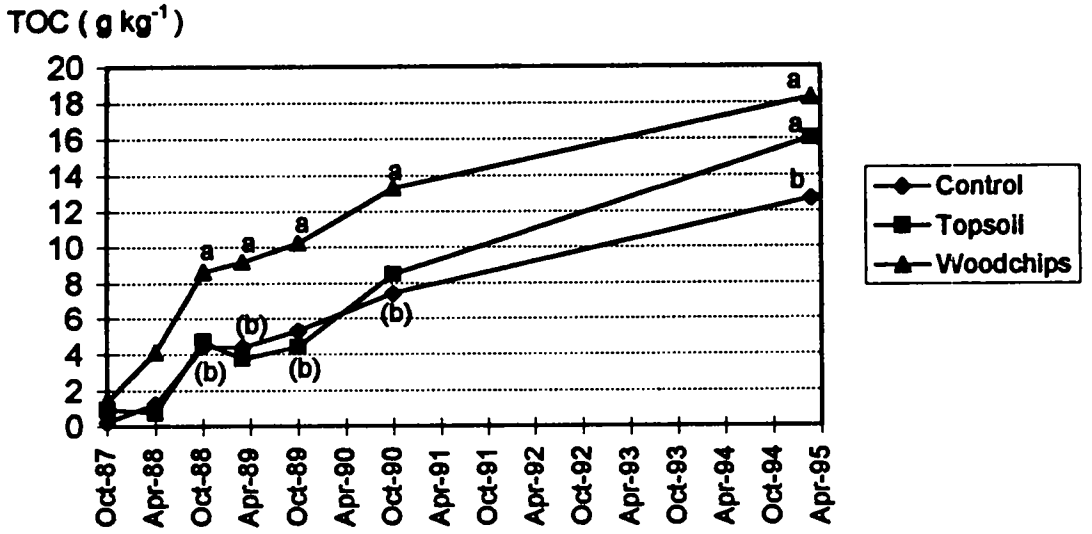

Figure 1. Effect of organic amendment on TOC accumulation in an Appalachian minesoil. For each date, different letters represent statistical differences at the $\alpha = 0.05$ level.

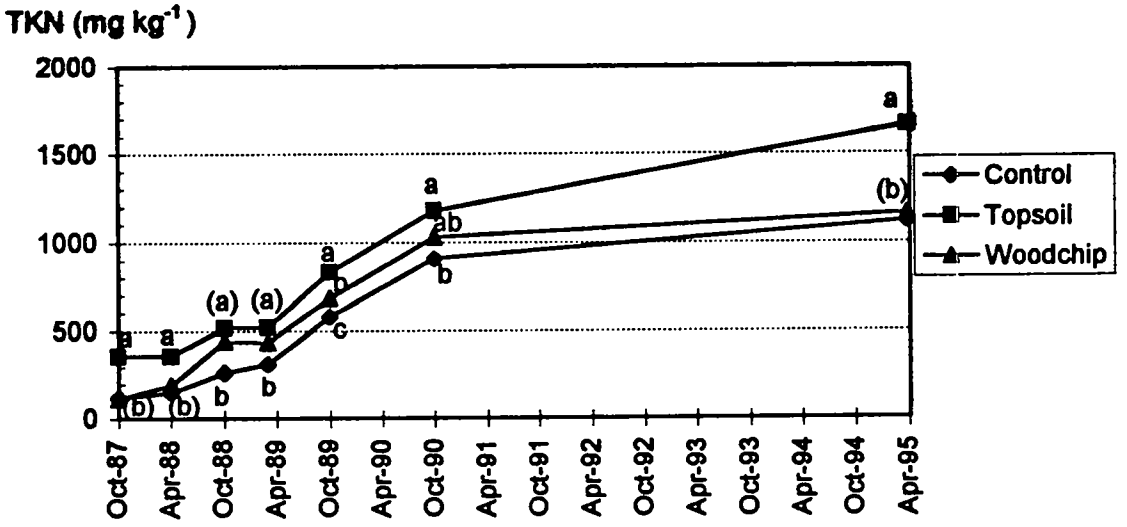

Figure 2. Effect of organic amendment on Total Kjeldahl N (TKN) in an Appalachian minesoil. For each date, different letters represent statistical differences at the $\alpha = 0.05$ level.

different, and TKN in the control and topsoil treatments was not significantly different (Table 1). Wells and Jorgensen (1975) reported that a 16-year-old loblolly pine stand in North Carolina on undisturbed soils had 1,753 kg N ha^{-1} in the mineral soil, 74 percent of the total N in the system. These 8-year-old minesoils have accumulated roughly half of the N found in a natural soil several thousands of years old.

C Mineralization

Organic C quality is an important determinant of C and N cycling. A C source that is highly resistant to decomposition will provide very little available N due to low C mineralization rates. Quality of organic matter can be evaluated by studying the activity of heterotrophic microbes that are responsible for decomposing and mineralizing organic matter, and microbial activity is reflected by the amount of CO_2 evolved from the system. During the first two years of the study, the woodchip treatment showed significantly higher in-situ CO_2 evolution than both the topsoil and control treatments (Table 2). However, by the seventh year, differences in CO_2 evolution rates were much smaller among the three treatments. Over 7 years, the control and topsoil treatments had accumulated more readily mineralizable forms of C, while the CO_2 evolution rates in the woodchip treatment remained essentially unchanged (Table 2).

Nitrogen Mineralization

N is the nutrient that plants require in the largest amount and that is often the most limiting in young minesoils (Woodmansee et al. 1978). The accumulation and mineralization of C in a minesoil are important factors in determining N mineralization, but it may not be the determining factor. While the woodchip treatment had the highest TOC concentration and the highest C mineralization values at year 8, the topsoil treatment had the highest N_o at year 8 (Figure 3). In a survey of 11 minesoils and two native soils, Stroo and Jencks (1982) found that microbial respiration was not significantly correlated with Total N or Mineralizable N.

The N_o for the woodchip treatment was not significantly different than the topsoil treatment in the first and second years; but, as the readily mineralizable forms of C were decomposed, the N_o of the woodchip treatment dropped to the same level as the control by the eighth year (Figure 3). The N_o was not significantly different among the three treatments in the third year. The N_o for the topsoil treatment was significantly higher than the N_o for the control throughout the 8 years of the study except for the third year. The decline of N_o in the woodchip and control treatments follows the decline in the rate of TKN accumulation in both treatments (Figure 2), and again may be linked to the drop in legume ground cover caused by the higher number of trees in the woodchip and control treatments.

Table 1. Effect of organic amendment on TOC and TKN (kg ha^{-1}) in an eight-year-old Appalachian minesoil.[1]

Treatment	Total Organic C	Total Kjeldahl N
	kg ha^{-1}	
Control	8893b	784ab
Topsoil	10923a	1132a
Woodchips	10698ab	679b

[1] Fine-soil bulk densities were 1.46, 1.42, and 1.33 for the control, topsoil, and woodchip treatments. Fine-earth fraction was 48, 48, and 44 percent for same.

Table 2. Effects of organic amendment on in-situ CO_2 evolution in an Appalachian minesoil.

Treatment	Evolved CO_2
	-- mg CO_2 m^{-2} $hour^{-1}$ --
October 1987	
Control	108b
Topsoil	125b
Woodchip	220a
October 1988	
Control	102b
Topsoil	110b
Woodchip	202a
October 1994	
Control	196b
Topsoil	211ab
Woodchip	221a

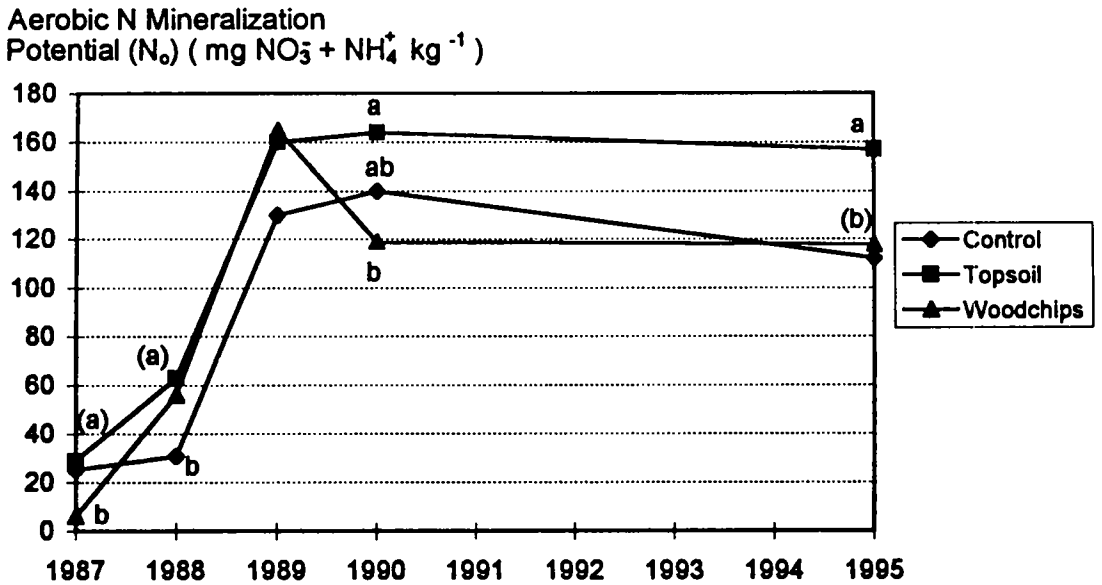

Figure 3. Effect of organic amendments on the N mineralization potential (N_o) of an Appalachian minesoil. A letter in () applies to the two closest data points. For each date, different letters represent statistical differences at the $\alpha = 0.05$ level.

The buried-bag data were collected monthly for the first three years of the experiment (Figure 4). This data showed that the lowest net N mineralization (or the highest net N immobilization) of the year occurred during August and September in each of the first three years. By the eighth year, all three treatments had net N mineralization during August and September (Figure 4). Net N mineralization in the control treatment was not significantly different from that of the woodchip treatment in August and was not significantly different from that of the topsoil treatment in September (Figure 4). Over eight years, the C/N ratio declined from about 50 in the control and woodchip treatments to about 23 and from about 39 to 19 in the topsoil treatment (Figure 4).

If N limitation is one of the primary problems in minesoils, then research must determine how much N is required to ensure short and long-term forest productivity and which treatments in this study met the requirements. A loblolly pine stand, at its most active growth stage, needs about 100 kg N ha^{-1} yr^{-1}. Most of this annual requirement is met by internal recycling, throughfall, and litterfall; atmospheric deposition and leaching losses roughly equal each other (Bormann et al. 1977; Keeney 1980). There is a demand, unmet by these sources, of from 5 to 25 kg N ha^{-1} yr^{-1} that is supplied mostly by the mineral soil (Keeney 1980). The best estimate of this N-supplying ability comes from the buried bag data which represents the closest approximation to field conditions.

Using the 1994 two month average of 7 mg N kg^{-1} $month^{-1}$ for the control treatment (Figure 4) and converting to a yearly rate, the control treatment had an in-situ N mineralization rate of at least 59 kg N ha^{-1} yr^{-1}. This is an indication that the control treatment can meet the 5 to 25 kg N ha^{-1} yr^{-1} required. These results also indicate that all three treatments had the ability to supply inorganic N in the amounts necessary to support trees, given that the woodchip and topsoil treatments had higher average N mineralization rates than the control (Figure 4). The foliar N concentration of the trees in all three treatments was higher than the 1.1 percent critical level reported by Allen (1987). Foliar N concentrations were 1.46, 1.40, and 1.35 percent for the control, topsoil, and woodchip treatments respectively and none were significantly

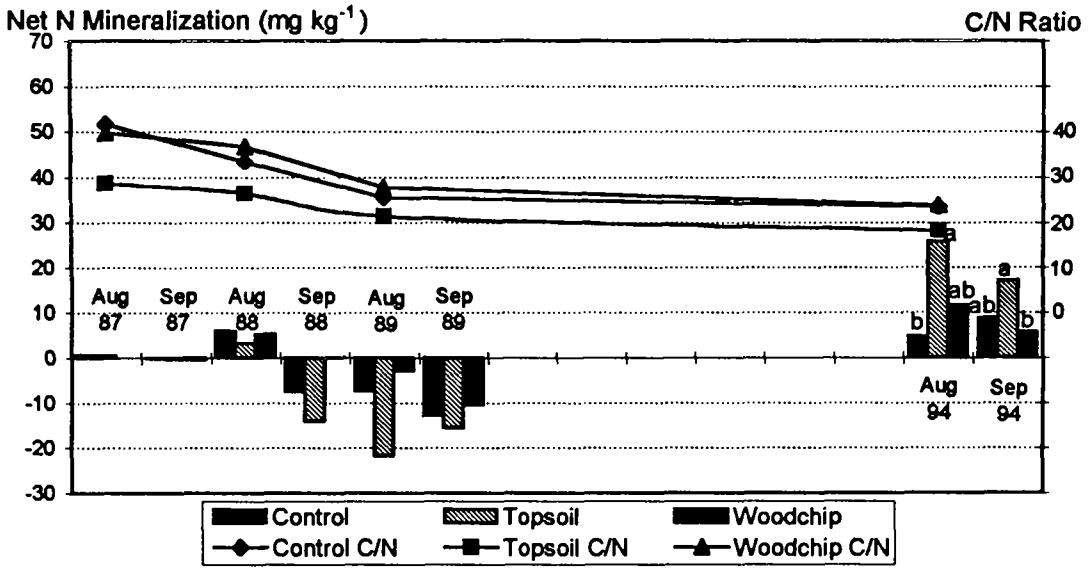

Figure 4. Effect of organic amendments on net N mineralization and C/N ratio in an Appalachian minesoil. Statistical means separation was not performed for mineralization data from the first three years or for the C/N ratio data. For each date, different letters represent statistical differences at the $\alpha = 0.05$ level.

different. The foliar N concentration data provide further evidence that the use of tree-compatible legumes has established sufficient N to support tree growth.

These young minesoils may not have the buffering ability with regard to total N supply that would allow them to withstand significant N removals at first rotation. However, the replacement of the N removed by sustainable harvesting methods, either through fertilization or reseeding with a tree-compatible legume cover crop at the time of replanting, should return these minesoils to a productive level until they have accumulated enough N to buffer natural or managed disturbances.

Conclusions

This research project showed that uncompacted, unamended topsoil substitutes of favorable material, planted with tree-compatible grasses and legumes, can provide the necessary conditions for trees and ground cover to survive and grow, and C and N to accumulate and cycle. Topsoil substitutes so treated can create the conditions necessary to support and sustain forests for the long-term, meet the reclamation requirements for bond release, and save coal operators grading costs and costs associated with topsoil or organic matter amendments. When minesoils are created following the reclamation forestry design, as described in this research, the problem of N limitation can be removed.

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TREE SHELTER AND INTERPLANTED N-FIXING SHRUB EFFECTS ON CROP TREE GROWTH ON A CALCAREOUS MINESOIL[1]

by

David A. Kost, Merlyn M. Larson, and John P. Vimmerstedt[2]

Abstract. In southeastern Ohio, we installed three studies to test the effects of plastic tube tree shelters and interplanted N-fixing shrubs on crop tree survival and growth. Study 1 on graded gray cast overburden used white ash (*Fraxinus americana* L.) and a hybrid willow (*Salix matsudana* x *alba*, or Austree) as crop trees. These were tested in factorial combination with 1.2 m tall tree shelters (present or absent) and N-fixing shrubs [none, bristly locust (*Robinia fertilis* Ashe), or Siberian peashrub (*Caragana arborescens* Lam.)]. Study 2 evaluated growth of bur oak (*Quercus macrocarpa* Michx.) as affected by soil surface (graded cast overburden or ripped topsoil), presence of tree shelters, and presence of interplanted Siberian peashrub. Study 3 tested Austree growth as affected by soil surface and presence of tree shelters. After four or five growing seasons, overall crop tree survivals were white ash (80%), bur oak (52%), and Austree (10% in study 1 and 25% in study 3). All crop tree species tended to survive better with tree shelters but only Austree in study 3 (36% with shelter vs. 14% without shelter) showed a significant difference. Tree shelters increased heights of white ash (102 cm with shelter vs. 21 cm without shelter) and bur oak (84 cm vs. 27 cm). Crop tree survival and growth were not affected significantly by interplanted N-fixing shrubs or by soil surface. In study 1 (cast overburden only), peashrub (64%) survived better than bristly locust (31%) but locust was taller (74 cm vs. 45 cm). In study 2, peashrub survived better on ripped topsoil (53%) than cast overburden (17%), but height was not impressive on either soil (41 ± 11 cm).

Additional Key Words: *Fraxinus americana* L., *Quercus macrocarpa* Michx., *Salix matsudana* x *alba*, *Robinia fertilis* Ashe, *Caragana arborescens* Lam.

Introduction

A variety of cultural techniques, such as fertilizing, herbiciding, soil ripping, spraying animal repellents, or planting containerized stock, can be used to improve tree survival and growth on surface mines. One of the newer practices is the use of tree shelters. Tree shelters are translucent plastic tubes available in various lengths up to 1.8 m that are installed over individual trees and supported by a stake. The shelter modifies the microenvironment around the tree seedling (increases humidity, decreases wind and light intensity) and promotes rapid height growth. The shelter also protects the seedling from vertebrate animal damage until it emerges from the tube. After the terminal bud emerges from the tube, additional growth may carry it beyond the reach of deer, although Ward and Stephens (1995) reported browse damage on trees that had emerged from 1.2 m tubes. Tree shelters show great promise for improving tree establishment but are expensive in terms of initial cost ($3.75 per 1.2 m tall shelter) and labor for installation and maintenance.

Tree growth on new minesoils, which are almost always deficient in nitrogen, may be improved by interplanting nitrogen-fixing shrubs. Although the primary expected benefit from the shrubs is an increase in available soil nitrogen, other possible benefits are the increased accumulation of organic matter and the reduction of wind.

There are a variety of N-fixing shrub species that can be interplanted with trees (Vogel,1981). Bristly locust is a native species that is well adapted to acid minesoils but also grows well on calcareous soils. It spreads by root suckers to form dense stands but is not considered a potential "pest" species

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(Vogel,1981). Root suckering is reduced where ground cover is dense. Siberian peashrub is native to Siberia and Manchuria but has been planted widely for windbreaks on farms and for hedges and outdoor screening in towns of the upper midwest United States (Dietz and Slabaugh,1974). It is adapted to sandy, alkaline soil and open, unshaded sites. Carpenter and Hensley (1979) evaluated seven species of N-fixing shrubs, including bristly locust and Siberian peashrub, for adaptation to an extremely acid minesoil amended with lime. These two species have also been planted on alkaline iron and copper tailings in Michigan (Shetron and Carroll, 1977).

These studies used bur oak, white ash, and Austree as crop tree species. Bur oak and both white and green ash have shown good to excellent survival in other studies on the Muskingum Mine. White ash has a high N requirement for best growth (Mitchell and Chandler, 1939) and so may show a strong response to the interplanted N-fixing shrubs. Austree is a hybrid between Salix matsudana Koidz (native to Korea) and Salix alba L. (white willow; native to Europe and north Africa). It has potential for rapid growth and use in windbreaks and its leaves are an excellent source of animal fodder.

These studies evaluated effects of tree shelters and interplanted N-fixing shrubs on survival and growth of white ash, bur oak, and Austree. Two of the studies also compared tree and shrub growth on ripped topsoil versus graded cast overburden.

Methods

The studies are on Central Ohio Coal Company's Muskingum Mine, 5 km south of Cumberland, Ohio. Study 1 is west of State Route 83 in Muskingum County, and studies 2 and 3 are east of State Route 83 in Noble County. The area was mined using pan scrapers to remove and stockpile the topsoil, followed by draglines to uncover the Meigs Creek No. 9 coal seam. The overburden is predominately gray shale and limestone, with lesser amounts of sandstone and claystone.

Reconstructed soil surfaces consisted of graded gray cast overburden without topsoil, and graded cast overburden with 30 cm of replaced topsoil. The topsoil was ripped to 30-cm depth at 30-cm spacing after grading and then disked (ripped topsoil). Ripped topsoil is classified as Morristown silty clay loam, a loamy-skeletal, mixed (calcareous), mesic Typic Udorthent (Waters and Roth,1990). Soil physical and chemical characteristics are listed in Table 1. Cast overburden has greater sand content but lower levels of extractable nutrients in study 1 relative to studies 2 and 3.

All study areas received 448 kg/ha of 8-32-16 (N-P-K) fertilizer, plus ammonium nitrate at 224 kg/ha for study 1 and at 112 kg/ha for studies 2 and 3. The areas were seeded with an herbaceous mix containing the following species and seeding rates (kg/ha): orchardgrass-Dactylis glomerata L. (6.7), timothy-Phleum pratense L. (11.2), perennial ryegrass-Lolium perenne L. (9.0), Kentucky bluegrass-Poa pratensis L. (5.6), Mammoth red clover-Trifolium pratense L. (5.6), and Empire birdsfoot trefoil-Lotus corniculatus L. (5.6). Wheat (Triticum aestivum L.) at 1.3 hL/ha was included as a cover crop for studies 2 and 3. Plots were mulched with hay after seeding. Fertilizing, seeding, and mulching were done in spring 1988 for study 1 and in late summer 1987 for studies 2 and 3.

Tree rows were sprayed with glyphosate herbicide (2.24 kg active ingredient/ha) in 0.8 m-wide strips in April 1991, and trees and shrubs (except Austree) were planted within two weeks. Austree planting was delayed until 1992. All tree rows were sprayed with oryzalin herbicide (4.48 kg active ingredient/ha) in 0.8 m-wide strips in May 1991.

All studies used the same subplot design. Each subplot (5.5 m by 7.3 m) was planted with six crop trees (either white ash, Austree, or bur oak) arranged in two rows at 1.8 m by 1.8 m spacing. Crop trees were also planted on the borders at the ends of all plots. Plots interplanted with bristly locust contained eight locust, so that each crop tree was flanked by two locust planted in the tree row. Plots interplanted with Siberian peashrub contained 24 peashrubs, with each crop tree flanked by two groups of peashrubs planted in the tree row. Each peashrub group contained three plants spaced about 0.3 m apart. Brown plastic tube tree shelters were installed on all six crop trees in those plots that received shelters.

All studies had three

Table 1. Soil physical and chemical characteristics for study areas on the Muskingum Mine.

	Cast Overburden		Ripped Topsoil
	Study 1	Studies 2 and 3	Studies 2 and 3
Sieve Analysis	% by weight		
> 4 mm	41	30	18
2 - 4 mm	7	14	7
< 2 mm	52	56	75
Texture Analysis[1]	%		
Sand	66	36	28
Silt	23	41	40
Clay	11	23	32
pH	7.7	7.0	7.0
Extractable Nutrients[2]	kg/ha		
P	4	13	16
K	211	377	285
Ca	9442	14381	5824
Mg	606	1157	877
	meq/100g		
Cation Exchange Capacity	24	37	17

[1] < 2 mm fraction
[2] P by Bray-1 extraction. K, Ca, and Mg by ammonium acetate (1 mole/L) extraction.

replications. Study 1 used a randomized complete block design to test three interplanted N-fixing shrub treatments (no shrub, bristly locust, or Siberian peashrub), two tree shelter treatments (no shelter or 1.2 m tube shelter), and two crop tree species (white ash or Austree) in a 3 x 2 x 2 factorial experiment. The soil surface is graded cast overburden. Each replication measured 21.9 m by 21.9 m.

Studies 2 and 3 were separate but contiguous experiments using split-plot designs with soil surface (graded cast overburden or ripped topsoil) as main plots. Main plots measured 11.0 m by 14.6 m in Study 2 and 5.5 m by 14.6 m in Study 3. Study 2 (bur oak crop tree) had factorial combinations of two tree shelter treatments (no shelter or 1.2 m tube shelter) with two interplanted shrub treatments (no shrub, or Siberian peashrub) as subplots. Study 3 (Austree crop tree) had two tree shelter treatments (no shelter or 1.2 m tube shelter) as subplots. European alder [Alnus glutinosa (L.) Gaertn.] was planted in all subplots of the Austree study so presence of alder was not an experimental factor in the study. Each subplot contained eight alder, with each Austree flanked by two alder in the tree row.

In 1991, white ash was a mixture of seedlings from three half-sib sources grown at the Marietta, OH state nursery and was relatively small one-year-old stock. Bristly locust and European alder (Vallonia, IN state nursery), Siberian peashrub (Lincoln-Oakes, Bismarck, ND nursery) and bur oak (General Andrews Nursery, Willow River,

MN) were purchased in 1990 and held as transplants in the OARDC nursery until 1991.

Because of considerable mortality due to a drought in 1991, all trees and shrubs that died, except European alder, were replanted in April 1992. Austree was also planted then. White ash and bur oak (Zanesville, OH state nursery), peashrub (Lincoln-Oakes nursery), and bristly locust (Augusta Forestry Center, Crimora, VA) were purchased new in 1992. Austree obtained from Austree, Inc. (Pescadero, CA) was received too late for outplanting in 1991 and so was held in the OARDC nursery until planting in 1992. Thus, Austree stock was stout, rooted cuttings about 30 cm long and 2-3 cm in diameter.

Tree shelters were installed on appropriate crop trees on April 27 and May 1, 1992.

We measured tree and shrub survival in May 1991, June 1992, and in September or October from 1991 through 1995. Crop tree heights (nearest cm) were measured in September 1992. Heights of all tree and shrub species were measured in September or October from 1993 through 1995. Survival and height of bristly locust were based upon the eight planted seedlings per subplot (sprouts not counted).

Data analysis used the ANOVA or GLM procedures of Statistical Analysis System (SAS) software (SAS Institute Inc.,1987). Survival percentages were transformed with the arcsine-square root function before analysis. If analysis of variance indicated significant ($p=0.05$) effects, we tested differences among means at $p=0.05$ using the LSD test.

Results

Crop Tree Survival and Growth

Study 1. Overall white ash survival was 57% the first year, increased to 86% after replanting the second year, and stabilized at 80% after five years. Tree shelters did not significantly affect ash survival but did produce a fivefold increase in height (Table 2). Height of ash without shelters showed little increase compared to the initial planting height of 15 to 20 cm.

Overall Austree survival was 55% the first year but decreased to 10% the second year (1993) presumably because of low rainfall (1.2 cm) from July 22 to September 1. Survival remained stable through the fourth growing season, but only 6 of 18 Austree plots had surviving Austree. First-year heights were significantly greater with shelters (95 cm) versus without shelters (35 cm). After four years, tree shelter effects on Austree survival and height were not statistically significant (Table 2).

Survival and height of ash or Austree were not affected by presence of interplanted shrubs (Table 2), or by significant interaction of shelter treatments with interplanted shrub treatments.

Study 2. Overall bur oak survival was 52% the first year (67% on ripped topsoil, 38% on cast overburden), increased to 75% after replanting the second year, and appeared to stabilize at 52% after five years. Bur oak tended to survive better on ripped topsoil and with tree shelters but the differences were not significant (Table 3). Tree shelters produced a threefold increase in height (Table 3). Heights of oak without shelters showed little increase compared to the initial planting height of 20 to 25 cm. Oak survival and height were not affected by interplanted Siberian peashrub (Table 3), or by significant interactions of the soil surface, shelter, and interplanted peashrub treatments.

Study 3. Overall Austree survival was 72% the first year (86% on ripped topsoil, 58% on cast overburden), decreased to 22% the second year (1993) because of low rainfall in August, and appeared to stabilize at 25% after four years. First-year heights were significantly greater with shelters (107 cm) versus without shelters (31 cm). After four years, Austree survived better with tree shelters and tended to survive better on ripped topsoil than on cast overburden (Table 3). Survival on cast overburden was similar in Study 3 (11%) and Study 1 (10%). Austree tended to be taller with tree shelters (Table 3).

Survival and Height of Interplanted N-fixing Shrubs and Trees - All Studies

In study 1 (graded cast overburden only) Siberian peashrub survived better than bristly locust but locust was taller (Table 4). Peashrub survival in individual plots ranged from 46% to 79%.

Table 2. Tree survival and height on graded cast overburden (Study 1) after four or five growing seasons as affected by tree shelters and interplanted N-fixing shrubs.

	Survival		Total Height	
	White Ash (yr 5)	Austree (yr 4)	White Ash (yr 5)	Austree (yr 4)
	--------- % --------		-------- cm --------	
Tree Shelter[1]				
Yes	87.0[2]	16.7	102 a	275
No	72.2	3.7	21 b	128
Interplanted Shrub				
None	77.8	5.6	65	250
Bristly locust	80.6	19.4	59	227
Siberian peashrub	80.6	5.6	62	212

[1]Tree shelters were installed on white ash at beginning of second growing season.
[2]Within a column and factor, means followed by different letters are significantly different at p = 0.05 using the LSD test.

Bristly locust survival in individual plots varied from 0% to 75%. Locust survival is based on the planted locust only (8 per plot). Several plots had volunteer locust from root sprouts, including one plot and adjoining area with 65 stems.

In study 2, peashrub survived better on ripped topsoil than on cast overburden (Table 4). Peashrub survival appeared lower in study 2 than study 1 (Table 4), with survival on cast overburden in study 2 being less than one-third that in study 1.

European alder in study 3 had excellent initial survival (97%) in June 1991. Survival decreased to 9% in September 1991 following the drought and stabilized at 7% after five growing seasons. The few surviving alder were on ripped topsoil (Table 4). European alder also showed a sharp decrease in survival (from 87% to 14%) during the 1991 drought in a herbicide study that benefited from two years of above average rainfall before the drought (Kost et al,1992).

Discussion

Tree shelters had a greater effect on tree survival and growth than interplanted N-fixing shrubs. Heights of white ash and bur oak and survival of Austree were increased by tree shelters. White ash and bur oak without tree shelters showed little height increase after five years, due in part to damage by rabbits and deer. These studies used relatively small plots (6 crop trees per plot) and tree shelters may have concentrated the animal damage on the unprotected trees.

Lack of height growth on unprotected trees in these studies does not agree with results from all other ongoing studies on the Muskingum mine. In a nitrogen fertilization study (Kost and Vimmerstedt,1994), white ash without shelters had similar survival (97%) and height (94 cm) after five years as trees with shelters in study 1 (Table 2). Thus, fertilized trees without shelters may grow as well as trees with shelters. In a species trial (unpublished data), white ash without shelters had similar survival (79%) but lesser height (34 cm) after four years compared to trees with shelters (74 cm) in study 1. In another study (unpublished data), bur oak without shelters had similar survival (56%) but lesser height (25 cm) after five years as trees with shelters in

Table 3. Tree survival and height after four or five growing seasons as affected by soil surface, tree shelters, and interplanted Siberian peashrub.[1]

	Bur oak (Study 2, yr 5)		Austree (Study 3, yr 4)	
	Survival	Total Height	Survival	Total Height
	---%---	-----cm-----	---%---	-----cm-----
Soil Surface				
Ripped topsoil	63.9[2]	72	38.9	219
Graded cast overburden	40.3	40	11.1	163
Tree Shelter[3]				
Yes	59.7	84 a	36.1 a	214
No	44.4	27 b	13.9 b	74
Interplanted peashrub				
Yes	52.8	53	---	---
No	51.4	60	---	---

[1]Interplanted peashrub was not a factor in Study 3 with Austree.
[2]Within a column and factor, means followed by different letters are significantly different at p = 0.05 using the LSD test.
[3]Tree shelters were installed on bur oak at beginning of second growing season.

Table 4. Survival and height of N-fixing woody plants after five growing seasons.

	Survival	Total Height
	-----%-----	-----cm----
Study 1- Graded cast overburden		
Bristly locust	30.8 b[1]	74 a
Siberian peashrub	64.2 a	45 b
Study 2- Siberian Peashrub		
Ripped topsoil	53.4 a	47
Graded cast overburden	17.2 b	34
Study 3- European alder		
Ripped topsoil	14.6	184
Graded cast overburden	0.0	--

[1]Within a study and column, means followed by different letters are significantly different at p = 0.05 using the LSD test.

study 2 (Table 3). In contrast to the tree shelter experiments, other ongoing studies were planted in 1989 or 1990 and had one or two years of above average rainfall before the drought in 1991. Thus, for species such as white ash and bur oak that survive well without shelters, the main benefit of shelters is increased height growth. Height growth may slow after seedlings emerge from the shelters (Lantagne,1995). By 1995, 33% of bur oak and 36% of white ash had emerged from shelters.

Austree survival was good on ripped topsoil but mediocre on cast overburden during the first growing season with adequate rainfall. The sharp decline in survival that probably resulted from the dry period in the second growing season suggests Austree, like other willows, requires abundant soil moisture. It will not be possible to monitor the long-term effects of the interplanted shrubs on Austree in study 1 because most plots do not have surviving Austree.

There are probably several reasons why the interplanted N-fixing shrubs have not affected crop tree growth. Nitrogen fixation by herbaceous legumes, which were present in all studies, may have nullified the effects of nitrogen fixation by the N-fixing shrubs. Although we did not measure the abundance of herbaceous legumes, we noticed that volunteer sweetclover (Melilotus officinalis Lam.) was prominent in the herbicide-treated tree rows in study 1 during the second season. The seeded birdsfoot trefoil was abundant in studies 1 and 2 in 1995. It is possible that the interplanted shrubs will contribute available N after herbaceous legumes are shaded out by crop trees. Long term studies would be required to test this.

Poor survival of bristly locust in some plots and overall poor growth of peashrub have reduced nitrogen fixation by these shrubs. We do not think that the nitrogen applied during ground cover establishment has prevented a crop tree response to nitrogen fixation by the shrubs. For example, in a study on the same site as study 1 (unpublished data), green ash grew taller with 168 or 336 kg N/ha in addition to N applied when ground cover was seeded.

Survival of bristly locust was variable in study 1 and in an adjacent study (unpublished data) on the same site. In the adjacent study, locust survived best in plots with the least ground cover when locust was planted. There were no obvious differences in ground cover in study 1 that would have contributed to the variation in locust survival. Although sprouting may allow one surviving locust to spread over a large area (Davidson,1979), several plots in study 1 had no surviving locust and thus no possibility of spreading by sprouting. Low initial survival for locust may not be a problem in routine reclamation plantings where locust is planted uniformly over a large area instead of being separated in research plots. To insure locust establishment it may be necessary for plantings to be spaced closer than the 1.8 m by 1.8 m used in study 1. Locust grew vigorously in several plots with abundant sprouts.

Siberian peashrub survived better than bristly locust, but peashrub growth was unimpressive due in part to animal damage. In northern Michigan peashrub survived better than bristly locust on alkaline iron tailings but neither species survived on alkaline copper tailings after five years (Shetron and Carroll,1977). On an acid minesoil in Kentucky, peashrub had 44% survival after four years but did not persist after 18 years (Wade et al,1985).

Conclusions

Tree heights were increased dramatically by tree shelters but it is too early to speculate if the increased growth would offset the costs of installing shelters. Bristly locust does not survive as well as Siberian peashrub but shows more potential for vigorous growth with concomitant nitrogen fixation.

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[1]EVALUATION OF NATURAL SUCCESSION OF RECLAIMED COAL MINE LAND IN WESTERN KENTUCKY

by

Dan L. Williamson and R. Brent Gray[2]

ABSTRACT: In 1989 Peabody Coal Company began a comprehensive inventory program on roughly 2000 acres of reclaimed land in various stages of reforestation. Although the information gathered was intended for in-house use, accurate maps and records were created.

Since the inception of public law 95-87, reclamation managers have discussed their observations that compaction and ground cover requirements make the establishment of tree seedlings extremely difficult and the role that this has played in natural regeneration. To examine this situation more closely an isolated area that had been seeded in 1987 to a tree compatible grass\legume ground cover was selected. The area was tree planted in the spring of 1988 and again in the spring of 1989. The area is approximately 250 acres and is surrounded by unmined remnants of upland forest and cast overburden areas planted to trees in the late 1950's. Trees observed in the unmined area included Red oak (*Quercus rubra*), White ash (*Fraxinus americana*) and Sugar maple (*Acer saccharum*). Trees observed on the previously mined area include Loblolly pine (*Pinus taeda*), White pine (*Pinus strobus*), Sycamore (*Platanus occidentalis*), Black locust (*Robinia pseudoacacia*) and various other reclamation type species planted in the late 1950's.

In August of 1990, 22 plots were run in a random method to determine tree survival. Plots were circular and 50 feet in diameter. Results from this inventory indicated that stocking was 388 trees per acre. Using the map from the 1990 survey the area was resurveyed in July 1995. The 1995 survey indicated an increase in stocking to 427 trees per acre. However, Winged sumac (*Rhus copallina*) and Eastern red cedar (*Juniperus virginiana*) were the only 2 tree species that have regenerated and accounted for the increase in stocking.

Ground cover changed greatly; Redtop grass (*Agrostis alba*) was the only component of the original grass\legume mix present to any extent. Serecia lespedeza (*Lespedeza cuneata*) was present on 17 of the 22 plots and was the dominant vegetation on 8 plots; only 1 plot was free of Serecia lespedeza.

It is our summation that this area will lag far behind pre-law areas in the amount of time required to regenerate to woodland.

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[1] Paper presented at the 13th Annual Meeting of the American Society for Surface Mining and Reclamation, Knoxville, Tennessee, May 22, 1996.

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Additional Key Words: natural regeneration, compaction, ground cover.

Introduction

With the implementation of approximate original contour (A.O.C.) grading in the late 1960's as required by Kentucky state law and the subsequent passage of public law 95-87, the contour of post mining land has changed dramatically in Western Kentucky. Obviously, topography is significantly different. Adherence to approximate original contour (A.O.C.) grading requirements has achieved the desired effect of blending mining areas into the surrounding unmined landscape. Along with the change in topography there is a pronounced change in vegetation cover.

As a rule, pre-law ungraded or strike-off graded sites are in some sort of tree cover, either planted or naturally regenerated. Conversely, graded sites are dominated by herbaceous plants or remnants of tree and shrub plantings and herbaceous plants.

Difficulty with establishing trees on graded sites has been well documented. As early as 1951 concerns were raised that grading could be detrimental to reforestation. In 1954 Kentucky adopted strike-off grading of spoil ridges and it soon became evident to foresters that even limited grading of sites produced a negative effect on reforestation. There have been numerous research studies since public law 95-87 to the substantiate compaction problems associated with grading and ground cover requirements (Vogel and Gray 1987), (Cunningham and Allen 1989), (Hutnik and Hughes 1990) and (Probert 1990). Also, numerous unpublished research studies exist in the Mid-west coal fields.

Reforestation of surface mine sites in Western Kentucky has dwindled since the enactment of public law 95-87. In reviewing Kentucky Reclamation Association, Inc., archives, we compiled a comparison of the total acres that have been revegetated to pasture hayland versus reforestation for the period 1980 through 1995. Since 1980 Kentucky Reclamation Association, Inc., has revegetated approximately 86,000 acres. Less than 1500 acres of this total were reforested and less that 4000 acres of the total received any wildlife plantings. Kentucky Reclamation Association, Inc., was founded in 1948 and has planted nearly 67,000,000 tree and seedlings since its inception. Only 10,000,000 trees and/or shrubs have been planted since 1980, mostly on Peabody Coal Company lands (Table #1).

Erosion control became the primary responsibility of reclamation managers when it became apparent that these graded, highly compacted sites were subject to severe erosion; especially those areas where topsoil was replaced. In order to control erosion, a dense grass\legume ground cover has to be established. The combination of compaction and ground cover made reforestation nearly unattainable for post mining land use. To compound the problem of reforestation, pasture hayland acreage was considered a higher priority for land use than reforestation by the public law 95-87.

Concern that (A.O.C.) and it's associated problems had effectively regulated reforestation out of existence was negated by the assumptions that eventually these lands would return to forest by natural succession. The mining industry and regulatory agencies assumed that land use change to pasture/hayland was only temporary. This assumption was based upon the natural regeneration that had occurred on ungraded pre-law mined areas.

In the early 1980's we began to notice some disturbing problems associated with graded spoil and graded spoil with replaced topsoil. A stand of grass and legumes could be established using proper tillage and soil amendments. The stand would maintain itself for a few growing seasons and then the legume component would die out. Coarse weeds began to replace the legumes and eventually the grasses. What has continued to surprise us is the relative absence of invasion by trees. We are finding that large tracts of reclaimed lands in Western Kentucky are not regenerating to forest but are being invaded and dominated by Serecia lespedeza.

Methods

To examine this situation in more detail we selected a unique site in Ohio County, Kentucky. The site at Peabody Coal Company's Ken mine is an isolated tract approximately 250 acres in size. All reclamation at the site was done under the regulations in Public law 95-87. The site is surrounded by cast overburden spoil ridges that were reforested in the 1950's (Table #2).

The mining area had a pre-mining vegetation survey performed by Dan Williamson in December 1982. The findings were as follows: This area has been extensively cut over within the last 5 years. Because such a large volume of timber was removed, there is not enough residual overstory present to constitute a logical size class ranking. The area is characterized by clumps of 8"-18" diameter at breast height Pignut hickory (*Carya pallida*), Mockernut hickory (*Carya tomentosa*), Shagbark hickory (*Carya ovata*) and Sugar maple with some White oak (*Quercus alba*), Blackgum (*Nyssa sylvatica*) and Sassafras (*Sassafras albidum*). These clumps of residuals are heavily fire scared or of poor form. Surrounding these clumps are large, relatively open sections with dense seedling and sapling size White oak, Red oak, Black oak (*Quercus velutina*), Southern red oak (*Quercus falcata*), Flowering dogwood (*Cornus florida*), Sugar maple, American beech (*Fagus grandifolia*), White ash, and Sassafras with Devils walking stick (*Aralia spinosa*), Greenbriar species (*Smilax sp.*) and Sumac (*Rhus*).

Soils present were predominantly Frondorf-Wellston-Rosine silt loams, 12-20% slopes on the side slopes and Zanesville 2-12% slopes along the ridges.

The area was mined using the "area" method. Topsoil was removed by dozer and scrapers before mining; overburden was removed by a dragline. The number 11 and 13 seams of coal were removed. After the coal was removed the spoil was graded to (A.O.C.) using dozers. Topsoil

was redistributed over the area by dozers and scrapers.

Final grading and seeding of permanent ground cover was completed in the fall of 1987. Soil amendments included 7 tons of agricultural lime per acre incoporated with a disc-harrow to a depth of 6 inches. The fertilizer used was a combination of 400 pounds of 18-46-0 and 200 pounds of 0-0-60 per acre. Seed mixture was a grass and legume mixture comprised of:

Species	**lbs.per acre**
Timothy (*Phleum pratense*)	5 lbs.
Orchard grass(*Dactylis glomerata*)	15 lbs.
Perennial ryegrass(*Lolium perenne*)	15 lbs.
Redtop (*Agrostis alba*)	5 lbs.
Korean lesp.(*Lespedezastipulocea*)	5 lbs.
Red clover (*Trifolium pratense*)	10 lbs.
White clover (*Trifolium repens*)	5 lbs.

In the spring of 1988 the site was planted with approximately 150,000 tree and shrub seedlings. Tree species planted included Loblolly pine, White pine, Virginia pine (*Pinus virginiana*), Eastern redbud (*Cercis canadensis*), Autumn olive (*Elaeagnus umbellata*), Crabapple (*Malus sp.*), Persimmon (*Diospyros virginiana*), Flowering dogwood, Shrub lespedeza (*Lespedeza bicolor*), Bayberry (*Myricaceae*), Silver maple (*Acer saccharinum*), Yellow-poplar (*Liridendron tulipifera*), Green ash (*Fraxinus pennsylvanica*), River birch (*Betula nigra*), Pin oak (*Quercus prinus*), Pecan (*Carya illinoensis*), Black walnut (*Juglano nigra*) and Willow oak (*Quercus phellos*). Planting was done by hand, using dibble bars. No site preparation was implemented. We assumed that the ground cover seeded was very compatible for reforestation and less competitive that a typical fescue\alfalfa mix.

Drought in the summer of 1988 led to poor seedling survival. The area was completely replanted in the Spring of 1989 using tree species similar to those planted in 1988.

Concern for tree survival and regeneration on this area began in 1989. In May 1989, 3, 1 acre observation plots were established. The plots were monitored monthly to determine when planted seedlings were dying; all natural regeneration was also recorded. During the summer of 1989 3 Winged sumac became established by natural regeneration on this study area.

In August 1990 a tree survival inventory was conducted using 22, 50' diameter circular plots. This inventory ascertained that only 3 Winged sumac had become established through natural regeneration and that the stocking rate for planting seedlings was 388 trees per acre.

During the summer of 1991 new observation plots were established to continue our investigation into tree survival. Based upon the finding of the 1990 inventory we determined that 5, 50' diameter circular plots would provide a commensurate example. During the summer of 1991 it was detected that no regeneration had occurred on the 5 plots with the exception of 1 Winged sumac.

We discontinued our observation plots in 1991. We had determined when seedling mortality was occurring and concluded that natural regeneration was not occurring

quickly enough to offset the loss of the planted seedlings.

In August 1995 we reinventoried the area utilizing the same 22, 50' diameter circular plots established in 1990. Stocking had increased to 427 trees per acre; Winged sumac and Eastern red cedar accounted for the increase in stocking. However, of the 22 plots only 10 plots had any natural regeneration. Four of these plots accounted for 85% of the natural regeneration. It was also discovered that seedling mortality was continuing to occur. Stocking for planted trees dropped from 338 seedling to 330 seedlings per acre. Summary as follows:

SUMMARY OF 1995 INVENTORY
(22 50' diameter plots)

plot #	Number & Species
1	1 Winged sumac
2	1 Eastern red ceder
5	8 Winged sumac
6	1 Winged sumac
12	1 Winged sumac
14	23 Winged sumac
15	30 Winged sumac
16	17 Winged sumac
19	1 Winged sumac
20	13 Eastern red cedar

Serecia lespedeza was never seeded on this site. The 1995 inventory found Serecia lespedeza was present on 17 of the 22 plots and was the dominant vegetation of 8 plots and only 1 plot was free of this species. Redtop grass was the only component of the original ground cover still well represented.

Soil samples were taken in 1995 to determine if soil chemistry was responsible for loss of ground cover; water p.H. results were between 6.2 and 6.4.

Conclusion

In our opinion the assumption that natural regeneration of the original pre-mining tree species will readily occur on compacted soil lands reclaimed under Public law 95-87 is inaccurate. High mortality with artificial regeneration on these sites is well documented. Reduced water capacity and impervious rooting zones are the result of unnecessary compaction; ground cover and soil chemistry also limit the chances for tree establishment. Colonization of these sites by highly competitive herbaceous plants like Serecia lespedeza will slow the return of pre-mined tree species.

Acknowledgments

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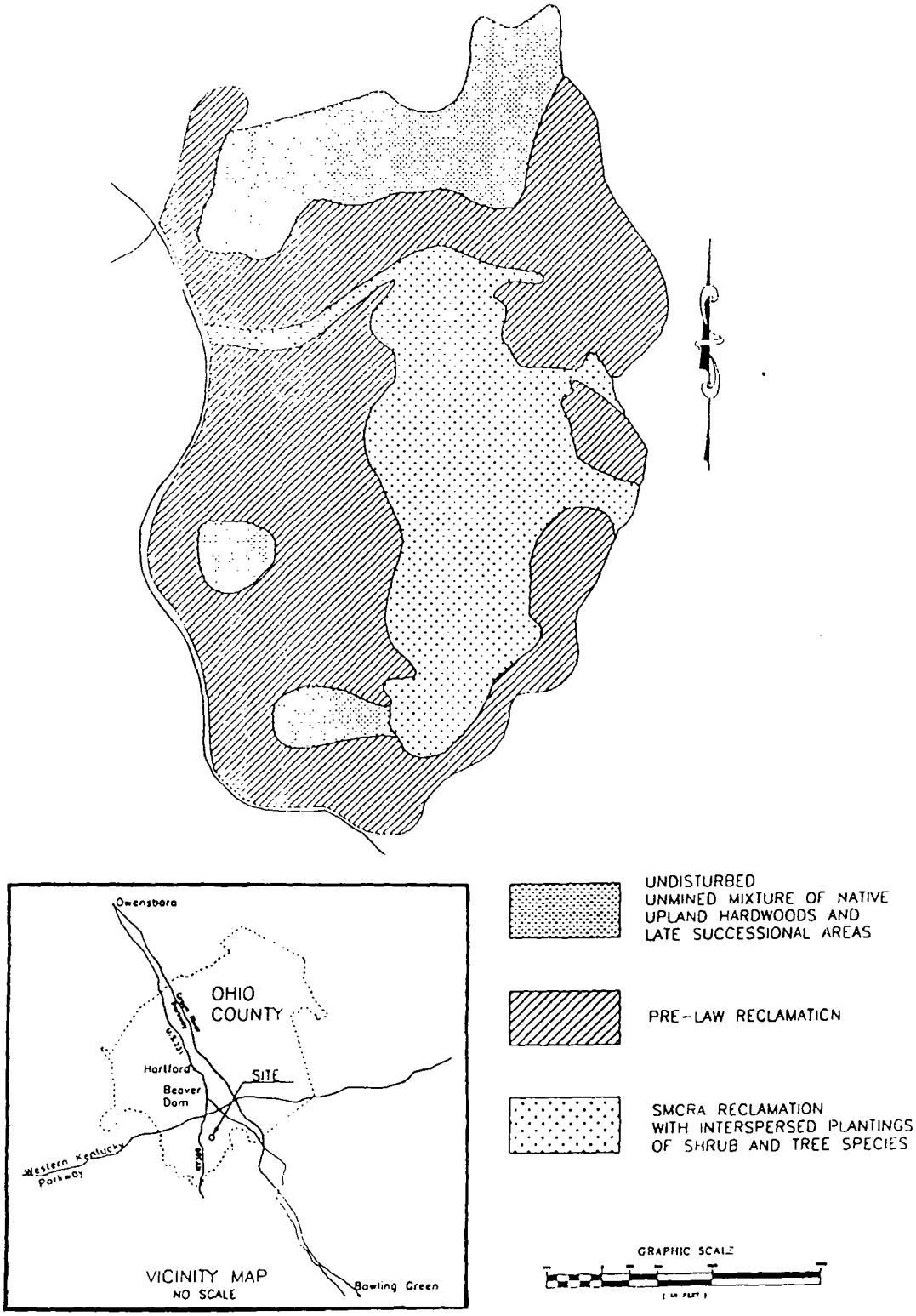
Table #1

Kentucky Reclamation Association, Inc.
Cumulative Summary
1994

Trees and Game Food 1948-1994 Inclusive	Western Kentucky	Eastern Kentucky	Total
Trees and Game Food	65,120,819	2,610,645	67,731,464
Estimated Acres	77,206	3,372	80,578
Forage Crop Seed - Lbs.	7,821,947	1,239,354	9,061,301
Estimated Acres	142,242	28,752	170,994
Estimated Total Acreage Planted and/or Seeded Since 1948*	186,228	31,749	217,977

* Acres reclaimed (planted to trees, planted and seeded both or seeded only).

Table #2


Owensboro
OHIO
COUNTY
Hartford
Beaver
Dam
SITE
Western Kentucky
Parkway
VICINITY MAP
NO SCALE
Bowling Green
UNDISTURBED
UNMINED MIXTURE OF NATIVE
UPLAND HARDWOODS AND
LATE SUCCESSIONAL AREAS
PRE-LAW RECLAMATION
SMCRA RECLAMATION
WITH INTERSPERSED PLANTINGS
OF SHRUB AND TREE SPECIES
GRAPHIC SCALE

INFLUENCE OF GRADING INTENSITY ON HERBACEOUS GROUND COVER, EROSION, AND TREE ESTABLISHMENT IN THE SOUTHERN APPALACHIANS [1]

John L. Torbert and James A.Burger [2]

Abstract: In the southern Appalachian coalfields of Virginia, southern West Virginia and eastern Kentucky, the opportunity exists to create productive forests capable of providing good timber management opportunities on surface mined land. Unfortunately, this opportunity is seldom realized, in part because minesoil compaction caused by excessive surface grading hinders tree establishment and long-term tree growth. This five-year study was established in eastern Kentucky to evaluate the effect of several surface grading treatments on minesoil erosion, herbaceous ground cover development, and tree survival and growth. Three grading treatments were each replicated three times on a 40% slope. The treatments were 1) the operational "intensive" grading practice used by the operator, 2) a more "moderate" grading, and 3) a "ripped" treatment that created a rough surface. During the course of the study, the intensive graded treatment had the greatest erosion and the poorest ground cover development and tree growth. This traditional practice of intensive surface grading to create a smooth surface is not only unnecessarily expensive but, as this study clearly indicates, compaction caused by grading can have detrimental effects for environmental quality (erosion) and forest productivity. When forestry is the post-mining land use, grading practices should be minimized to leave minesoil in a loose condition to reduce erosion and increase tree growth.

Additional Key Words: compaction, minesoils, reclamation, productivity.

Introduction

Since passage of the Surface Mining Control and Reclamation Act (SMCRA), reclamation throughout the southern Appalachians has been conducted in a fashion that creates a "golf course appearance" by making a smooth landscape covered with lush grass. This reclamation scenario has become standard operating practice for most coal operators, and regulators have come to expect smoothly finished surfaces with dense vegetation, regardless of the designated post-mining land use. These practices may be desirable for creating a "hayland/pasture" land use, but they are counter-productive for a forestry land use, an important land use in the southern Appalachians.

Following a 30-year assessment of tree plantings on graded and ungraded spoil in Ohio, Larsen and Vimmerstedt (1983) concluded that spoil compaction was the most important SMCRA-related problem in need of solution for forest-land reclamation. Other researchers have voiced concern that SMCRA encourages excessive spoil grading that results in growth-limiting levels of compaction. Soil compaction increases soil strength, decreases soil aeration and water infiltration, and increases surface runoff. Compacted soils are difficult to plant, seedling mortality is often high, and surviving trees have slower growth rates.

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Minesoil compaction seems most detrimental to tree growth on level areas. After the landforming stage is finished, operators typically do a "*final grading*" to smooth the surface and remove any protruding boulders, large roots, or any other debris that may be included in the spoil material. Finally, before the site is seeded, it is "*walked-in*", "*tracked-in*", or "*trammed*" using a bulldozer to cover the entire surface with indentations from the bulldozer treads. This practice breaks the surface crust that may have developed between the time of grading and seeding, removes any rills or gullies that formed, and creates a uniform distribution of small microsites to capture grass seed and produce a uniform ground cover.

SMCRA does not explicitly require the intensive surface grading that has become so common. Section 816.102 pertains to general requirements for backfilling and grading, and states that "*Disturbed areas shall be backfilled and graded to: 1) achieve the approximate original contour ..., 2) eliminate all highwalls ..., 3) achieve a post-mining slope that does not exceed the angle of repose ..., 4) minimize erosion ..., and 5) support the approved post-mining land use.*" This underlined requirement suggests that grading practices should be land-use specific. We contend that, when forestry is the post-mining land use, level and gently sloping land (where erosion hazard is slight) should be "rough-graded" and left in a loose condition. It should be acceptable for some rocks, depressions, and woody debris to remain on the surface of reclaimed land that will be used for growing trees.

Minesoil compaction effects on tree growth

Early reports on the adverse effects of spoil grading on tree growth were presented by Limstrom (1952) and Chapman (1967). In Ohio, Larsen and Vimmerstedt (1983) found that yellow-poplar height and diameter were 142% and 67% greater after 30 years on ungraded versus graded spoil banks. White pine height and diameter were 32% and 23% greater in the ungraded spoil.

In Illinois, Josiah and Philo (1985) contrasted the physical properties of unmined soil, ungraded spoil, and graded spoil. The bulk density of the ungraded spoil and unmined soil were both 1.3 Mg m^{-3}, whereas the bulk density of the graded spoil was 1.8 Mg m^{-3}. Four years after planting, black walnut (*Juglans nigra*) trees were 35% taller and stem diameter was 31% greater in the ungraded spoil compared to the graded spoil. Where graded spoil was loosened by ripping, height and diameter were increased 38% and 55%.

Torbert and Burger (1990) compared the survival and growth of six commercially important tree species planted on two adjacent slopes, each comprised of the same spoil material. One was operationally regraded and tracked-in, and the other was left in a rough-graded condition. After two years, tree survival averaged 42% on the conventionally regraded site and 70% on the rough-graded slope. For some species, average height growth was almost doubled by eliminating the compacting process.

Surface grading effects on erosion

Rough graded sites are less prone to erosion since the loose soil has a higher infiltration rate (Merz and Finn, 1951). In a discussion about mined land shaping and grading, Glover et al. (1978) listed five practices to reduce or detain surface runoff. First on the list was "roughening and loosening the soil" (followed by mulching and revegetation, topsoiling and use of soil amendments, reduction of slope length or gradient, and use of concave slopes). Minesoil that is left in a loose condition, either as a result of rough grading to avoid compaction, or by ripping to ameliorate compaction, has a greater infiltration rate that decreases overland flow and erosion. Furthermore, the lower strength of uncompacted soils is more conducive to root growth for trees and other plants and ensures that better vegetative cover will be capable of further protecting the soil.

Despite the common-sense knowledge

that loose soils have a greater infiltration rate and less runoff than compacted soils, there is a common belief among some reclamation contractors and inspectors that intensive grading is necessary to reduce erosion. To dispel the belief that intensive grading is necessary to prevent erosion, a research/demonstration project was established in eastern Kentucky to evaluate the effect of surface roughness on ground cover establishment, erosion, and tree growth.

Objectives

The specific objectives of this study were to:

1) compare the influence of three surface grading treatments on erosion. The treatments are i) standard intensive grading, ii) a moderate level of grading, and iii) intensive grading followed by ripping.

2) compare the influence of these three surface grading treatments on herbaceous ground cover development.

3) compare the influence of these three surface grading treatments on survival and growth of five tree species: white pine (*Pinus strobus*), loblolly pine (*Pinus taeda*), sycamore (*Platanus occidentalis*), sweetgum (*Liquidambar styraciflua*), and yellow-poplar (*Liriodendron tulipifera*).

Methods

Site Selection and Treatments

The study is located on land mined by Martiki Coal Corp. near Lovely, KY. A slope, approximately 50 m (upslope) by 500 m (along the contour) had been reconstructed and moderately graded to its final contour (40% slope). The site was not topsoiled. The topsoil substitute was an alkaline spoil derived primarily from gray siltstone with a minor component of gray and oxidized sandstone. Spoil pH before hydroseeding ranged from 7.7 to 8.8. The site was awaiting final surface grading and hydroseeding when it was selected for study in January 1991.

The slope was divided into nine plots (50 x 50 m) that were used for three replications of three grading treatments. The three grading treatments, installed on March 26, 1991 were:

Intensive grading. This treatment was the standard operational practice used by Martiki and other operators in the region. Bulldozers (D-9 Caterpillars) smoothed the slopes by backblading (dragging their blades as they backed downhill), afterwhich the surface was tracked-in by running the bulldozer up and down the slope until the entire surface was covered with indentations from cleats on the bulldozer treads.

Moderate grading. For this treatment, no further grading was applied. Grading already completed when the study site was selected resulted in a fairly smooth surface, although some rocks and rills were present. A hard crust was present at the surface since six months elapsed between the time of grading and seeding.

Ripped. This was an ameliorative treatment intended to mimic the surface conditions that might be created by rough-grading. The initial study design called for a rough grading treatment where the slope would be returned to its approximate original contour with a minimal amount of grading, leaving some boulders, depressions, and loose soil at the surface. Unfortunately, when selected, the study site was already graded to the moderate level. The decision was made to backblade and track-in these plots to the intensive level of treatment, and then ameliorate the compaction by ripping with a 1-m deep subsoiling shank pulled directly downslope by the D-9 bulldozers. Rips were created at 3-m (10 ft) intervals. Ripping created a very rough surface in the immediate vicinity of the rips. Some large boulders (> 1 meter) were pulled to the surface and some deep holes were opened. This treatment did not loosen the entire soil; approximately 1.5 m between the rips remained compacted and unaffected by ripping.

Tree Planting

Trees were planted on April 1-2 1991, less than one week after installation of the grading treatments. Five species of trees were planted: white pine, loblolly pine, yellow-poplar, sweetgum, and sycamore. Approximately 40 of each pine species and 20 of each hardwood were planted in each plot on a 3 m x 3 m (10 ft x 10 ft) spacing. Two rows of each hardwood species and four rows of each pine species were planted. All species were 1-year-old seedlings except white pine which were 2-years-old. Trees in the ripped plots were planted in or very near the rip. After the fifth growing season, tree heights were measured and survival was determined.

Ground Cover Establishment

On April 16, two weeks after tree planting, a "tree-compatible" ground cover (Table 1) was established by hydroseeding. After the first and fifth growing seasons, three 30 m (100 ft) transects were installed along the contour of each plot, approximately one-fourth, one-half, and three-fourths of the distance from the bottom to top of the slope. At 0.6 m (2 ft) intervals along the transect, a 2.5 cm sighting tube was used to assess ground cover. If more than half the area observed through the tube consisted of bare soil, the point was tallied as such. If more than half the area was vegetated, the vegetation was tallied as grass or legume. This was done at 150 points per plot.

Erosion Measurements

Soil movement from the slope and deposition at the toe of the slope were monitored by measuring the changes in the distance between the soil surface and the top of metal rods installed in each plot. Three rows of metal rods were installed along the contour of each plot, approximately one-fourth, one-half, and three-fourths of the distance from the bottom to top of the slope. Ten rods were installed in each row, 3 m (10 ft) apart from each other. The above-mentioned vegetation transect was conducted along these rows of rods. A row of measurement rods was also placed at the toe of each plot on the level area. Rods were measured in October 1991, and October 1993, after the first and third growing season.

Results

The study was conceived with the goal of quantifying the amount of erosion and the response of herbaceous and woody vegetation to rough grading on a minesoil with desirable chemical properties for tree growth. Unfortunately, the ripped treatment did not accurately imitate rough grading because the area between rips was still compacted and the area within the rip was rougher than desired. The objective of rough grading is to leave an uncompacted surface that is similar to the surface of natural forest land with respect to undulations of the surface and the presence of rocks and boulders on the ground.

The ripped treatment in this study represented what many people would consider to be a worst-case scenario. If rough grading were likely to increase erosion, then deep ripping directly up and down a steep slope would surely represent a worst case opportunity for erosion to occur. Although the authors believe oxidized (brown) sandstone spoil is the best medium for tree growth in the southern Appalachians (Burger and Torbert 1990), and oxidized sandstone existed at this mining site, it was not possible to locate experimental plots where oxidized sandstone was placed at the surface. Unlike most coal operators in the southern Appalachians, Martiki's spoil handling is mostly done with a dragline. As such, the spoil that exists closest to the surface (oxidized sandstone) is placed immediately at the bottom of the adjacent, previously mined pit. In this situation, the best overburden for trees is buried about 20 m below the surface, and the spoil that exists immediately above the last coal seam will almost always end up on the final surface. For operations such as this, where the desired overburden material for forest productivity is not

economically available, a research challenge exists to develop alternative reforestation strategies. The hardwood species used in this experiment were selected to evaluate their performance on near-neutral pH minesoils.

Ground Cover Establishment

All plots had a southerly aspect, which resulted in high surface temperatures on the dark gray spoil during the summer in the first year. The site received no rainfall from July to September in the first year, and rainfall in June occurred as short thunderstorms. As a result, ground cover was very sparse during the first year. Average ground cover after the first year was only 44% and unaffected by grading treatment. After five years, average ground cover increased to 86% and was still unaffected by treatment (Table 2). Most of the ground cover during the first three years was provided by weeping lovegrass (*Eragrostis curvula*), a heat-tolerant species. On some of the moderate and intensively graded plots, weeping lovegrass was almost the only species at the end of the first growing season. Although it is generally considered to be an acid-tolerant species (Vogel 1981) it survived very well on this dark, droughty, alkaline spoil.

Average legume cover was only 12% after the first year and 61% after three years. Birdsfoot trefoil (*Lotus corniculatus*) was the predominant legume species during the first three years, but crownvetch (*Coronilla varia*) was the most common after the fifth year. For the most part, ground cover on the moderately and intensively graded treatments was uniformly distributed across the plot. On ripped plots, however, values in Table 2 are a weighted average of the relatively sparse vegetation that existed in the area between rips, and the relatively vigorous vegetation that existed within the loose soil of the rips. Within a 1-meter band up the rip, ground cover was 100%.

Minesoil erosion

Compaction on intensively graded plots increased surface runoff. During a thunderstorm in June of the first year, surface runoff was observed flowing off the slope at the base of the intensively graded plots. On the ripped plots, there was overland flow occurring on the compacted soil between individual rips, but within the rips and the soil immediately adjacent to the rip, water was absorbed into the soil and no overland flow occurred. No water flowed from the rips at the bottom of the slope. Rips did not develop into gullies during the course of this study. The surface roughness created by ripping diminished as soil next to the rip moved into the rip and collected.

Erosion was quantified by two approaches: 1) as the average depth of spoil lost from the 30 erosion rods on each plot, and 2) by the average deposition at the erosion rods at the base of the plot (Table 3).

After the first growing season, erosion, based on measurements from erosion rods in the treatment plots, was 18 times greater in the intensively graded treatment than in the ripped treatment, and almost twice as high as the moderate-graded treatment. Variability among treatment replicates was so high that these differences were not statistically different, based on an analysis of variance and a probability level of 0.1.

Erosion results from third year measurements are confusing and probably unreliable. They indicate that no erosion occurred during the course of the study and that soil accumulated on the slope since year one. Measurements of the rods indicate a net deposition of approximately 0.4 cm on the slope in the ripped and moderate graded plots and no deposition or erosion from the intensively graded plots. Perhaps some loosening or swelling of the soil occurred as a result of disintegration of rock fragments or freezing in the winter.

Erosion appraisal based on a

measurement of deposition at the base of the slope seems to be a more meaningful assessment, although there are still some unexplainable effects between the first and third years. If soil freezing and expansion confounded results on the slope, it would also affect results at the base of the slope. After the first growing season, deposition at the base of the intensively graded plots was about 130% greater than the ripped treatment, and almost 600% greater than the moderately graded treatment. After three years, deposition at the base of the intensively graded plots was about 20% greater than the ripped treatment, and 118% greater than the moderately graded treatment.

Our method of measuring erosion does not provide an opportunity to calculate the Mg ha^{-1} of soil movement from the slope with any definitive level of confidence. Results are still valid however for accomplishing the fundamental purpose of this study: to compare the relative advantage or disadvantage of intensive grading to other treatments which leave the surface in a looser condition. It is reasonable to conclude that intensive grading did not prevent or decrease the amount of erosion compared to the treatments with looser soil. It's obvious that the intensively graded treatment had as much or more erosion than the moderate or ripped treatments. These results should help dispel the idea that intensive grading is beneficial for reducing erosion.

Tree survival and growth

The study included two species of pines (white and loblolly) and three hardwoods (sycamore, sweetgum, and yellow-poplar). It was known beforehand that the pH would be undesirable for pine growth, and thus the hardwood species were included. They were selected because of their reported ability to tolerate neutral-pH to slightly alkaline soils (Vogel 1981).

As anticipated, the pine species suffered on this minesoil (Table 4). Only 3% of the white pines survived to the fifth year. Almost all of the loblolly pines appeared dead within two months of planting, but by age five, 29% were still alive. The surviving loblolly pines were healthier than white pines, but still not vigorous. Foliage on loblolly pines was yellow, and some trees displayed reddish coloration at the tips of needles suggesting high soluble salt concentrations. The overall poor general health of these trees did not provide a good opportunity to evaluate the effect of these grading treatments on pine growth. Nonetheless, the height of loblolly pines was significantly lower on the intensively graded treatment than the moderate graded and ripped treatments (64, 120, and 99 cm, respectively).

Sycamore performed better in this study than any other species, and because it was relatively tolerant of the minesoil chemical conditions, it provides the best demonstration of compaction effects on survival and growth. In the ripped and moderate graded treatments, sycamore survival averaged 70%, but in the intensively graded treatment, survival was only 50%. Growth followed the compaction gradient created by treatment with the best growth occurring on the ripped treatment. Compared to the intensively graded treatment, average height was about 34% greater in the moderate treatment, and 74% greater in the ripped treatment.

Survival of sweetgum and yellow-poplar was significantly greater in the ripped versus intensively graded treatment. Only 3% of yellow-poplar survived in the intensively graded plots whereas almost 70% survived in the ripped plots. Growth of yellow-poplar was improved about 78% by ripping, but even in the ripped treatment, growth was not good for yellow-poplar or sweetgum. Overall poor growth was attributed to chemical properties of the minesoil. Even though these species are tolerant of a high pH, they would grow better in minesoils with a pH more typical of natural forest soils.

Conclusions

The study was established to compare the traditional intensive grading treatment with a less intensive "rough grading" treatment with respect to erosion, ground cover development, and tree performance. The experiment did not include a treatment that fairly represented rough grading conditions. The ripped treatment was an ameliorative treatment that mimicked rough grading to some extent. Ripping did not loosen soil between the rips, and the loose soil in and immediately adjacent to the rip was much rougher than would result from rough grading. The ripping process pulled some boulders (more than one meter in diameter) to the surface, creating huge depressions and open chasms that could be hazardous to people walking on the site.

Even though this study did not provide the opportunity to quantify the beneficial effect of a rough-graded oxidized sandstone, it is useful for concluding that the intensively graded treatment, which represents the standard operating practice for most coal operators in the Appalachians, did not provide any improvements in erosion control, ground cover establishment or tree survival and growth. All measures of performance were lowest on the intensively graded treatment. In some respects, the ripping treatment provides the opportunity to assess what many would consider a worst-case scenario: ripping a steep slope (where erosion potential was high) with rips oriented directly up and down the slope. The fact that ripping did not result in gullies or any observable increase in erosion provides evidence that the presence of loose soil does not necessarily lead to increased erosion.

Other researchers have found ripping to be beneficial to improving tree rooting and growth, and have recommended ripping as a standard practice where trees are planted on minesoils (Josiah and Philo 1985, Berry 1985). Although ripping can ameliorate compacted soils, the wiser approach would be to avoid the compaction in the first place. Ripping is an expensive operation which, if necessary to achieve decent tree survival and growth, could discourage operators from selecting forestry as a post-mining land use. Furthermore, ripping can create hazards by uplifting large boulders and creating holes that could be dangerous for anyone walking on the site.

We believe all aspects of reclamation, including final surface grading, should be land-use specific. In order for operators to satisfactorially reclaim surface-mined forest land to a level of productivity at least as high as prior to mining (as required by SMCRA) and to reduce erosion caused by surface runoff, operators should reduce the intensity of bulldozer activity when preparing the final soil surface. Reduced grading will not only decrease the operator's reclamation costs, but it will result in a looser, more productive soil which improves the success of tree planting, long term growth, and environmental protection.

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Table 1. Ground cover species and seeding rates used in experiment.

Species	Application Rate (kg ha^{-1})
Winter rye (*Secale cereale*)	10
Perennial ryegrass (*Lolium perenne*)	5
Orchard grass (*Dactylis glomerata*)	5
Weeping lovegrass (*Eragrostis curvula*)	3
Redtop (*Agrostis gigantea*)	3
Kobe lespedeza (*Lespedeza striata* var Kobe)	5
Appalow lespedeza (*Lespedeza cuneata* var Appalow)	5
Birdsfoot trefoil (*Lotus corniculatus*)	5
Ladino clover (*Trifolium repens*)	3
Crownvetch (*Coronilla varia*)	1

Table 2. Ground cover (%) after five years on a reclaimed minesoil in Kentucky as affected by grading treatment.

Grading Treatment	Total Cover (%)		Legume Cover (%)	
	average	range	average	range
Moderate	90	71-100	68	42-92
Intensive	85	62-97	61	33-81
Ripped	83	68-95	54	36-82

Table 3. Average soil loss and deposition after the first and third growing seasons on a reclaimed minesoil in Kentucky as affected by grading treatment.

Grading Treatment	Soil Loss from Slope (cm)		Soil Deposition at Base (cm)	
	Year 1	Year 3	Year 1	Year 3
Moderate	.44	-.39	0.50 b	0.85
Intensive	.72	0.0	3.48 a	2.21
Ripped	.04	-.38	1.51 ab	1.84

Values within a column followed by different letters indicate a statistically significant difference according to Duncan's Multiple Range Test at a probability level of 0.10.

Table 4. Survival and height of five tree species after five years on a reclaimed minesoil in Kentucky as affected by grading treatment.

Tree Species	Ripped		Moderately Graded		Intensively Graded	
	Survival - % -	Height - cm -	Survival - % -	Height - cm -	Survival - % -	Height - cm -
White pine	2	42	7	51	0	--
Loblolly pine	43	99 ab	32	120 a	15	64 b
Sweetgum	74 c	61	41 b	83	39 b	52
Yellow-poplar	69 a	79 ab	38 b	99 a	3 c	44 b
Sycamore	77 a	142 a	63 ab	109 ab	50 b	82 b

Values for survival and height within a species/grading treatment combination followed by different letters indicate a statistically significant difference according to Duncan's Multiple Range Test at a probability level of 0.10.

SESSION 11: GEOCHEMISTRY OF ACID MINE DRAINAGE

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NEUTRALIZATION AND RELEASE OF METALS PROPERTIES OF FLY ASH[1]

by

Hrissi K. Karapanagioti[2] and Asmare Atalay[3]

Abstract. The neutralization property and the release of metals for three types of coal combustion ash were evaluated. Fly ash (FA), fluidized bed ash (FBA), and hydrated fly ash (HFA) were assessed for their variation in physical, chemical and mineralogical properties. Acidic solutions of different pH were used to equilibrate the ash in conservative and nonconservative systems. Release studies were conducted for the following metals: iron, manganese, zinc, magnesium, aluminum, calcium, cadmium, sodium, chromium, copper, and nickel. Data indicated that pH was a very significant factor for the release of metals from the fly ash surface. At high pH iron was the least released cation from fly ash, whereas calcium was the most desorbed cation. Comparison of the three ashes indicated that FA and FBA were excellent buffering materials. However in the case of FBA, there seemed to be a critical pH at which a drop in the proton titration curve was apparent, and large amounts of metals were released into solution. On the other hand, HFA showed lower buffering intensity compared to the other two fly ashes. Changes in pH with time were also studied since time had an impact on the titration curve at the equivalence point. Although pH was increasing with time, it had no impact on the titration curve after neutralization had occurred. Comparison of fly ash with limestone for neutralization properties revealed that fly ash had less buffer intensity than limestone, but it gave solutions a higher initial pH.

Additional Key Words: Neutralization, Desorption, Metals, Fly ash.

Introduction

Fly ash is a residue left after coal has been burned; it is collected from gas stacks using specialized devices. The properties of fly ash are diverse depending on the nature of the coal and the combustion process. One of several characteristics of most fly ashes is their alkalinity. Some fly ashes have pH as low as 4.5 while others have pH as high as 12.0. The subbituminous and lignite coal ashes produce alkaline solutions upon contact with water. Alkaline fly ash is associated with the existence of minerals such as calcite, amorphous silicates, hematite, quartz, mullite, metal oxides, and free carbon (EPA, 1986). According to EPRI (1988) the alkalinity depends on the calcium content since this element is in the form of highly reactive CaO. The contribution of calcium to alkalinity can be described by the following equation:

$$CaO + H_2O \rightarrow Ca(OH)_2 \qquad (1)$$

Theis and Wirth (1977) have mentioned that soluble calcium, which is associated with the lime fraction, represents the basic component of fly ash. The pH measured for several 1:1 H_2O:fly ash suspensions generally

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shifted with time (EPRI, 1993). Initially, the pH may be lowered by a condensate of sulfuric acid on the surface of the ash particles. Later a rapid rise in pH is observed which is caused by the neutralization of the sulfuric acid via dissolution of alkali and alkaline earth oxides (e.g. CaO, MgO, K_2O, and Na_2O). These oxides are present in the samples in excess of the sulfuric acid condensate. In such cases, the pH measurements should be done after a minimum of 24 hours. Other studies (EPRI, 1993) have reported changes in pH of fly ash suspensions up to 21 days. There seems to be a correlation between ash pH and the amount of sulfate in the saturated paste extract (Daniels et al., 1993); the lower the amount of sulfate present, the higher is the pH of the fly ash slurry. The quantity of fly ash added to an aqueous solution has significant impact on pH. For instance, Reed et al., (1976) reported that decreasing the percentage of fly ash from 1.00 % to 0.03 % only decreased the pH by one unit. That means the addition of a small quantity of fly ash can result to a significant increase in the solution pH.

The desorption of trace metals from fly ash surfaces in aqueous solutions follows a predictable pattern of decreasing release with increasing pH (Theis and Wirth, 1977). Most trace metals show minimum release at pH values around 9. The degree of desorption of trace metals from the fly ash surface is determined by the extent of solubilization of the oxides they are attached to. On the other hand, zinc is quite soluble in the resulting solutions, but it is very poorly desorbed in the neutral pH range. Lead is relatively insoluble and yet it is released from fly ash to a greater extent than other more soluble species. Large amounts of chloride and sulfate are typically released from fly ash, and it is probable that soluble inorganic complexes are formed (Theis and Wirth, 1977). Trace metals bonded with the surface silica will be released only through the action of long-term weathering processes. Iron oxide was found to control sorbed trace metals more than aluminum oxide. Manganese oxide was found to have a greater sorptive capacity than iron oxide, but it is present in comparatively very small amounts. On the average, Theis and Wirth (1977) have associated each trace metal with a certain oxide. From their observations, it is apparent that in order to prevent the release of trace metals, the dissolution of iron and manganese should be controlled. Desorption of trace metals from fly ash by leaching alone is generally low. To increase the effectiveness of leaching, the use of higher solvent volume, higher temperature, longer contact time and lower solid-to-liquid ratios are required. However, it is not probable that the above conditions will occur in natural systems (Burnet, 1987). Fruchter et al. (1990) studied the effect of solubility on metal release from fly ash samples. Calcium and sulfur were found to be the major soluble elements in pore waters and leachate. Most of the solubility controlling solids were found to be (in the referred article) sulfate and hydroxide compounds.

Materials and Methods

Materials

Both fly ash (FA) and hydrated fly ash (HFA) samples were shipped in plastic containers from Oklahoma Gas and Electric Plant, Muskogee, OK. Fluidized bed ash (FBA) samples were generated at the Shady Point Power Plant, Latimer County, OK and shipped to us in plastic containers by Brazil Creek Minerals, Inc., Fort Smith, AR. The limestone that was used for a comparison study with fly ash was purchased from Fisher Scientific Company, Pittsburgh, PA.

Sample Preparation

Two sulfuric acid solutions were prepared, one at pH=1 and another at pH=4. These solutions were used as titrants along with double deionized water (DDI water), which had been purified with a triple-bed filter consisting of cation, anion, and organic exchangers. A series of 10 g samples of fly ash and limestone ($CaCO_3$) were weighed and placed separately in high density polyethylene flasks. For the first batch of samples, different amounts of prepared acid solutions, ranging from

20 to 1900 ml, were added in each flask. The flasks were placed on a shaker and shaken at constant speed for variable time periods. Each flask was tightly capped. In the end, the pH was measured using an Orion pH meter, model 470A. For the first three days sample pH was measured every day; later, the measurements were taken once in three days. When the pH had stabilized, samples were assumed to have reached equilibrium. This method simulates conservative conditions which can be found in systems where the water enters the system and stays for a long period of time (Corbitt, 1990).

For the second batch of samples, a continuous leaching method was employed. In this case, 200 ml of prepared acid solutions or double deionized water were added in the flasks containing 10 g fly ash samples. The flasks were shaken for two days and pH measurements were taken as described above. This method simulates nonconservative conditions which can be found in systems where the water enters but stays only for a short period of time (Corbitt, 1990). The samples were then filtered and fresh acid solutions added to the remaining fly ash samples. This procedure was repeated until the pH of the fly ash solution was below 2 or 5 depending on the initial pH of the sulfuric acid solution. In both batches, the samples were filtered with Whatman No. 2 filter paper. The solutions were digested with nitric acid using a Tecator Digestion Apparatus. Each digested sample was diluted to 100 ml with DDI water containing 0.2 % lanthanum oxide. The diluted samples were stored in high-density polyethylene bottles until they were analyzed for metals.

Sample Analysis

Digested samples were analyzed for metals using Buck Scientific Atomic Absorption Spectrophotometer VGP System Model 210. High concentration standard metals (1000 mg/l) were purchased from Fisher Scientific and used to prepare diluted standards for calibration. Nitrous oxide/acetylene flame was used to measure aluminum concentration. All other metals were analyzed with air/acetylene flame as suggested by Welz (1985).

Results and Discussion

Neutralization Capacity of Fly Ash

The fly ash used in this study was considered alkaline, and titration curves were generated for its neutralization with mineral acids. Three parameters were monitored that evaluated the neutralization capacity of fly ash: pH, amount of acid added, and time. The results were separated into two groups based on conservative or nonconservative control conditions as described in the materials and method section.

Buffer intensity is one characteristic of fly ash which defines its neutralization capacity. In this case, buffer intensity is defined as the number of moles of strong acid required to change the pH of the solution by one pH unit (Benefield et al., 1982). Buffer intensity is best represented by a differential such that:

$$\text{Buffer Intensity} = dC / dpH \qquad (2)$$

where dC = differential quantity of strong acid added to the solution.
dpH = differential change in pH due to the addition of a dC amount of strong acid.

High buffer intensity means strong resistance to pH change. The pH value where the solution presents high intensity is where the titration curve is flat, and tends to remain in that range even after continued addition of the titrant. The end point is another characteristic that can be determined from titration curves. The end point may be defined as the point where the amount of alkali material, for instance has been neutralized by the added acid and the solution starts to become acidic. The amount of acid needed to neutralize a given amount of fly ash provides useful information for the utilization of fly ash in remediation of acid impacted environments. In order to calculate the end point of the titrations and the Calcium Carbonate Equivalence (CCE), the method proposed by

Table 1: Buffer characteristics of three fly ashes under a conservative (contained) and noconcervative (free flow) conditions. (CCE: calcium carbonate equivalence).

Titrator	Type of fly ash or $CaCO_3$	Conservative system		Nonconservative system	
		pH with higher buffer intensity	ml acid to reach end point (%CCE)	pH with higher buffer inten-sity	ml acid to reach end point (%CCE)
0.1 M H_2SO_4	FA	10.5, 8.5	450 (45)	4	250 (25)
	FBA	12.5, 9.8	500 (50)	12.5, 12	550 (55)
	HFA	10.5	500 (50)	4	550 (55)
	$CaCO_3$	6.5	1000 (100)	-	-
10^{-4}M H_2SO_4	FA	10.5	1300 (0.13)	5.5	800 (0.08)
	FBA	11.5	850 (0.08)	12.2	1100 (0.11)
	HFA	10.5	550 (0.05)	6.5	550 (0.05)
double deionized water	FA	-	-	12, 10	-
	FBA	-	-	12, 10	-
	HFA	-	-	12, 10	-

Benefield et al. (1982) was employed. This method involves calculation of the second derivative of the pH versus the volume added, as well as conversion of the volume of acid added to equivalence of calcium carbonate.

Table 1 presents the titration characteristics of fly ash under conservative and nonconservative conditions. For the conservative conditions the systems were allowed an average of seven days in order to reach equilibrium, whereas for the nonconservative conditions the solutions where repeatedly renewed every two days.

When the titrator was 0.1 M sulfuric acid, and the pH was raised to > 8.5, FBA showed higher buffer intensity than either FA or HFA. At low pH (i.e. pH = 4) and nonconservative conditions, FA and HFA maintained higher pH values than FBA. Use of $CaCO_3$ under conservative conditions provided twice the buffering intensity of any of the fly ashes. As the concentration of H_2SO_4 was diluted to 10^{-4} M, FA was a better buffer at pH > 10.5 in the conservative systems. Under nonconservative condition the 10^{-4} M sulfuric acid reduced the pH of FA to 5.5 and HFA to 6.5 where as the pH of FBA remained 12.2.

The high volume of 10^{-4} M sulfuric acid (1100 ml) needed to reach the equivalence point with FBA further indicates that FBA is a better buffer than either FA or HFA.

Figure 1 presents a comparison of the buffering potentials of FA, FBA, HFA, and calcium carbonate when titrated against sulfuric acid of pH=1.

The stepwise titration curves observed for each component is indicative of its buffering behavior. For instance, FA demonstrates several

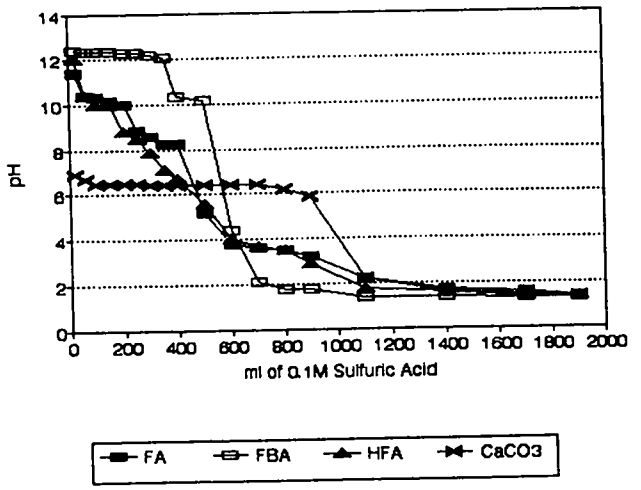

Figure 1 Titration curves for 10 g of FA, FBA, and $CaCO_3$ against 0.1M H_2SO_4 at pH=1 in a conservative system with an average equilibration time of 7 days.

steps before it reaches the end point whereas FBA demonstrates two flat titration steps, and the curve is very steep at the point of neutralization. HFA demonstrates a smooth and continuous neutralization curve with very short titration steps, almost linearly decreasing in each buffering potential. The greatest buffering was obtained when pure $CaCO_3$ was used as a source of alkalinity. In this particular study, the buffering power of $CaCO_3$ is directly related to its equilibrium reaction with added sulfuric acid:

$$CaCO_3 + H_2SO_4 \rightarrow CaSO_4 + H_2CO_3 \quad (3)$$

$$H_2CO_3 \rightarrow H^+ + HCO_3^- \quad (4)$$

$$HCO_3^- \rightarrow H^+ + CO_3^{2-} \quad (5)$$

In reaction (3) gypsum is formed, which traps all of the SO_4^{-2} as $CaSO_4$ and all the hydrogen ions react with the CO_3^{-2} species to form H_2CO_3 in this manner all of the acid is neutralized effectively. Carbonic acid is a weak acid with $pKa_1=4.2x10^{-7}$ for reaction (4) and $pKa_2=4.8x10^{-11}$ for reaction (5). The resulting carbonate-bicarbonate system was able to buffer the acidity much longer (until 1000 ml of the acid was added) than any of the fly ashes. However, once the buffer was exhausted with continuous additions of the acid, the pH quickly dropped from approximately 6.5 to 2.0. Limestone has a Calcium Carbonate Equivalence (CCE) of 100%, whereas fly ash has a CCE of about 50%.

Figure 2 presents the titration curves for FA, FBA, and HFA titrated with sulfuric acid of pH=4. Dramatic changes in the titration curve patterns are apparent. All three ashes showed similar buffering potential when the acid load was low. In terms of utilizing these ashes for treating acid mine drainage any of

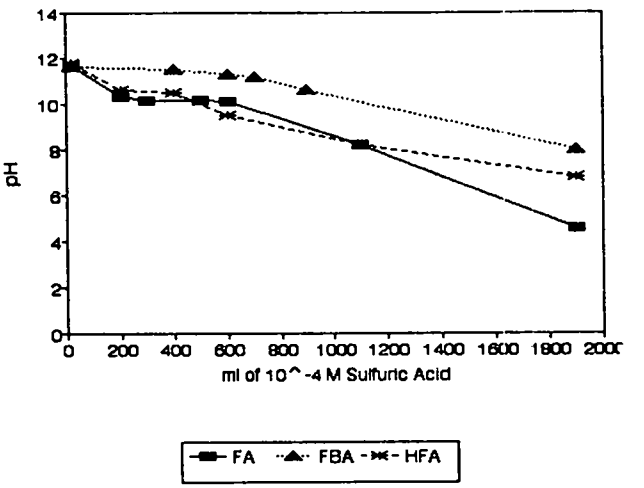

Figure 2 Titration curve for 10 g of FA, FBA, and HFA against 10^{-4}M H_2SO_4 at pH=4 in a conservative system with an average of equilibration time of 7 days.

them could be used. However, FBA shows greater buffering with increased addition of the diprotic acid. After the addition of almost 2 liters of 10^{-4} M sulfuric acid, the pH in the initial 10 g FBA was only reduced from 12 to 9. Comparing the two sulfuric acid solutions of pH 1 and 4 (Figures 1 and 2), it is apparent that the solution with pH 1 causes more release of neutralizing material from the fly ash than the solution with pH 4. It appears that FA and HFA have a limited amount of buffering material, which they release constantly under low acidity conditions. FBA, having a large amount of buffering material, shows similar behavior when exposed to mineral plus acid solutions of pH 1 and 4. In the nonconservative system

the titration curves were very similar to those simulating the conservative system (Titration curves not shown).

The contribution of contact time towards pH stabilization (buffering) was studied in detail using sulfuric acid as the titrant for the conservative system. Figure 3 presents variation of pH with time for FA equilibrium with different quantities of acid. The data points at the beginning of each titration curve reflect the inherent variability in the source of alkalinity in fly ash. The reason for this variation is not clear, but could be caused by interactions between the various cations and anions which are released from the fly ash surface during the initial reaction with the added acid. This finding is in agreement with a similar observation made by Theis and Wirth (1977). After neutralization, the points of the titration curves reached their equilibrium value (indicated by the long plateau

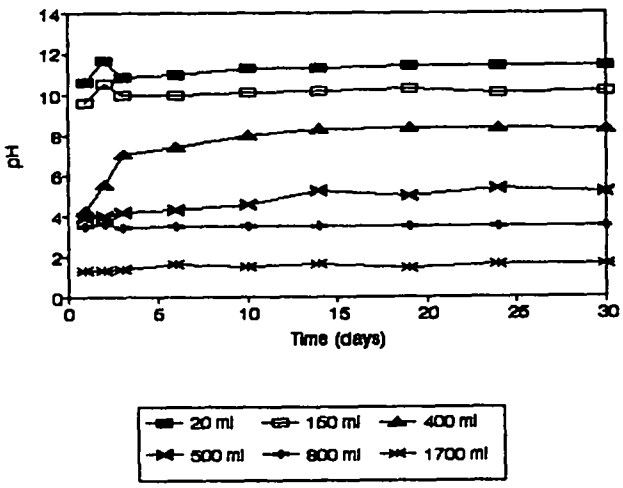

Figure 3 pH vs time for titration of 10 g FA samples against 0.1M H_2SO_4 at pH=1 in a conservative system.

region) the first day. Although fly ash may have released all of its buffering material initially in order to neutralize the added acid, its buffering potential was not significantly reduced with time upon the addition of 400 ml of the acid. When the amount of acid added was increased from 500 ml to 1700 ml, the fly ash was overwhelmed and its buffering capacity was exhausted. As the result, the pH of the solution remained below 4.0. This observation agrees with those made by other investigators (EPRI, 1993). The highest shift in pH observed was 4 units for FA sample titrated against 400 ml of sulfuric acid.

From the discussion presented, it may be inferred that the controlled conditions that were created to effect sufficient contact time between the fly ash and the acid were not critical factors that change the chemistry of the system with high acidity. For systems with low acidity, contact time and quantity of buffering material present in fly ash could be important. Given sufficient equilibration time, more alkalinity might be released from fly ash to neutralize the acidity present in the solution. Fly ash with high alkalinity content can keep the pH of acidic solutions high by releasing all of its buffering materials until no more alkalinity is left, at which time the pH drops suddenly.

FA, FBA, and HFA were titrated with DDI water of pH 5 simulating a nonconservative system. Under conditions where fly ash comes in contact with non-acidic solutions, the pH of these solutions will be influenced by the added alkaline material. The double deionized water had a pH of 5, as the result, addition of fly ash raised the pH of the solution above 10 almost immediately.

Release of Cations from Fly Ash

Conservative and nonconservative systems served as control conditions to study the release of metals from surfaces of uncontaminated fly ashes. A sulfuric acid solution of pH=1 was used. Such a study was conducted to find out the type and quantity of metals and their potential release from raw fly ash material as the pH of the solution was gradually decreased. It appeared that each metal had a specific pattern of release from fly ash which is related to pH. In the conservative system with sulfuric acid as the titrant, as the pH

Table 2: Higher Concentrations of Metals Released in 0.1M Sulfuric Acid Solutions.

Metals	Conservative Systems			Nonconservative Systems		
	FA	FBA	HFA	FA	FBA	HFA
	Maximum Amount Released in Solutions (mg/kg)					
Calcium	85000	95000	95000	12000	26000	12000
Sodium	5720	1140	7980	660	120	1540
Nickel	-	-	-	28	14	2.2
Magnesium	45000	15500	35000	22000	11600	32000
Lead	33	57	38	4	7	12
Manganese	150	475	225	62	200	100
Copper	176	42	171	64	4	42
Zinc	175	25	375	75	50	90
Aluminum	70000	5000	60000	18000	4000	20000
Iron	27500	20000	20000	9000	4000	9000
Chromium	19	19	38	16	6	8

decreased, the order of release of metals from fly ash was as follows:

Ca-Na-Ni-Mg-Pb-Mn-Cu-Zn-Al-Fe-Cr

(most released at high pH) (most released at low pH)

calcium and sodium were easily released into solution at high pH, whereas iron and chromium need very acidic conditions for their release. In the nonconservative system, the order of release was similar to that observed in the conservative system. However, for each addition of acid a different metal showed a maximum release. In each system, FA presented the highest concentration released for nickel, copper, chromium, and iron; FBA for manganese and calcium; HFA for magnesium, aluminum, zinc, sodium, and lead (Table 2).

In the conservative system, the highest amount of iron released into the solution occurred after the addition of 1900 ml of acid solution to 10 g of FA (Figure 4). The highest concentrations of iron in solution were found after the addition of 900 and 1100 ml of nitric and sulfuric acid, respectively. By contrast, in the nonconservative system the concentrations of iron released were the highest after the addition of 1200 ml of the acid. Figures 4 and 5 present the quantity of iron released from FA, FBA, and HFA under conservative and nonconservative conditions respectively. In the conservative system, when pH=1 H_2SO_4 solution was used as the titrant, iron started to be released from FA after 500 ml of the acid has been added and the pH approached 4.0. In the nonconservative system, where pH=1 H_2SO_4 solution was used as the titrant, iron was released from FA at pH=2, which is lower than that observed in the conservative system. This was true for all three different ash materials. Almost no iron was released from the solution until the pH of maximum release (pH=4) was reached. Also, in the nonconservative system, where the contact time between the acid and the ash was shorter high volume (1200 ml) of the acid was needed to effect iron release.

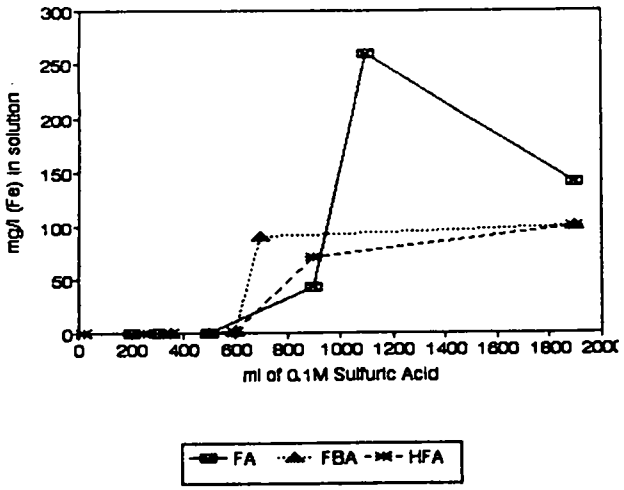

Figure 4 Iron release from 10 g FA, FBA, and HFA under conservative system using 0.1M sulfuric acid.

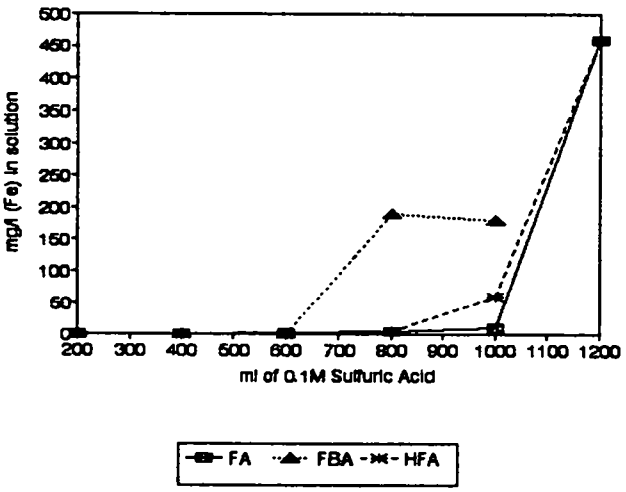

Figure 5 Iron release from 10 g FA, FBA, and HFA under nonconservative system using 0.1M sulfuric acid.

Figures 6 and 7 present calcium release from FA, FBA, and HFA using a similar acidic environment as in Figure 4 and 5, respectively. In the conservative system, when sulfuric acid was used as the titrant, calcium release depended on the amount of acid present in the solution. Calcium started to be released as soon as titration began regardless of the type of acid used. The release was, however, greatest when nitric acid was used as the titrant. In the nonconservative system, FBA showed its highest calcium release at pH 11 after the addition of 400 ml of pH=1 H_2SO_4. Comparatively greater amount of calcium was released from fly ash under nonconservative conditions than under conservative conditions.

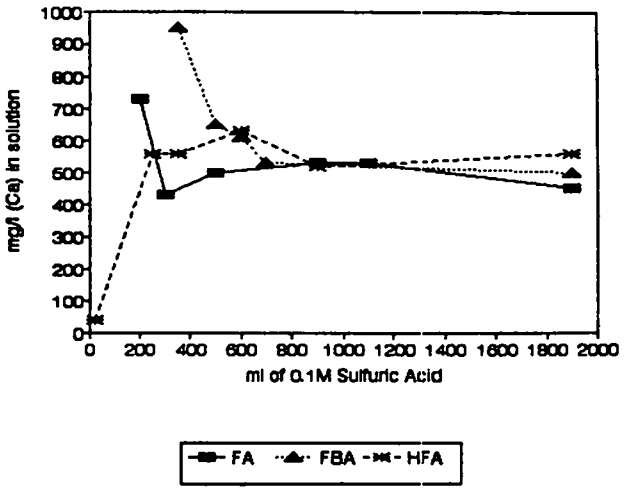

Figure 6 Calcium release from 10 g FA, FBA, and HFA under conservative system using 0.1M sulfuric acid.

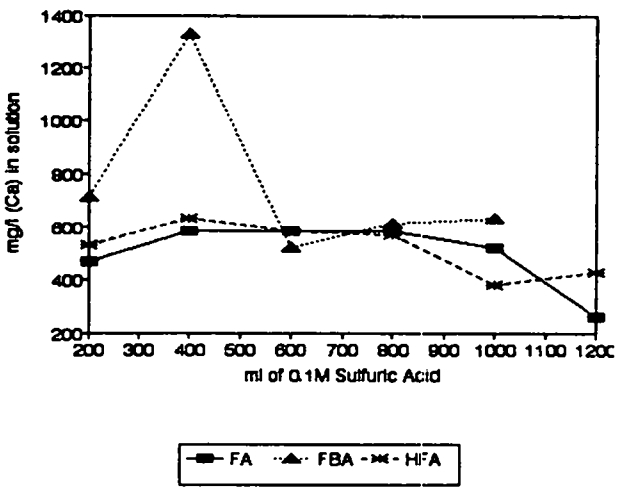

Figure 7 Calcium release from 10 g FA, FBA, and HFA under nonconservative system using 0.1M sulfuric acid.

The release of these metals was also monitored during the titration of fly ash with DDI water. The release of other metals, besides calcium, was not significant and did not follow any distinguishable pattern. This could be interpreted

to mean that no true desorption had occurred, but only a random release of cations that were loosely bonded to the surface of fly ash was apparent. FBA released the same amount of calcium in DDI water as it did when it was titrated with nitric or sulfuric acid solution of pH 1. This fact could lead to the conclusion that the calcium release is more dependent on the calcium concentration in solution than the pH of the solution. This observation is in agreement with Fruchter et al. (1990).

The solubility of the metals studied in this paper have been found (Fruchter et al., 1990) to be controlled by either sulfate or hydroxide compounds. Qualitatively, for metals such as calcium which their solubility is basically controlled by the sulfate compound should not be an issue for the systems where sulfuric acid is added. High concentrations of sulfate anions are introduced in the solution in the form of sulfuric acid and should lead to the precipitation of the compounds instead of their dissolution. The presence of calcium and other metals in the solution should be explained by other mechanisms, such as desorption or ion exchange. The presence of aluminum could be attributed in dissolution since its solubility is more dependent on the hydroxide compound. Iron solubility is also dependent on the hydroxide compound however the concentrations found in solution in the present study were much above those that would be caused by dissolution mechanism only. Considering these observations the mechanism of desorption is the predominant cause for the release of metals in the fly ash aqueous solutions.

Conclusions

The ash products used in this study were proved to be excellent buffering materials. Fluidized bed ash (FBA) showed a stronger buffering capacity than FA or HFA by releasing large quantities of alkaline material to the acidic titrant. As a result, FBA-treated acid would be expected to have a higher pH when compared with FA-treated acid with equal amount of ash added. For highly acidic solutions, FBA released all its buffering materials and kept the pH at very high levels until the buffering material was exhausted. HFA was not a good buffering material since a decrease in pH directly corresponded to increased addition of the acidic solution. These conclusions were made based on the ash products that are described in the material and methods section. Calcium and sodium were the metals most released at high pH whereas aluminum, iron, and chromium were released at low pH.

Acknowledgements

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ADSORPTION OF ACID MINE DRAINAGE METALS ON FLY ASH[1]

by

Hrissi K. Karapanagioti[2] and Asmare Atalay[3]

Abstract. Fly ash (FA), fluidized bed ash (FBA), and hydrated fly ash (HFA) were assessed for their sorptive properties of metals from acid mine drainage (AMD). Metals in sulfuric acid solutions at different pH conditions were equilibrated with fly ash for varied detention time. Preferential adsorption studies were conducted with respect to six major metals that were found in AMD: iron, manganese, zinc, aluminum, magnesium, and calcium. Comparison of the three fly ashes indicated that all had excellent sorptive properties. However, in the case of the most acidic solution (pH=1), no significant adsorption occurred. After the solution had dropped below a certain pH, adsorption decreased dramatically and release of metals into solution began. Among the six metals, iron seemed to be preferentially adsorbed on fly ash; the adsorption of calcium was not observed. Results indicated that pH was a very significant factor for the adsorption of metals on the fly ash surface. Data obtained from the use of real AMD samples for fly ash treatment were in agreement with experimental observations. Comparison of fly ash with two clays for sorptive properties revealed that fly ash was the best alternative for treatment of acid mine drainage. Treatment of AMD with clay resulted in solutions of low pH and high concentrations of cations in solution.

Additional Key Words: Adsorption, Fly ash, Metals, Acid mine drainage.

Introduction

Acid mine drainage (AMD) can be defined as the pollution resulting from mining activities involving exposure of pyrite to water and oxygen. It is formed primarily by chemical and biological oxidation of pyrite. The main characteristics of AMD are low pH and elevated concentrations of metals. AMD from abandoned mines is a non-point source pollution, thus it contributes greatly to nearby water quality degradation. Although several methods have been developed for AMD treatment, more research, which considers cost and environmental impact, is needed for achieving optimum solution. Perhaps one cost effective solution is treatment of AMD with fly ash. Fly ash is a fine particle left during coal combustion; it is recovered by air pollution control equipment, and collected in hoppers. The main hypothesis of this study is that fly ash surface is negatively charged, and will remain that way after its contact with acid mine drainage. To test this hypothesis three different types of fly ash with varying physical and chemical properties were evaluated. The high surface area of fly ash and its negative surface charge should induce cation sorption from AMD.

Preliminary investigation has shown that fly ash is capable of

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neutralizing AMD and adsorbing the metals present in this effluent (Atalay, 1992; Moheeth, 1993; Jackson, 1993). Fly ash has also been used in several adsorption studies involving cations; and many of them reported positive results. The only cation that fly ash was reportedly unable to adsorb efficiently was mercury (Gangoli et al., 1975). Moreover, the extent of adsorption of metal ions on the fly ash surface was substantially reduced when the pH of the final solution was in the acid range (Gangoli et al., 1975). The alkaline nature of fly ash is a major contributing factor in the adsorption of metal ions. For lead, the percentage adsorption on fly ash surface increased in the range between pH 3 to 5. The removal of cadmium by adsorption on fly ash was found to increase with increased pH values from 2 to 8, and was maximum between 7 and 8 (Viraraghavan and Rao, 1991). For zinc, the maximum removal was noted at pH 7.5. Several authors (Singh et al., 1991; Mathur and Rupainwar, 1988; Sen et al., 1987; Panday et al., 1985) have reported the pH at which maximum adsorption of selected metals on fly ash surfaces occurs and the oxides responsible for the adsorption of certain ions. Decreasing trend in adsorption was observed for fly ash at very high pH values where the cations were involved in the formation of soluble hydroxy complexes (Yadava et al., 1987).

The alumina content in fly ash is primarily responsible for the adsorption of anions (Panday et al., 1984; Diamadopoulos et al., 1993; Gangoli et al., 1975). A study in which chemical oxygen demand (COD) removal was attempted with adsorption on fly ash also gave positive results (Pankajavalli et al., 1987). At pH > 2.5 the fly ash surface was found not to favor anion adsorption. The adsorption of anions on fly ash is thought to be governed by diffusion followed by surface compound formation. Viraraghavan and Rao (1991) specified the above theory when they were studying the bonding phenomenon between the alumina sites on fly ash and chromate anion.

Studies of Cr(VI) adsorption on fly ash surface (Panday et al., 1984) showed maximum adsorption at pH 2. The desorption studies for the same element concluded that the weakly bonded adsorbate ions were removed at acid to neutral pH. In alkaline solutions, the Cr(VI) anion was totally desorbed due to the formation of soluble compounds. Diamadopoulos et al. (1993) observed that the maximum adsorption of As(V) on the fly ash surface occurred at pH 4. The arsenic used in their study was in the form of arsenate, which indicated anionic chemistry.

Materials and Methods

Sample Collection

Both fly ash (FA) and hydrated fly ash (HFA) samples were shipped in plastic containers from Oklahoma Gas and Electric Plant, Muskogee, OK. Fluidized bed ash (FBA) samples were generated at the Shady Point Power Plant, Latimer County, OK and shipped to us in plastic containers by Brazil Creek Minerals, Inc., Fort Smith, AR. Acid mine drainage samples (AMD1) and (AMD2) were collected from abandoned mines, "Mine No 7", at Pittsburgh, OK, and "Red Oak Mine" at Latimer, OK, respectively. The bentonite clay was purchased from Central Bag Company, Kansas City, MO. The kaolinite clay was purchased from The Feldspar Corporation, Edgar, FL.

Sample Preparation

Two sulfuric acid solutions were prepared, one at pH=1 and another at pH=4. The following concentrations of metals were dissolved into the above acid solutions: 500 mg/l Fe, 22 mg/l Mn, 5 mg/l Zn, 400 mg/l Ca, 400 mg/l Mg, and 100 mg/l Al. These concentrations reflected the levels of metal present in the actual AMD1 sample. The ratio between the adsorbent and solution was 2 g of fly ash per 100 ml of synthetic AMD. Chaluvadi (1991) and Moheeth (1993) proposed a ratio of 20 g of fly ash per liter of solution. The order with which the six metals were adsorbed on the fly ash surface under the same conditions was determined through preferential adsorption studies. Equal concentrations of each metal, ranging from 1 mg/l to 5 mg/l, were used in solutions of the same ratio as above: 2 g of fly ash and 100 ml of solution. For the screening test,

Table 1: Experimental variables used for the adsorption and neutralization study.

Property Tested	Control Condition	Materials Tested	Acid Solution Used
Adsorption	Conservative[4]	FA, FBA, HFA	Mineral plus Acid Solutions of pH 1 and 4
	Nonconservative[5]	FA, FBA, HFA Kaolinite, Na-Bentonite	Mineral plus Acid Solutions of pH 1 and 4
Adsorption and Desorption Tests	Nonconservative	FA, FBA, HFA Kaolinite, Na-Bentonite	4 Different Mineral plus Acid Solutions, Iron plus Acid Solution
Preferential Adsorption	Nonconservative	FA, FBA, HFA Kaolinite, Na-Bentonite	1, 2, 3, 4, 5 mg/l solutions
Screening test (Neutralization capacity and adsorption ability)	Nonconservative	FA, FBA, HFA Kaolinite, Na-Bentonite	AMD2 sample

actual AMD samples were used. Table 1 presents the different experimental variables used in testing cation adsorption on fly ash.

All of the above tests were performed under two controlled conditions: conservative and nonconservative. According to Corbitt (1990) receiving waters can be classified as conservative or nonconservative systems depending on their confined or unconfined nature. For example, reservoirs or lakes are considered conservative systems whereas rivers or estuaries are nonconservative systems. With this analogy in place, the first controlled condition (Table 1) was designed to simulate a conservative system that holds most of the incoming pollutants. Similarly the second controlled condition simulated a nonconservative system with shorter residence times for the pollutants than the first. In order to simulate a conservative system, the solution added to each sample was shaken in constant speed undisturbed until the solution reached a stable pH value. This stage was defined to simulate equilibrium condition for the conservative system. Equilibrium in this case was reached in an average of seven days. The nonconservative system was simulated with repeated additions and removals of the same amount of solution at constant time intervals. The time intervals for this case were two days each. All experiments were conducted at room temperature.

After the measured amounts of synthetic acid mine drainage and the fly ash samples had been shaken together in high density polyethylene Erlenmeyer flasks, the solid part of the sample was filtered with Whatman No. 2 filter paper from the solution. The solution was digested with nitric acid using a Tecator Digestion Apparatus. Each digested sample was

[4]Systems with long residence time that reach stable pH values.

[5]Systems with short residence times which do not reach stable pH values.

diluted to 100 ml with double deionized water containing 0.2 % lanthanum oxide. The same digestion method was used to analyze raw samples of fly ash. In the case of solid samples filtration was used to separate the digested solid sample from the leachate. The diluted samples were stored in high-density polyethylene bottles until they were analyzed for metals.

Sample Analysis

Digested samples were analyzed for metals using Buck Scientific Atomic Absorption Spectrophotometer VGP System Model 210. High concentration standard metals (1000 mg/l) were purchased from Fisher Scientific and used to prepare diluted standards for calibration. Nitrous oxide/acetylene flame was used to measure aluminum concentration. All other metals were analyzed with air/acetylene flame. Each metal was analyzed using a different wavelength as suggested by Welz (1985).

Results and Discussion

Adsorption of Cations at Varied pH

Table 2 presents the metal concentrations in fly ash and AMD samples. It is apparent from the data that all three ashes contain high levels of calcium, which came from the coal combustion process. In particular, the calcium content in FBA is more than in FA or HFA. This has a lot to do with the uniqueness of the fluidized bed combustion process which operates at a lower temperature and utilizes lime to strip out sulfur from the coal during combustion. In addition, aluminum, iron, and magnesium are found at elevated levels in all three ashes. The two acid mine drainage samples show different metal compositions. AMD1 contains higher levels of iron, magnesium, and calcium than AMD2. The greatest environmental concern from AMD is the high content of iron, manganese, and perhaps aluminum. The concentration of metals in AMD will vary with rainfall events and seepage.

Table 3 indicates that for the conservative system, where the less acidic, pH=4 solution was used as the titrant, total metal removal was observed in the system except for calcium and magnesium. However, after a certain volume of the titrant was added, some of the cations began to elute into the solution. The concentration of metals observed after the release was higher than those added, which meant that some of the cations originally present in fly ash were also released. This phenomenon was observed mainly in solutions of pH=1. For this reason, adsorption data which resulted from this solution are not presented in graphical forms.

Figure 1 presents the adsorption pattern for iron in the conservative systems. FBA is the fly ash that adsorbs iron totally on its surface after the 600 ml of synthetic AMD have been added to 10 g of fly ash. FA and HFA also adsorb iron for a range of volumes of synthetic AMD added.

Figure 2 also presents the competitive adsorption of iron on fly ash, but for a nonconservative system. Comparison of iron adsorption under the two systems revealed unique differences. Under the conservative system (Figure 1) each ash adsorbed the iron completely when 400, 500, and 600 ml of the acid were added. Consequently, the concentration of iron in solution decreased from 500 mg/l to 0 mg/l. Increased addition of the acid solution, up to 1100 ml, still showed total adsorption of iron. However, increasing the volume of added solution beyond 1200 ml resulted in decreased adsorption of iron from FA and HFA, but not from FBA. Figure 2 showed that total adsorption occurred when 200 ml of the solution was added as opposed to 400 ml to 600 ml in the conservative system.

However, beyond the addition of 400 ml of the solution, both FA and HFA decreased in their adsorption. This is in contrast to the 1100 ml needed to effect iron adsorption in the conservative system. Moreover, the decrease in adsorption was more gradual in the nonconservative system than in the conservative system. FBA exhibited similar behavior in both systems, which was total adsorption of iron throughout the addition of the pH 4 mineral plus sulfuric acid solution. This observed high adsorption capacity of FBA for iron could be beneficial in the use of this ash for treatment of acid mine drainage.

Table 2: Concentrations of metals in fly ash and AMD samples.

Sample	FA	FBA	HFA	AMD1	AMD2
pH	11.9	12.4	12.0	4.5	4.5
Metals	mg/kg			mg/l	
Mn	170	510	270	23	6.8
Fe	35,000	55,000	28,000	600	185
Zn	180	250	190	7.5	3.8
Al	51,500	12,500	44,000	15	8
Mg	40,000	17,500	47,500	595	55.8
Ca	205,000	295,000	190,000	455	75
Na	7,500	1,000	17,800	-	-
K	2,850	3,250	3,000	35	4.1
Cu	2,200	35	185	0.06	-
Cr	65	50	75	1.8	-
Ni	110	70	100	0.45	-
Pb	68	85	170	0.73	-
Cd	< 2	< 2	< 2	< 0.01	-
Ag	< 30	< 30	< 30	< 0.6	-

For the nonconservative system (Table 3), using the same titrant of pH 4, adsorption of cations except calcium occurred throughout the additions. Initially the metals were removed in large quantities, then the amount removed gradually decreased. At the final stage, the concentration of metals in solution reached the initial concentration and in some cases it was even higher, perhaps due to release of metals initially present in the fly ash.

Table 4 shows that for the conservative system, when the most acidic titrant of pH=1 was used, no discernible adsorption occurred. Actually, after a certain pH, release of metals into solution occurred. However, the amount released from the fly ash as compared to the concentration initially present in the solution was not significant. When the volume of solution that was added was low, the concentration was high. As the volume added was increased, the concentration of metals in the solution decreased, approaching the initial concentration.

For the nonconservative system using the same titrant at pH 1 (Table 4), adsorption occurred only during the first couple of additions of mineral solutions. During successive additions of a mineral solution at this low pH, the quantity of metals released into the solution was the sum of all metals added with the mineral solution and that which was released from the fly ash. Adsorption virtually ceased when release of metals began. Calcium was the cation that was not adsorbed on any of the adsorbents tested, but it was continuously released into the solution. This is due to the fact that the fly ash used contained lime in amounts of 15 to 20 percent by weight. In the nonconservative systems, for both titrants (see Tables 3 and 4) fly ash seems to have a higher capacity in holding metals adsorbed onto its surface than in conservative systems while there is a

Table 3: Metal adsorption from 10^{-4} M metal plus sulfuric acid solution of pH 4 treated with fly ash.

Metal (Initial Concentration mg/l)	Fly Ash	Conservative (for 10 g of fly ash)			Nonconservative (for 10 g of fly ash and 200 ml/addition)		
		ml added while adsorption lasted	pH range for 100 % adsorption	max mg/kg adsorbed	addition of last adsorption	pH while adsorption lasted	max mg/kg adsorbed
Mn (22)	FA	1900	12-8	2200	4	4.5	12800
	FBA	>2000	12-< 8	>3800	8	6	31400
	HFA	>2000	12- 8	2200	5	6	12800
Fe (500)	FA	>2000	12- 8	55000	8	4.5	51200
	FBA	>2000	12-< 8	>95000	>9	<6	88000
	HFA	>2000	12- 8	55000	>8	<5.5	47800
Zn (5)	FA	>2000	12- 8	275	>9	<4	422
	FBA	>2000	12-< 8	475	7	7.5	410
	HFA	>2000	12- 7.5	855	>8	<5.5	418
Al (100)	FA	>2000	12-<4.5	19000	>9	<4	15800
	FBA	>2000	12- 11	7600	>9	<6	15340
	HFA	>2000	12- 8.5	9500	>8	<5.5	14920
Mg (400)	FA	1700	12- 10	20000	2	7	60
	FBA	>2000	12-10.5	36000	6	8.5	41000
	HFA	950	12- 9	24000	2	8.5	9400
Ca (400)	FA	0	-	0	0	7	800
	FBA	0	-	0	0	12	0
	HFA	0	-	0	1	8.5	400

decreased tendency in adsorbing magnesium .

The adsorption studies were based on the worse case scenario of AMD, i.e., the use of a sulfuric acid solution of pH 1 and high concentration of metals. Figure 3 presents the adsorption of iron on all three ashes tested; FBA1 and FBA2 indicate the use of this ash with synthetic AMD and iron solution, respectively. The results are in agreement with the discussion presented for Figures 1 and 2 in that FBA is the ash which adsorbed the most iron from the solution. It is apparent that FA and HFA are comparatively poor adsorbents of iron. However, both FBA1 and FBA2 were able to adsorb between 75 and 95 percent of the added iron. This observation was consistent within the range of 140 to 700 mg/l Fe addition. When the concentration of iron in the solution increased to 840 mg/l, the amount adsorbed by FBA1 and FBA2 decreased by 30 and 50 percent (from their highest adsorption), respectively. This implies that the available adsorption

site on the surface of FBA had been exhausted. The adsorption study for iron as a single component in solution provided a unique information. The comparison did not show any obvious

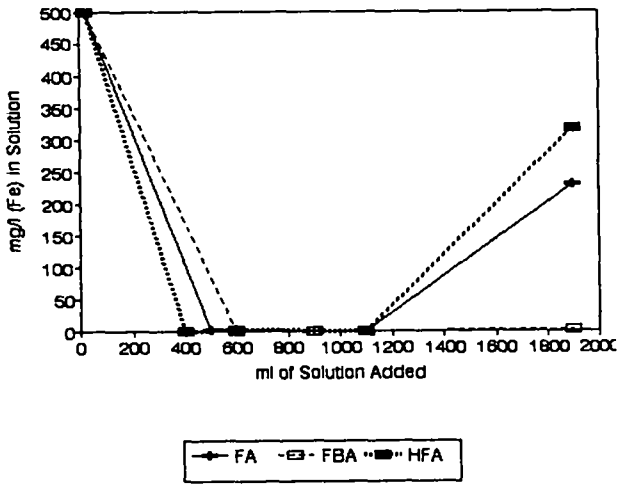

Figure 1: Iron adsorption on fly ash from sulfuric acid and metals solution in conservative systems at pH 4.

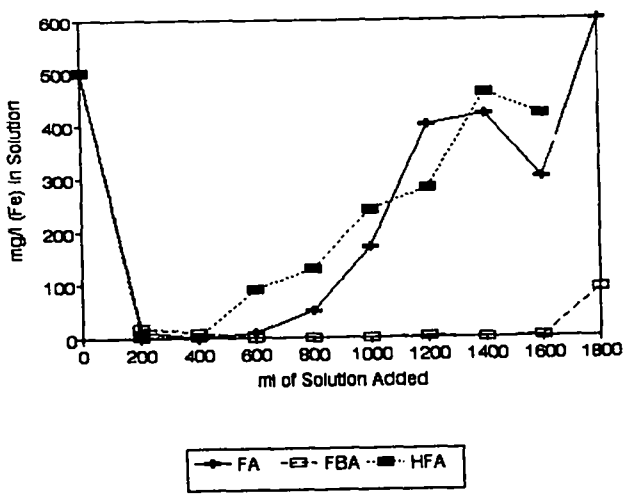

Figure 2: Iron adsorption from pH 4 sulfuric acid and metals solution on fly ash surface in nonconservative systems.

differences in adsorption patterns. This information can lead to the conclusion that iron is adsorbed on specific adsorption sites where the presence of the other cations is not preferred.

This study did not result in any adsorption or desorption constants because the concentrations were not chosen to produce isotherms. Rather, concentrations were chosen to simulate AMD and to study the patterns of adsorption of cations present in multi-component solutions.

Preferential Adsorption

Preferential adsorption was studied by preparing equal concentrations (1-5 mg/l) of metals in solution and equilibrating them with fly ash. This test was different from any of the above because the initial concentration of metals in solution was much lower than those used in the above studies; and the initial concentration was equal for all six metals (iron, manganese, zinc, aluminum, magnesium, and calcium).

Figure 4 presents the removal of metals using fly ash (FA) during the preferential adsorption tests. Iron, manganese, and magnesium are totally adsorbed by all three types of fly ash. Zinc was partially adsorbed but the adsorption did not show a clear pattern. Calcium and aluminum were released from fly ash in a consistent manner. The presence of varied concentrations of calcium and aluminum in the initial solution did not affect their release from fly ash. Calcium was released in such a manner that its concentration was constant in all samples regardless of its initial concentration in the solution.

Theoretical Adsorption Mechanism

Experimental results showed fly ash to be a suitable adsorbent for the removal of metals from AMD. Surfaces which contain proportions of layer silicates, oxides of iron, aluminum, and manganese are known to have a wide range of trace metal sorption characteristics. Fly ash has such surfaces, moreover, the porous nature of fly ash provides an opportunity for intraparticle transport from bulk to solid-solution interface. The high surface area of fly ash makes many sites available for adsorption interactions.

The theory of metal adsorption on fly ash surface is based on the surface charge. For different pH values, oxides at the surface develop

Table 4: Metal adsorption from 0.1 M metal plus sulfuric acid solution of pH 1 titrated with fly ash.

Metal (Initial Concentration mg/l)	Fly Ash	Conservative (10 g of fly ash)		Nonconservative (for 10 g of fly ash and 200 ml/addition)		
		ml added while adsorption lasted	max mg/kg adsorbed	addition where last adsorption occurs	pH while adsorption lasted	max mg/kg adsorbed
Mn (22)	FA	20	44	1	5	440
	FBA	20	44	2	3	860
	HFA	200	440	1	5	440
Fe (500)	FA	600	30000	1	2	10000
	FBA	200	10000	2	2.5	20000
	HFA	500	10000	1	2	12000
Zn (5)	FA	20	10	1	4	40
	FBA	200	100	2	1.8	180
	HFA	300	100	2	2	160
Al (100)	FA	200	200	2	3	4000
	FBA	20	200	0	2.5	0
	HFA	300	2000	2	3	4000
Mg (400)	FA	20	800	0	-	0
	FBA	20	800	1	-	800
	HFA	20	800	0	-	0
Ca (400)	FA	0	0	0	1.8	0
	FBA	0	0	0	12	0
	HFA	0	0	0	2.5	0

different charges. The pH of the system determines whether cationic or anionic adsorption will occur.

In an acidic medium, a positive charge is developed on the oxides of fly ash surfaces. This is because the hydrogen ions are in excess, and they bond with the oxygen on the oxide surface. The bonding that occurs can be described by the following equation:

$$M\text{-}OH + (H^+) \rightarrow M\text{-}OH_2^+ \qquad (1)$$

The resultant positive charge does not favor the adsorption of positively charged metals on the fly ash surface. In an alkaline medium the oxide surface develops a negative charge. This is because the hydroxide ions are in higher concentration around the fly ash surface. In this case, the hydroxyl ion has little or no affinity to the oxide surface. However, it serves as a titrator to remove the positive charges from the surface. The reaction can be described by the following equation:

$$M\text{-}OH + (OH^-) \rightarrow M\text{-}O^- + H_2O \qquad (2)$$

This negatively charged surface favors the adsorption of positively charged metals on the surface. The following exchange reaction describes how the adsorption could take place:

$$M\text{-}O^- + X^+ \;\text{-->}\; M\text{-}O\text{-}X \qquad (3)$$

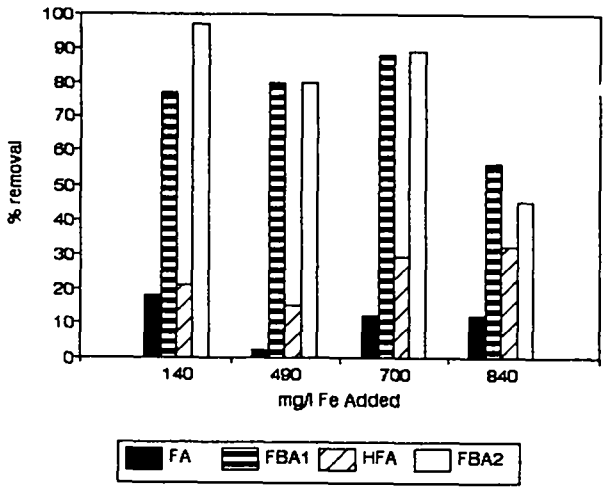

Figure 3: Results for iron adsorption tests using 2 g fly ash samples with 100 ml of metal solutions (Indicators 1 and 2 for FBA samples refer to : 1 mixing with synthetic AMD and 2 mixing with solutions containing only Fe).

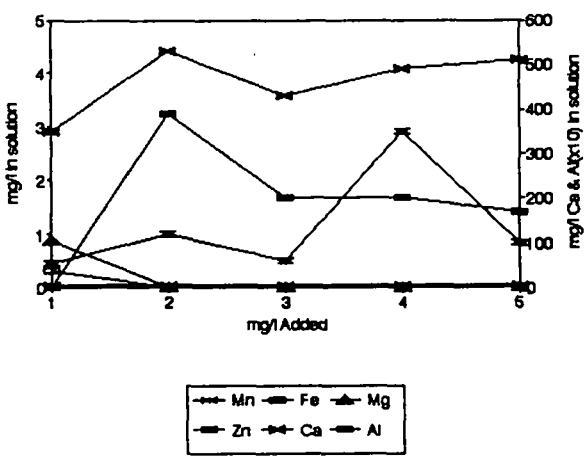

Figure 4: Preferential adsorption of 2 g fly ash samples with 100 ml of meatal solution containing equal concentrations of selected metals ranging from 1 to 5 mg/l (Mn, Fe, Mg, Zn, Ca, and Al).

However, there are cases where adsorption takes place at low pH. This occurs when the surface is neutral and the surface oxygen is bonded with one hydrogen. At relatively low pH, the metal replaces the hydrogen ion on the surface oxide. The exchange reaction that occurs can be described as follows:

$$M\text{-}O\text{-}H + X^+ \;\text{-->}\; M\text{-}O\text{-}X + H^+ \qquad (4)$$

Theis and Wirth, (1977) have discussed in detail the sorptive behavior of trace metals on fly ash. Gangoli et al. (1975) have concurred with the same theory for the removal of heavy metals. Later, many authors (Panday et al., 1984; 1985; Yadava et al., 1987; Singh et al., 1991; Zachara et al., 1992; etc.) used this theory to discuss adsorption on various surfaces.

Using MINTEQA2 to predict precipitation

The geochemical equilibrium model (MINTEQA2) was used to investigate the probability of precipitation of minerals during the sorption process of metals on fly ash surfaces. The input files simulate the metal solutions without the addition of fly ash. However, the pH was set at the same value as that obtained after the addition of fly ash.

Concentrations that were used to prepare the metal solutions were used for a sweep in the MINTEQA2 input file from pH 1 to 12. Results indicated that aluminum precipitated totally at pH 5. Iron and zinc precipitate at pH 8, and magnesium and manganese at pH 10. From tests on preferential adsorption that reached pH values above 12, it could be assumed that aluminum dissolves partially at pH above 12.

Considering the above results from MINTEQA2, and comparing them with the data obtained from the lab, metal removal that occurred at pH below 8 was due to adsorption from solution except for aluminum. At higher pH values, adsorption has to compete with precipitation and the removal could be due to both processes.

Removal of Metals from AMD Using Fly Ash and Clay

Table 5: pH and percent removal after 2 g of FA, FBA, HFA, Kaolinite (Kao), Na-Bentonite (NaB) have been titrated with 100 ml of AMD. Negative values indicate release of metals compared to initial concentrations.

Case I: Acidified AMD, pH < 2.

Material	Final pH	Fe (190 mg/l)	Mn (7 mg/l)	Zn (4.2 mg/l)	Ca (70 mg/l)	Mg (56.5 mg/l)	Al (8 mg/l)
FA	5.7	95.8	70.7	9.5	-1571	-128	62.5
FBA	12.2	100	99.4	54.8	-3100	100	0
HFA	7.7	100	85.1	47.6	-1714	-261	25
Kao	1.3	-8.4	13.7	23.8	-228.6	2.6	75
NaB	2.9	36.8	-94.9	26.2	-1000	-34.5	-162

Case II: Natural AMD, pH = 4.5.

Material	Final pH	Fe (180 mg/l)	Mn (6.5 mg/l)	Zn (2.5 mg/l)	Ca (80 mg/l)	Mg (55 mg/l)	Al (8 mg/l)
FA	11.1	99.7	100	52	-437.5	100	-400
FBA	12.3	100	99.4	40	-2025	100	-100
HFA	10.6	100	98.6	-460	-562.5	100	100
Kao	2.4	82.1	5.5	-40	-200	5.5	0
NaB	6.4	69.6	74.2	-220	-212.5	63.6	-62

The titration of fly ash with real AMD sample resulted in values that were expected. Table 5 presents the pH and percent removal of metals in AMD. The data in Case I presents AMD2 samples which were further acidified to meet the worst case scenario (i.e., pH<2). The data in Case II was obtained from experiments that utilized actual AMD2 samples obtained from the field.

All fly ash samples were able to remove metals present in both acidified and unacidified AMD2 samples effectively. Iron and manganese were totally removed from both solutions, whereas zinc was partially removed. The release of basic cations (calcium and magnesium) from fly ash was also assayed during the titration process. It was found that magnesium was not released from fly ash when the titrant was the natural AMD2 sample, but it was released in high amounts when the pH of the natural AMD2 was decreased successively (Table 5). In both situations, FBA showed the same pH and metal removal capacity. However, when the pH of AMD2 was low, calcium release was also low.

In order to compare the sorption characteristics of fly ash to known surfacial materials of similar sorptive behaviors, two clays (kaolinite and sodium bentonite) were evaluated. Similar experiments conducted showed that kaolinite adsorbed some iron, zinc, and manganese. However, only manganese was adsorbed by kaolinite during the preferential adsorption studies, but the adsorption was only 10 % of the manganese added. In the case where kaolinite was titrated with the natural AMD sample, it adsorbed more iron than manganese (see Table 5). When it was titrated with the acidified AMD (pH<2), kaolinite adsorbed manganese, zinc, and aluminum at higher levels than when the pH was 4.5 (natural AMD).

Na-bentonite adsorbed only zinc (from 25 to 50%) during the adsorption and preferential adsorption tests. During titration with the acidified AMD sample, Na-bentonite adsorbed some iron and zinc (see Table 5). However, during titration with the natural AMD, Na-bentonite adsorbed some iron, manganese, and magnesium. Although the pH of Na-bentonite solution was higher (pH=6.4) than the pH of the FA solution (pH=5.7), the percent adsorption that resulted for Na-bentonite was lower than that for FA.

Conclusions

This study evaluated the adsorption capacities of the three forms of fly ash. Both FA and HFA showed significant metal removal potential from natural AMD and prepared solutions. In general, FBA exhibited superior quality for metal removal. It should be noted that adsorption is a pH dependant phenomena; therefore for different pH values the same fly ash could exhibit different behavior. FA seems to adsorb cations at higher pH values. In contrast, at low pH values FA not only desorbs the cations which it adsorbed from AMD solutions, but also releases cations initially present on its surface. Preferential adsorption tests showed that iron, magnesium, zinc, and manganese were the most adsorbed cations, whereas calcium and aluminum were released from the fly ash surface. A comparison study showed that fly ash was a better metal adsorber than clay.

Acknowledgements

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EQUILBRIUM MODELING OF TRACE METAL TRANSPORT FROM DULUTH COMPLEX ROCKPILE[1]

by

Paula D. Kelsey[2], Ronald W. Klusman[2] and Kim Lapakko[3]

Abstract: Geochemical modeling was used to predict weathering processes and the formation of trace metal-adsorbing secondary phases in a waste rock stockpile containing Cu-Ni ore mined from the Duluth Complex, MN. Amorphous ferric hydroxide was identified as a secondary phase within the pile, from observation and geochemical modeling of the weathering process. Due to the high content of cobalt, copper, nickel, and zinc in the primary minerals of the waste rock and in the effluent, it was hypothesized that the predicted and observed precipitant ferric hydroxide would adsorb small quantities of these trace metals. This was verified using sequential extractions and simulated using adsorption geochemical modeling. It was concluded that the trace metals were adsorbed in small quantities, and adsorption onto the amorphous ferric hydroxide was in decreasing order of Cu > Ni > Zn > Co. The low degree of adsorption was due to low pH water and competition for adsorption sites with other ions in solution.

Additional Key Words: geochemical modeling, sequential extractions.

Introduction

The Duluth Complex

The Duluth Complex is a layered igneous intrusion in northeastern Minnesota which is well known as a potential future source of copper and nickel ore (Johnson, 1970 and Sims, 1967). It also contains substantial amounts of Co and platinum-group metals (Boucher, 1975). In the early 1970's AMAX Exploration Inc. began evaluating the potential of an area in the vicinity of the town of Babbitt, MN for the mining and extraction of copper and nickel (Lapakko, 1994). In 1977, a joint effort between the State of Minnesota and AMAX produced six stockpiles containing lean ore material from a shaft dug in 1975. The piles were constructed in order to evaluate what drainage problems may occur if Cu-Ni mining were to be developed in Minnesota.

The construction of pile 1 was completed April 20, 1977. Effluent from the base of the pile was monitored from 1978 to the present by the Minnesota Department of Natural Resources (MNDNR), Division of Minerals. Stockpile samples, effluent and mineralogical data, all from pile 1, were provided by Kim Lapakko, Principal Engineer, Minnesota Department of Natural Resources, Division of Minerals.

Site Description and Geology

Geology of the Babbitt region has been described by Bonnichsen (1972) as being part of the Upper Precambrian Keweenawan formation, comprised of troctolitic gabbro, which consists mainly of calcic plagioclase and olivine. The sulfide minerals in this area generally decrease in concentration from pyrrhotite ($Fe_{1-x}S$), chalcopyrite ($CuFeS_2$) to pentlandite [$(Fe,Ni)_9S_8$]. A grab sample taken from pile 1 was subjected to a mineralogical analysis. The silicate minerals identified from the grab sample, in volume percentages, were; plagioclase (59%), clinopyroxene (11%), olivine (11%), orthopyroxene (3.7%), and amphibole (3.6%). The same grab sample also contained sulfide minerals, in volume percentages, pyrrhotite (0.84%), chalcopyrite-cubanite (0.77%) and pentlandite (0.037%) (Stevenson et al., 1979).

Table 1 contains effluent data from stockpile no. 1 from 10/16/89 to 11/07/91. From the data it is clear that the effluent is continually evolving and has

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yet to reach steady state equilibrium with infiltrating meteoric water. Over a period of roughly two years, the pH has dropped from 5.88 to 5.24 (s.u.). Sulfate concentrations have increased almost two-fold, a result highly dependent on fluctuations in transport flow. However, according to Lapakko (1994), the annual sulfate release rate has remained fairly steady, increasing only slightly from 1978-1991, indicating the oxidation of sulfide minerals within the pile. Concentrations of trace metals Co, Cu, Ni and Zn have also increased with time, a result of the decreasing effluent pH over time. Species such as Fe, Mn and Cl were measured in 1981-82 to be roughly 0.02-0.06 mg/L, 0.05-0.13 mg/L and 5 mg/L, respectively. Current data reveals that both Cl and Fe concentrations have remained steady while Mn has increased with time to roughly 5 mg/L. The low content of iron in the effluent indicates that the formation of $Fe(OH)_3$ within the pile appears to be a possibility, for the appearance of ferric hydroxide under oxidizing conditions occurs at a pH greater than approximately 4.0 (s.u.).

The stockpile is approximately 4 m high and 15 m by 25 m at the base. It was built in four lifts, with lift 1 located at the top of the pile and lift 4 located at the bottom. Samples are identified with three numbers, which represent the pile number, the lift from which it came, and a third number used to distinguish samples from the same lift. An example is 1.3.2, this sample came from pile no. 1, lift 3. Noticeable oxidation was observed in stockpile samples taken from lifts 1 and 2. Visible observations show the samples have a reddish tint to them, while samples taken from lifts 3 and 4 were a medium color gray typical of unoxidized rock. It is hypothesized that the red color was most likely a result of pyrrhotite oxidation, with subsequent precipitation of ferric hydroxide.

Oxidation of Pyrrhotite

Wastes from non-ferrous metal mining operations often contain iron sulfide minerals, made up mostly of mixtures of pyrite (FeS_2) and pyrrhotite ($Fe_{1-x}S$). Pyrrhotite is particularly important in wastes

Table 1: Measured effluent concentrations from stockpile no. 1 over a two year period.

Date	pH (s.u.)	SO4 (mg/L)	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	Cu (mg/L)	Ni (mg/L)	Co (mg/L)	Zn (mg/L)
10/16/89*	5.88	800	212	54	22	5.90	25	1.90	1.0
11/6/89	5.5	680	166	50	15	5.3	25	2.6	1.0
3/28/90	5.05	290	64	21.4	7.4	10.2	19	10.2	1.01
4/12/90*	5.82	685	148	42	10	9.43	32	1.54	1.54
5/17/90	5.72	1250	286	84	20	10	39	1.91	1.78
6/18/90	5.85	1040	202	70	22	7.78	30	1.27	1.14
7/16/90*	5.79	975	214	66	20	7.82	27	1.25	1.19
8/13/90	5.77	920	228	66	22	5.21	21	1.01	0.93
9/10/90	5.42	940	212	74	18	20	56	2.82	2.27
10/15/90	5.29	1850	358	134	30	35	93	4.35	3.1
11/16/90*	5.31	2400	680	202	54	29	88	4.04	3.55
4/22/91	4.96	1030	194	88	16	19.9	68	2.6	1.8
5/20/91*	5.18	1200	252	107	20	20.3	68	2.4	1.5
6/17/91	5.25	1110	214	104	20	27	71	3	1.6
7/15/91	5.27	1260	278	124	26	21	59	2.4	1.5
8/12/91	5.27	1370	318	110	34	18	54	2.3	1.3
9/9/91*	5.22	1420	268	110	28	30	84	3.4	2.0
10/7/91	5.31	1360	272	108	32	21	67	2.7	1.7
11/7/91*	5.24	1270	252	112	30	27	80	3.2	2.1

* Dates used in geochemical modeling runs.

from mafic rocks. When these minerals are exposed to atmospheric oxygen and water, they are oxidized and release acidity to the environment. Sulfide oxidation, namely that of pyrite, has been thoroughly studied as a major cause of acid mine drainage (Nicholson and Scharer, 1992). Pyrite has received much of the attention because it is found to be prevalent in rocks associated with coal deposits and with most metallic and precious mineral deposits. Although pyrrhotite has not been as extensively researched as pyrite as a contributor of acid generation, in many instances it is the only significant gangue sulfide mineral at the site, as is the case in the Duluth Complex.

The oxidation of pyrrhotite in the presence of water and oxygen is as follows; note that the formula for the monoclinic form of pyrrhotite is Fe_7S_8, but it is also found in hexagonal forms (Fe_9S_{10}) within the pile (Lapakko and Antonson, 1993). The overall reaction for the oxidation of pyrrhotite is:

$$Fe_7S_8 + 17.25O_2 + 18.50H_2O = 7Fe(OH)_3\,(s) + 16H^+ + 8SO_4^{2-} \qquad (1)$$

where sixteen acidic protons are released for every mole of pyrrhotite oxidized.

Methods

Initial Inverse Geochemical Modeling

The NETPATH modeling program was used to interpret geochemical mass-balance reactions between an initial and final water along a hydrologic flow path (Plummer et al., 1993). The net geochemical mass-balance reaction consists of the masses (per kilogram of water) of plausible minerals and gases which must enter (dissolve) and/or leave (precipitate) the initial water in order to achieve the final water composition. The data base is slightly limited in the number of minerals that it contains, consequently the model was used only to determine the dissolution/ precipitation of the more common minerals which contained the major constituents found in the aqueous phases. Also, NETPATH does not include some trace metals which were of interest in this study, such as cobalt, nickel and zinc. This limited the usefulness of the NETPATH program, but it still provided information which made the more complex geochemical modeling easier to manage.

Geochemical modeling of the entire complex system was accomplished using the MINTEQAK (Klusman, 1993) model, which was developed from MINTEQA2 (Allison et al., 1991). The code was modified in order to predict the change in composition of an aqueous flow as it traverses a wetland/reactor system for treatment of mine drainage. In this application it was used to extend the results of NETPATH by incorporating weathering of solid solutions and weathering of trace constituents.

Sequential Extraction Procedure

The following is a description of the extraction procedure used for 1 g (-270 mesh) of each sample from stockpile no. 1. The sequential extraction procedure used was modified from Tessier and others (1979) and Chao and Zhou (1983). Each sample was placed in a polyethylene centrifuge tube (50 mL) with a lid. After an extraction medium was added, the sample was placed in a sonicator (Bransonic 220). When the allotted time for each extraction had passed, the sample was centrifuged (Damon IEC model K at 14,000 rpm) for 20 minutes. The supernatant was decanted into polyethylene bottles and saved for subsequent analysis. The remaining residue was then rinsed with 10 mL deionized water, centrifuged for 20 minutes and the liquid was decanted and discarded. The next extractant medium in the sequence was then added to the residual sample. This process was used for extractions E1-E4. The extractions were analyzed for metals using a Perkin Elmer Optima 3000 ICP-AES.

Extraction 1 (E1). Water Soluble. The original 1 g samples were leached with 10 mL deionized water for 30 minutes.

Extraction 2 (E2). Exchangeable (Tessier et al., 1979).The residue from E1 was extracted at room temperature for 1 hour with 10 mL of a magnesium chloride solution (1 M $MgCl_2$, pH 7).

Extraction 3 (E3). Associated with Iron Oxides (Chao and Zhou, 1983). 20 mL 0.25M $NH_2OH{\cdot}HCl$ in 0.25 M HCl was added to the residue remaining from E2. This was placed in a 50 °C water bath (P/S Thelco model 84) for 30 minutes with intermittent shaking.

Extraction 4 (E4). Sulfides (Tessier et al., 1979). The residue from E3 was extracted with 3 mL of 0.02 M HNO_3 and 5 mL of 30% H_2O_2 adjusted to pH 2 with HNO_3 and heated in a water bath at 85 °C for 2 hours. A second 3-mL aliquot of 30% H_2O_2 was added and heated again at 85 °C for 3 hours. After cooling, 5 mL of 3.2 M NH_4OAc in 20% (v/v) HNO_3 was added, the sample was then diluted to 20 mL and cooled. The addition of NH_4OAc was to prevent adsorption of extracted metals onto the oxidized sediment (Gupta

and Chen, 1975).

Extraction 5 (E5). Residual (Briggs, 1994). The final residue was digested using a HCl- HNO_3-$HClO_4$-HF mixture. The samples were quantitatively transferred from the centrifuge tubes to 100 mL Teflon® beakers. To each sample was sequentially added: 15 mL HCl, 10 mL HNO_3, 5 mL $HClO_4$ and 10 mL HF. All acids were in concentrated form. The samples were then heated to dryness overnight at 110 °C. The next day 5 mL $HClO_4$ was added to each sample, the temperature was increased to 150 °C and the samples were again heated to dryness overnight. The following day the samples were cooled and to each sample 5 mL aqua regia was added. The samples should be completely dissolved at this point. The solutions were then transferred into 50 mL volumetric flasks and brought to volume with 1% HNO_3. After thorough mixing, the samples were then transferred to polyethylene bottles and saved for later analysis.

Extraction 6 (E6). Total digestion (Briggs, 1994). This extraction was separate from extractions E1-E5. In this case, 0.200 g of fresh sample was totally digested using the same procedure as outlined for E5. Since only one-fifth of the original sample weight from E1-E5 was used, the volume of all acids added was decreased by one-fifth, but the procedure remained the same.

MINTEQAK Adsorption Modeling

Adsorption of a solute molecule on the surface of a solid can involve removing solvent from the solid surface, then removing the solute molecule from the solution by attaching the solute to the surface of the solid (Stumm and Morgan, 1981). This phenomenon can be described based upon the concept of relative surface excess. For the accumulation of solute *i* at the mineral-water interface, the relative surface excess, Γ_i , or adsorption density of solute *i*, may be defined as:

$$\Gamma_i = n_i / A , \qquad (2)$$

where n_i is the moles of surface excess of solute *i* per unit mass of the mineral phase, and A is the specific surface area of the mineral phase. n_i can be defined by two quantities, strong and weak adsorbing surface sites. Strong surface sites refer to a small set of high affinity cation binding sites and the weak surface sites are the total reactive sites available for sorption of protons, cations and anions.

Adsorption of the four trace metals onto ferric hydroxide was achieved using the triple-layer model available in MINTEQAK. Surface sites are represented as SOH groups where S represents the metals associated with the solid structure, located at the solid-liquid interface. The adsorption of a divalent cation, M^{2+}, onto an adsorption site would be represented by the reaction:

$$SOH + M^{2+} = (SO{\cdot}M)^{+} + H^{-} \qquad (3)$$

then,

$$K_1 = \frac{[SO{\cdot}M^{+}]\,[H^{+}]}{[SOH]\,[M^{2+}]} , \qquad (4)$$

where K_1 is the surface complexation constant.

Results and Discussion

Initial Inverse Geochemical Modeling

Of the twenty possible effluent sampling dates to model, seven dates were chosen in order to model the most varied compositions in the effluent from the pile. It was important to model data at the beginning and end of the sampling time interval as well as during different seasons. Therefore, the first and last effluent sampling dates were chosen to be modeled and the other five dates were selected randomly from the remaining effluent sampling dates. The main objectives of the initial inverse geochemical modeling were:

1) Predict what minerals were dissolving and precipitating within the pile and compare predicted minerals to observations from the pile.

2) Predict the effluent composition of the pile and compare to measured effluent data.

NETPATH Modeling

Two waters (initial and final) and their actual dissolved concentrations were added into the NETPATH model. The composite rainfall data used as the initial water was obtained from the Marcell Experimental forest in Itasca County approximately 75 miles west of the stockpile (National Atmospheric Deposition Program, 1995). Final water composition was entered into the NETPATH model using effluent data from the pile (Table 1). The Eh of the system was

not measured, but was assumed to be an oxidizing system and entered as 800 mV. Studies conducted to determine the sensitivity of the model to variations in Eh demonstrated that differences in Eh (± 100 mV) did not significantly affect results. Average annual temperature in the area was entered as 3.6 °C. Ripley and Alawi (1986) determined that in the Babbitt region, plagioclase composition was $(Ca_{0.60},Na_{0.40})Al_{1.36}Si_{2.64}O_8$ and olivine composition was $(Mg_{0.997}, Fe_{1.00}, Ni_{0.003})SiO_4$. These compositions were entered into the geochemical database, so the modeling would reflect actual mineral compositions being weathered. For each specific date, the amount of each mineral within the pile that dissolved/ precipitated was recorded in the output of the NETPATH model. The amounts of the minerals for sampling date 10/16/89 that were predicted to dissolve are given in Table 2.

MINTEQAK Modeling

Information obtained through the NETPATH modeling of major mineral constituents was incorporated into the MINTEQAK modeling of the system including all trace metals found in the effluent. All of the models run with MINTEQAK allowed ferrihydrite, montmorillonite, kaolinite, SiO_2, gypsum, and the hydroxide forms of the four trace metals of interest, Co, Cu, Ni, and Zn to precipitate if supersaturation were to occur. The modeling showed that SiO_2, montmorillonite and large quantities of ferrihydrite would precipitate, all of which had been verified as precipitates by MNDNR researchers (Table 3). The model did not predict, however the precipitation of gypsum, which had also been identified by MNDNR as a precipitate. It was hypothesized that perhaps gypsum was precipitating with summer evaporation. When evaporation was simulated using MINTEQAK, gypsum did begin precipitation after 15% evaporation.

Table 2: Minerals predicted to dissolve by NETPATH from 10/16/89 effluent data.

Mineral	mg/kg water
Plagioclase	555.1
Olivine	1092.0
Diopside	749.3
Biotite	11.6

The MINTEQAK predicted effluent matched measured effluent very well, with the exception of magnesium (Table 4). The MINTEQAK predicted values for magnesium are high. This suggests that some unknown magnesium mineral may have precipitated in the pile that is not present in the MINTEQAK data base. Available magnesium-containing minerals, such as Mg-nontronite and magnesite, were modeled but none became supersaturated or gave satisfactory results. Downward adjustment of the magnesium content of the olivine being weathered within reasonable limits did not decrease the concentration sufficiently to match the effluent of the pile.

The precipitation of such large quantities of $Fe(OH)_3$ within the pile was considered to be a influential factor which would control the composition of the effluent over time. This led to the hypothesis that trace metals such as cobalt, copper, nickel and zinc were being adsorbed to the large quantities of ferric hydroxide, which was being formed from the oxidation of pyrrhotite and other sulfide minerals. Such a hypothesis would also explain why the trace element hydroxides and carbonates were not predicted to saturate and precipitate by MINTEQAK.

Table 3: Minerals predicted to precipitate by MINTEQAK from 10/16/89 effluent data.

Mineral	mg/kg water
$Fe(OH)_3$	1452.0
SiO_2	696.0
Montmorillonite	714.9

Table 4: MINTEQAK predicted effluent versus actual effluent data for 10/16/89.

Ionic Species	MINTEQAK (mg/L)	Effluent (mg/L)
SO_4^{2-}	801.1	800.0
Ca^{2+}	212.3	212.0
Mg^{2+}	**226.0**	**54.0**
Na^{+}	18.8	22.0
Cu^{2+}	5.91	5.9
Ni^{2+}	23.5	25.0
Co^{2+}	1.89	1.9
Zn^{2+}	0.98	1.0

Iron oxides have long been identified as strong scavengers of trace metals and have been extensively researched in the past (Morgan and Stumm, 1965; Jenne, 1968). They can be present as nodules, concretions, matrix components, cement between particles or as a coating on particles and have been shown to control trace metal distributions in many cases. Hydrous oxide minerals possess proton-bearing surface functional groups and therefore adsorption onto these solid phases is pH dependent. Jenne (1968) determined that sorption by these oxides tend to control Zn and Cu levels in soils. Because of this scavenging ability, it was hypothesized that the iron oxides precipitating in the pile may be adsorbing and perhaps be largely controlling trace metal concentrations such as Cu, Co, Ni and Zn in the effluent.

To verify this hypothesis, the secondary phase ferric hydroxide, and the associated trace metals, were chemically separated from samples within the pile. This was accomplished using the technique of sequential extractions, which allows sediments to be broken down into different mineral associated constituents. After the sequential extractions were performed, concentrations of the trace metals and iron were determined using ICP-AES to see the extent of the adsorption. The E3 extraction would contain ferric hydroxide and associated trace metals, and thus was the extraction of the most interest.

Sequential Extractions

The samples were extracted using the procedure described earlier and were analyzed by ICP-AES. The results discussed below pertain to data from iron and the trace metals of interest: Co, Cu, Ni and Zn with detection limits in ppb: Co (3.42), Cu (0.36), Fe (3.00), Ni (5.79) and Zn (5.33). Extraction E1 (water soluble) held minuscule concentrations of cobalt and nickel and copper, with zinc and iron concentrations less than their detection limits. Extraction E2 was found to have concentrations less than the detection limits for all five metals, indicating that these metals were not weakly adsorbed, but strongly bound to ferric hydroxide. Only the distribution of the metals in extractions E3, E4 and E5 will be discussed, as they are the extractions of primary interest. The results for copper and nickel are plotted in Figures 1-3. It must be noted that although figures for cobalt and zinc are not represented, they similarly follow the same trends as copper and nickel in extractions E3, E4 and E5.

Extraction E3, or the extraction associated with iron oxides, revealed that trace metal concentrations increase with depth, indicating downward transport by water percolation as the sulfide minerals are weathered. Concentrations of these trace metals were moderate in comparison to the ferric hydroxide concentrations and decreased in the order of Cu > Ni > Zn > Co (Table 5). Iron concentrations in E3 were converted to $Fe(OH)_3$ concentrations and were found to have a relatively constant concentration throughout the pile. Average concentrations for iron and the trace metals are given in Table 5. Extraction E4, associated with sulfides, and the residual extraction, E5 revealed an oxidation pattern for all of the trace metals contained in sulfide minerals, including iron. Weathering of the sulfide (E4) and resistant minerals (E5) were accelerated near the top of the pile and significantly decreased with depth. A notable observation is that an apparent oxidation cut-off is seen at sample number 1.3.2, whereby below this sample in the pile, oxidation is much less. At this depth in the pile it can be concluded that roughly two-thirds of the pile had been oxidized.

Extraction E3, associated with iron oxides, was the extraction of the most interest. Trace metals found in this extraction must have been strongly adsorbed to the ferric hydroxide or they would have been removed by extraction E2, which was designed to remove metal cations which are easily exchanged by other metals, or weakly adsorbed. To verify that strong adsorption had indeed occurred, MINTEQAK was again employed, this time incorporating adsorption modeling.

MINTEQAK Adsorption Modeling

Adsorption of the four trace metals onto ferric hydroxide for each of the seven sampling dates originally modeled without adsorption was completed. In order to compare the sequential extraction results from extraction E3 (associated with iron oxides) to the adsorption modeling, it was necessary to determine the type of adsorption occurring onto ferric hydroxide. Weakly adsorbed trace metals would have been stripped from the ferric hydroxide during extraction E2 (exchangeable), leaving only those trace metals which were tightly adsorbed to, or had coprecipitated with the ferric hydroxide in extraction E3. Subsequently, to produce accurate modeling results, site densities (N_s) and surface complexation constants (K_1) for strongly adsorbing species were used in order to simulate as close as possible the conditions under which trace metals would have been originally found in extraction E3 (Dzombak and Morel, 1990).

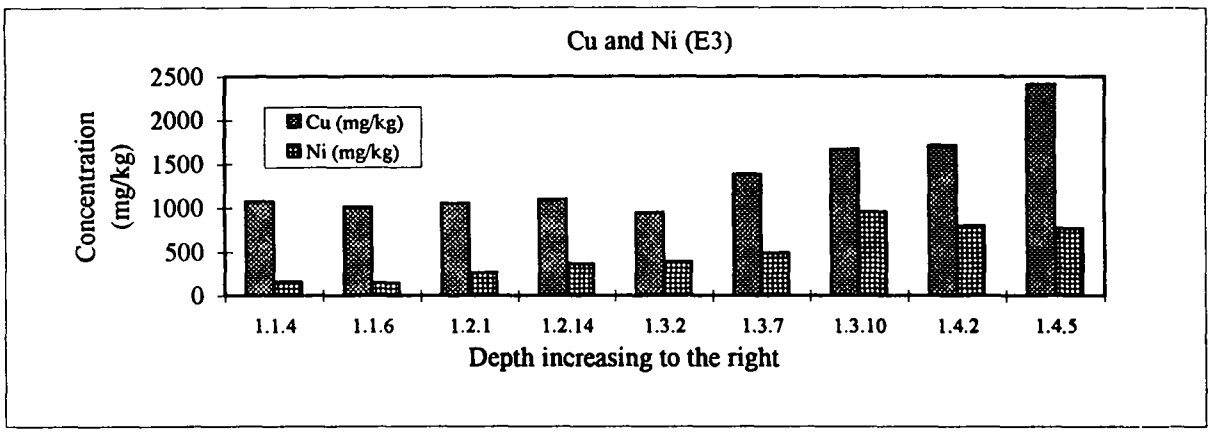

Figure 1. Distribution of copper and nickel concentrations in extraction E3 (associated with iron oxides).

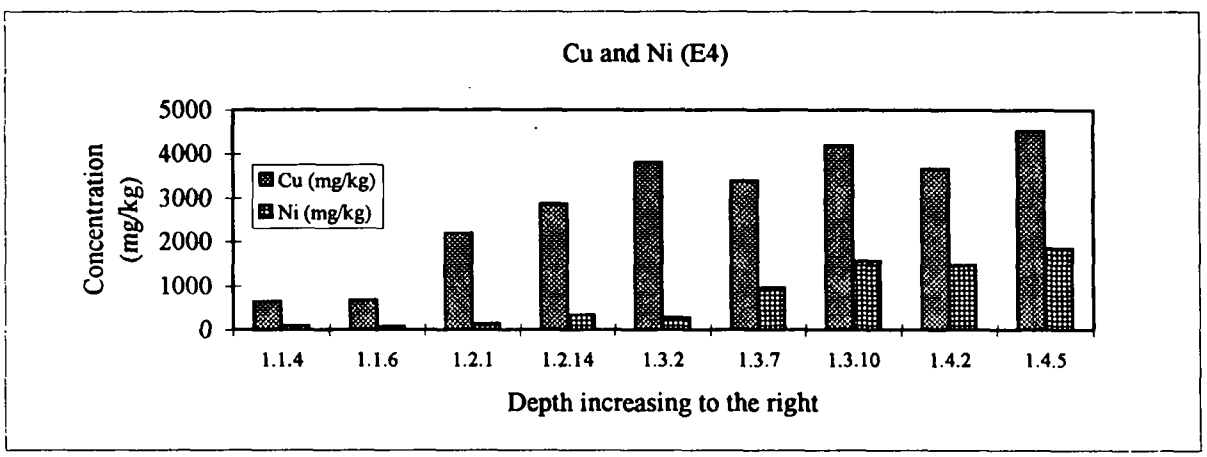

Figure 2. Distribution of copper and nickel concentrations in extraction E4 (sulfides).

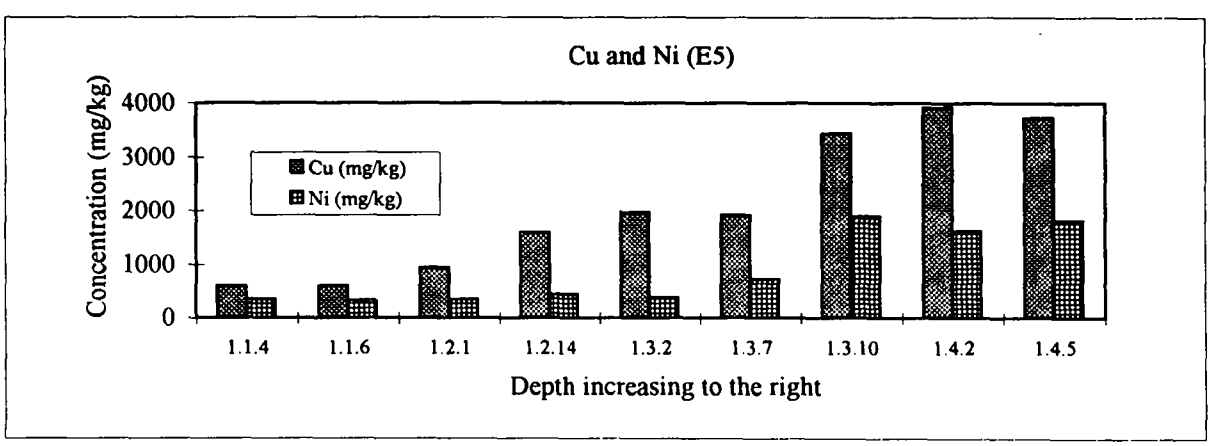

Figure 3. Distribution of copper and nickel concentrations in extraction E5 (residual).

Table 5: Average concentrations (mg/kg) for the elements Co, Cu, Fe, Ni, and Zn in extraction E3.

Element	Average E3 concentration (mg/kg)
Cobalt	18.4
Copper	1740.8
Iron	14680.4
Nickel	460.3
Zinc	64.9

The amount of $Fe(OH)_3$ available to adsorb trace metals was derived from output data from previous NETPATH trials. The average amount of $Fe(OH)_3$ precipitated (1.571 g of $Fe(OH)_3$/L of water or 1.5×10^{-2} M) for all seven of the sampling dates modeled was used in order to have a constant amount of $Fe(OH)_3$. Each of the four trace metals are allowed to compete for sites on the ferric hydroxide surface.

Of the four trace metals, copper was predicted by MINTEQAK to have by far the highest adsorption. Both nickel and zinc had minor amounts of adsorption onto ferric hydroxide (< 1% adsorption), with nickel adsorption the higher of the two metals. Cobalt was found to have "zero" adsorption onto ferric hydroxide. This result was anticipated since the surface complexation constant for cobalt is negative and because cobalt is present in small concentrations in the effluent. For the elements that were adsorbed, percent adsorption varied depending on the pH of the effluent. Figure 4 suggests that the adsorption of copper decreases with decreasing pH, demonstrating competition with protons in solution. This adsorption relationship with pH is also demonstrated with nickel and zinc, although not as pronounced. Decreasing adsorption can also be attributed to site saturation. As the effluent becomes more acidic, the dissolved concentrations of all four trace metals increase, thus saturating the adsorption sites with both the higher trace metal concentrations and protons, limiting the amount of trace metal adsorption. The ferric hydroxide is no longer capable of adsorbing the increasing concentrations of metals produced by weathering. Other reasons for limitations in adsorption include complexation and competition with other ions in solution, and the high pH_{zpc} of $Fe(OH)_3$ (pH_{zpc}= 8.0, Dzombak and Morel, 1990).

The results from the adsorption modeling verify that the trace metals found in extraction E3 were indeed strongly adsorbed to ferric hydroxide. The hypothesis that the strong adsorption sites were saturated and removed only limited amounts of the trace metals found in the effluent accounts for the small concentrations of trace metals in extraction E3. Both the adsorption modeling, and the measured concentrations of trace metals in extraction E3 confirms that copper was the most strongly adsorbed of the four trace metals, a consequence of high copper concentrations in the effluent and its strong complexation constant. Zinc concentrations in E3 and predicted adsorption onto ferric hydroxide had good correlation. Adsorption of cobalt was predicted by the modeling to be nonexistent, but minute amounts of Co in extraction E3 suggests that slight adsorption was taking place. Modeling and sequential extraction results, however differed in amounts of nickel and zinc adsorbed in extraction E3 and predicted to adsorb by MINTEQAK. The modeling predicted that nickel had adsorbed much less than copper, and in similar amounts as zinc, but the sequential extraction results demonstrated that adsorbed nickel concentrations were much closer in concentration to copper than zinc.

This inconsistency can be attributed to the inability of the model to accurately predict the influence of elements in high concentration in solution on adsorption. The high concentrations in the effluent increased adsorption, an influence poorly modeled.

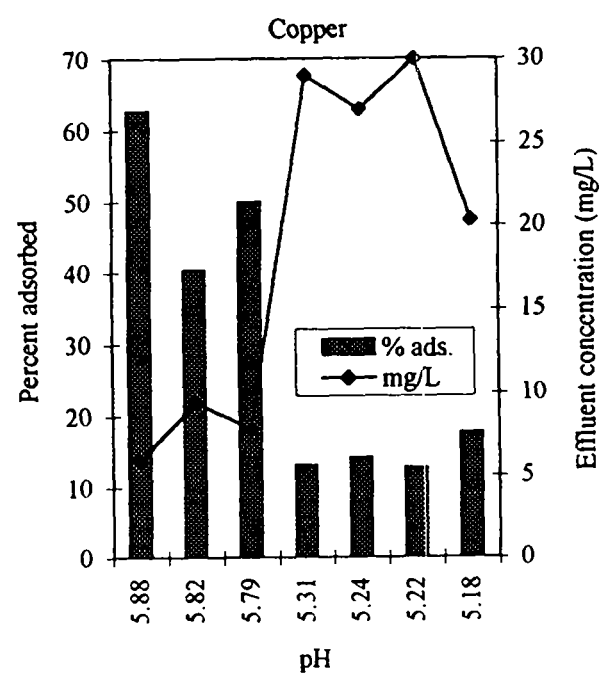

Figure 4. Percent adsorption of copper using MINTEQAK and concentration of copper in the effluent as a function of pH for seven sampling dates.

The fact that nickel adsorbed more than was predicted may verify research by Benjamin and Leckie (1981), who hypothesized that many of the strong binding sites for one metal are not preferred binding sites for other metals. This implies that nickel may have specific sites to which it is attracted to, which then inhibits adsorption by other metals with higher complexation constants. This site selectivity would not be compensated for in the model using simple surface complexation constants.

Summary and Conclusions

The primary objective of this research was to determine the nature of the weathering processes which were occurring within the pile and the effects of these processes on water quality. The evidence presented here illustrates that geochemical modeling was effectively used to confirm observations of minerals that were dissolving/ precipitating within the pile Geochemical modeling was also effective in supporting evidence of the adsorption of trace metals onto ferric hydroxide found in the extraction E3, or associated with iron oxides. Because adsorption of trace metals onto ferric hydroxide is minor, precautionary measures should be taken if Cu-Ni mining in the Duluth Complex develops. Rates of oxidation and weathering of sulfide minerals should be slowed as much as possible. This can possibly be achieved by the use of a layering system, consisting of polyethylene liners, soil and vegetation, covering the pile. Alternately, reduction of meteoric water input will also slow oxidation. This will help in the retardation of the oxidation and weathering processes, which result in the release of toxic levels of trace metals in the effluent and possibly to ground water as well.

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SOLUBILITY OF Fe (III) AND Al IN AMD BY MODELLING AND EXPERIMENT[1]

by

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Abstract: Studies of Fe(III) and Al species in acid mine drainage (AMD) alone and in contact with limestone were conducted by MINTEQA2 modelling and by experiments. The objectives of these studies were to: 1) determine at what pH Fe(III) and Al would be in solution in AMD such that the water would be harmful to an anoxic limestone drain (ALD), and 2) evaluate the theoretical limits to the amount of alkalinity that could be generated by an ALD. Using $Fe(OH)_3$ as the primary species and the standard values for MINTEQA2, Fe(III) precipitates at pH 2.90 when the concentration is over 453 mg/L. Al precipitates at a pH of 4.00 when the concentration is over 108 mg/L. Experiments found that over 90 % of Fe(III) and 45 % of Al were precipitated at these pH's. Experimental verification of Fe(III) concentrations of pH's from 2.90 to 4.0 found that modelling agreed with experiment when ferrihydrite is the primary solid and the log Ksp (solubility product) is -38.9. For Al, gibbsite would be the primary solid and log Ksp is -34.1. For AMD in contact with $CaCO_3$ when CO_2 is conserved, final alkalinity is higher when mineral acidity is higher even though pH of the final solution is lower. This modelling result was confirmed by experiment. Higher mineral acidity causes generation of more CO_2 that reacts with $CaCO_3$ to generate more dissolved HCO_3^-.

Additional Key Words: Anoxic Limestone Drain, Mineral Acidity, Alkalinity

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Introduction

Anoxic Limestone Drains (ALD) are an important component in the arsenal of methods that can be used to passively treat acid mine drainage (AMD) (Brodie, et al., 1991; Nairn, Hedin, and Watzlaf, 1991). The function of an ALD is to add alkalinity to the water through the dissolution of calcium carbonate. The chemical reactions that govern this dissolution are:

$$CaCO_3\,(s) + 2H^+ = Ca^{2+} + CO_2\,(aq) + H_2O \quad (pH < 6.4)$$

$$CaCO_3\,(s) + CO_2\,(aq) + H_2O = Ca^{2+} + 2HCO_3^-$$

$$CaCO_3\,(s) + H^+ = Ca^{2+} + HCO_3^- \quad (pH > 6.4)$$

Alkalinity is produced in the form of HCO_3^-.

Below a pH of 6.4, aqueous CO_2 is the primary carbonate species, while above pH of 6.4 HCO_3^- becomes the primary species. An oxidation pond is placed after the ALD to oxidize Fe(II) to Fe(III) and use the alkalinity generated to buffer the H^+ acidity generated

during $Fe(OH)_3$ precipitation. The stepwise reactions are:

$$Fe^{3+} + 3\ H_2O \text{ ----> } Fe(OH)_3 + 3\ H^+$$

$$HCO_3^- + H^+ \text{ ---> } H_2O + CO_2\,(aq)$$

This gives an overall net reaction of:

$$Fe^{3+} + 3\,HCO_3^-\ (aq) \text{ -----> } Fe(OH)_3 + 3\ CO_2\,(aq)$$

In this paper, Fe(II), Fe(III), and Al(III) are used to designate all dissolved species of the metal. For example, Fe(III) could include dissolved Fe^{3+}, $Fe(OH)^{2+}$, and $Fe(OH)_2^+$.

Early in the development of ALD's it was found that dissolved Fe(III) and Al(III) impaired the function of an ALD by armoring the limestone with hydroxide precipitates. This restricted the dissolution of limestone (Brodie, et al., 1991). In addition, buildup of hydroxide precipitates caused ALD's to plug (Watzlaf et al. 1994). Brodie and others (1991) suggested 10 mg/L of Al(III) and 1.0 mg/L of Fe(III) as the maximum amount of Al and Fe in the AMD flowing into an ALD. Hedin and Nairn (1992) advised that if the concentration of Al(III) and Fe(III) exceeded 1.0 mg/L that performance could be compromised. In the latest USBM guidelines on passive treatment, Hedin, Nairn, and Kleinmann (1994) placed the maximum concentrations of Al(III) and Fe(III) that could be tolerated within an ALD at 1.0 mg/L.

Both Al(III) and Fe(III) cause this armoring because they will react with water even in fairly acidic solution to form hydroxides. The reaction for Al is:

$$Al^{3+} + 3\,H_2O \text{ ------> } Al(OH)_3 + 3\,H^+$$

For Fe(III), hydrolysis occurs at pH's between 2.75 and 3.25; for Al(III), at pH's between 4.0 and 4.5.

Because the concentrations of Al(III) and Fe(III) are so critical to determining whether an ALD can be used for treatment, the first objective of this study was to determine what concentrations of these metals were predicted by an equilibrium modelling program such as MINTEQA2 (Felmy, Girvin, and Jenne 1983, Allison, et al., 1991). The second objective was to determine whether the concentrations predicted by MINTEQA2 modelling were confirmed by Fe(III) and Al(III) solutions in the laboratory.

In addition to determining Al and Fe concentrations in AMD, another objective was to determine the maximum amount of alkalinity that can be generated by an ALD. Hedin, Watzlaf, and Nairn (1994) noted significant differences between the alkalinity in two ALD systems that received water of very similar quality. Also, Watzlaf and Hedin (1993) had developed a method for predicting the alkalinity that could be generated in an ALD using cubitainers. The same methods of modelling confirmed by laboratory experiments were used to determine the pH and alkalinity that could be theoretically generated when limestone was brought into contact with various acidic solutions containing Al(III) and Fe(III).

This paper reports on the results of modelling and laboratory studies that were performed on Al(III) and Fe(III) solutions and these solutions in contact with $CaCO_3$.

Experimental Methods

For the MINTEQA2 modelling exercises, the equilibrium pH was set at various acidic values, Al(III) or Fe(III) was added at various concentrations at the set pH to determine the minimum concentration that must be present for precipitates to form. Sulfate was used to balance charges, and no other ions were entered. For Fe, ferrihydrite ($Fe(OH)_3$) with log Ksp (solubility product) of -37.1 (with

respect to Fe^{3+}) was entered as the possible solid. For Al, crystalline gibbsite ($Al(OH)_3$) with log Ksp of -33.2 (with respect to Al^{3+}) was entered as the possible solid. In separate runs, amorphous $Al(OH)_3$ with log Ksp of -31.6 was entered as the possible solid.

For modelling the interaction of $CaCO_3$ with acidic solutions, $CaCO_3$ was set as an infinite solid and the above hydroxide precipitates were set as possible solids. Initial pH and Fe(III) or Al(III) concentrations were set, and the system was allowed to come to a new equilibrium. In the calculation, CO_2 partial pressure was allowed to exceed the atmospheric value of $10^{-3.5}$. This is the case in actual ALD's where attempts are made to retain the CO_2 that is generated (Brodie, et al., 1991; Hedin, Watzlaf, and Nairn, 1994). In all cases, 100 % of the Al and Fe precipitated as hydroxides. At the new equilibrium pH, the alkalinity was calculated from the concentration of dissolved HCO_3^-.

In the laboratory experiments where the precipitation curves for Al(III) and Fe(III) were determined, solution concentrations from 1 to 1000 mg/L of the cation were used. Ferric ammonium sulfate and hydrated aluminum sulfates were used to make the stock solutions. The pH of the solution was set using NaOH and H_2SO_4. Aliquots of the solution were taken at 24, 72, and 96 hours and the pH of the solution was readjusted to the initial value after the first two aliquots were taken. The aliquots were filtered through 0.45 micrometer filters. Fe concentration was determined by flame atomic absorption and aluminum was determined colorimetrically using aluminon reagent.

In the limestone experiments, 3.8 liter collapsible, low-density polyethylene cubitainers were used in a configuration similar to that of Watzlaf and Hedin (1993). A quantity of 4.0 kg of washed limestone of greater than 90 % $CaCO_3$ in quality, of 0.3 to 1.0 cm in diameter was added to the cubitainer. Approximately 2 liters of Al(III) and Fe(III) solutions, whose pH's had been set, were added until the cubitainer overflowed and no air was present. At times of 3, 6, 12, 24, and 48 hours, aliquots of 20 mL were removed by squeezing the cubitainer. For each aliquot, alkalinity and pH were measured, then the sample was filtered and acidified and Fe and Al concentrations were determined.

Further details on the modelling and laboratory studies can be found in Mitchell (1994).

Results of the Precipitation and Limestone Equilibration Studies

The modelling results compared with the laboratory-determined, saturation concentrations of Fe(III) and Al(III) are shown in Figures 1 and 2 respectively. The modelling runs for iron were quite surprising because, out to a pH of 4.0, Fe(III) was still in solution at concentrations above 1.0 mg/L. Based on field experiences in Colorado, the rule of thumb we use is that Fe(III) should not be in solution beyond a pH of 3. The rule of thumb was confirmed in the laboratory studies. In experiments where the pH was set to approximately 3.5, the Fe(III) concentrations averaged 0.10 mg/L. In experiments where the pH ranged from 2.99 to 3.01 the concentration of Fe(III) ranged from 1.3 to 4.2 mg/L. The difference between the experimental and modelling results is probably due to the inclusion of Fe(III) complexes, such as $Fe(OH)_2^+$, in the model.

For aluminum, the experimental results generally followed the modelling concentrations when Al(III) is in equilibrium with crystalline gibbsite. In experiments where the pH was set to approximately 4.0, the Al concentrations ranged from 19 to 46 mg/L. In experiments

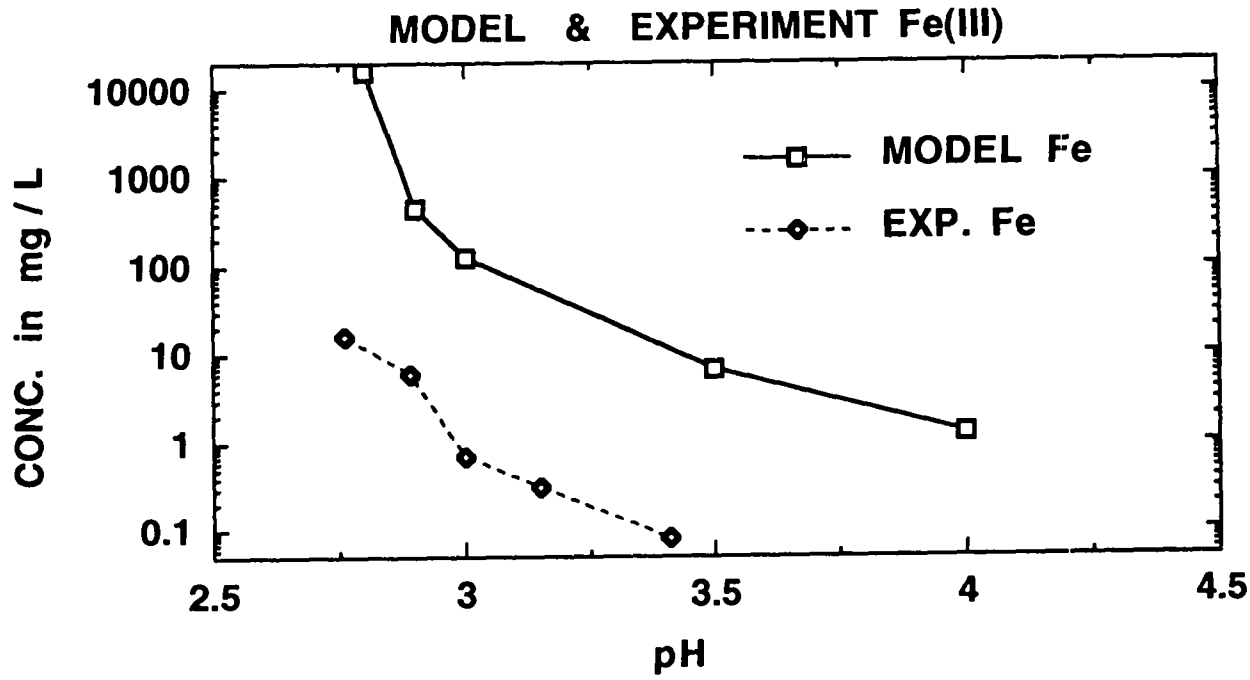

Figure 1. Model and experiment concentrations of Fe(III) in equilibrium with ferric solids at acidic pH values.

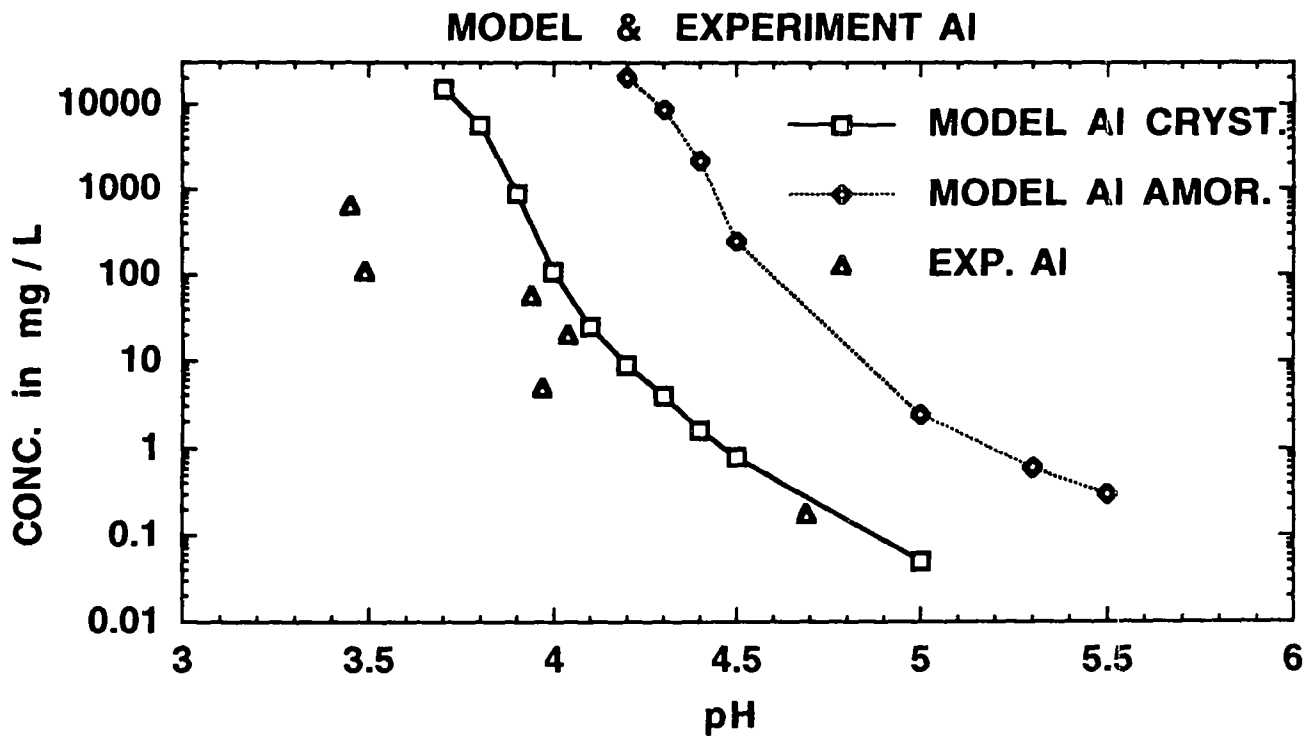

Figure 2. Model and experiment concentrations of Al(III) in equilibrium with aluminum solids at acidic pH.

Table 1. Modelling and laboratory results of limestone equilibrated with solutions of Fe(III) and Al(III) at various pH values. Alkalinity is in units of mg $CaCO_3$ / L. Laboratory results were after 48 hours of reaction.

			Modelling Results			Laboratory Results	
Initial pH	Initial Fe(III)	Initial Al	Final pH	Final Alkalinity	Log P (CO2)	Final pH	Final Alkalinity
2.9	50.0	0.0	7.16	152	-1.94	7.5	131
2.9	50.0	0.0	7.16	152	-1.94	7.5	132
2.9	100	0.0	6.87	213	-1.48	7.35	160
2.9	0.0	0.0	7.84	58	-4.41	7.55	87
4.0	0.0	50	6.93	190	-1.58	7.30	173
4.0	0.0	100	6.57	270	-1.07	7.30	173
3.5	1.0	100	6.57	277	-2.56	7.04	187
3.5	5.0	400	6.02	500	-1.78	6.66	280
3.5	5.0	400	6.02	500	-1.78	6.57	288

where the pH was set to approximately 4.7, the concentration of Al ranged from 0.12 to 0.27 mg/L.

In Table 1, the modelling and laboratory results of limestone equilibrated with waters containing various concentrations of Al(III), Fe (III), and H^+ are presented. In Figure 3, the experiment and model pH's are compared. In Figure 4, the experiment and model alkalinities are compared.

The modelling results show that when CO_2 gas is conserved, it indeed does exceed the atmospheric value of $10^{-3.5}$. In turn, this does indeed increase the alkalinity. For the laboratory samples, except for the acid solution with no Fe(III) and Al(III), the final alkalinity is lower and the pH higher than the model predictions. In Table 1, the final experimental values of the alkalinity taken after 48 hours are given. In systems with high mineral acidity, the excess CO_2 gas far exceeds the atmospheric value. In experiments with 100 and 400 mg /L of Al(III) where there was excess CO_2, the alkalinity peaked at between 12 and 24 hours and then consistently decreased. The maximum values for these experiments are also plotted in Figure 4.

Discussion

For the precipitation experiments, additional modelling was performed to test the possibility that phases other than ferrihydrite and gibbsite were controlling the Fe(III) and Al(III) concentrations. For Fe, maghemite (Fe_2O_3) gave the best fit. However, this is an unrealistic phase to form in a near surface environment. Use of lepidocrocite (FeOOH) caused the precipitation of too much iron, and use of hydrogen jarosite ($HFe(SO_4)_2(OH)_6$) caused too much Fe(III) precipitation at low pH's and not enough at high pH's. For Al, diaspore (AlOOH) and jurbanite ($AlOSO_4$) caused the precipitation of too much aluminum. When boehmite (AlOOH) and basaluminite ($Al_4(OH)_{10}SO_4$) were tried, not enough aluminum precipitated. In the study of Butte, Montana Berkeley Pit water, Davis and Ashenberg (1989) also found that the AMD was supersaturated with respect to jurbanite. Apparently, precipitation of jurbanite is kinetically hindered.

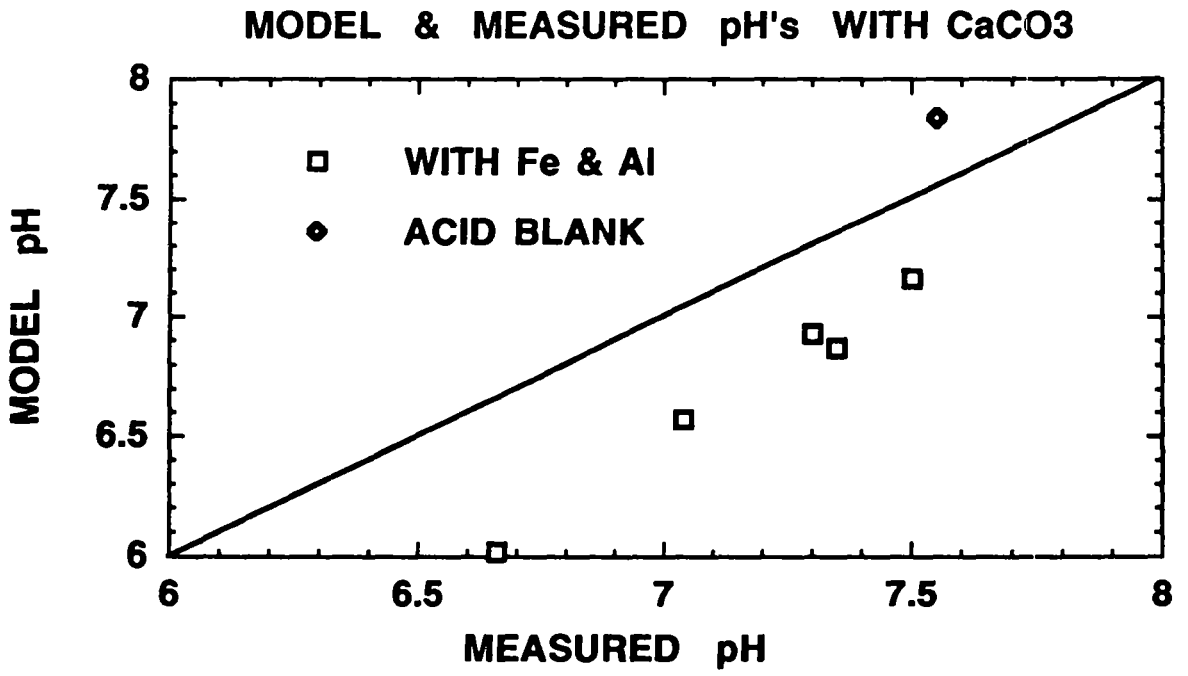

Figure 3. Model and experiment pH values from the reaction of acidic solutions of Fe(III) and Al(III) equilibrated with $CaCO_3$. In the modelling and experiments, the CO_2 generated was retained.

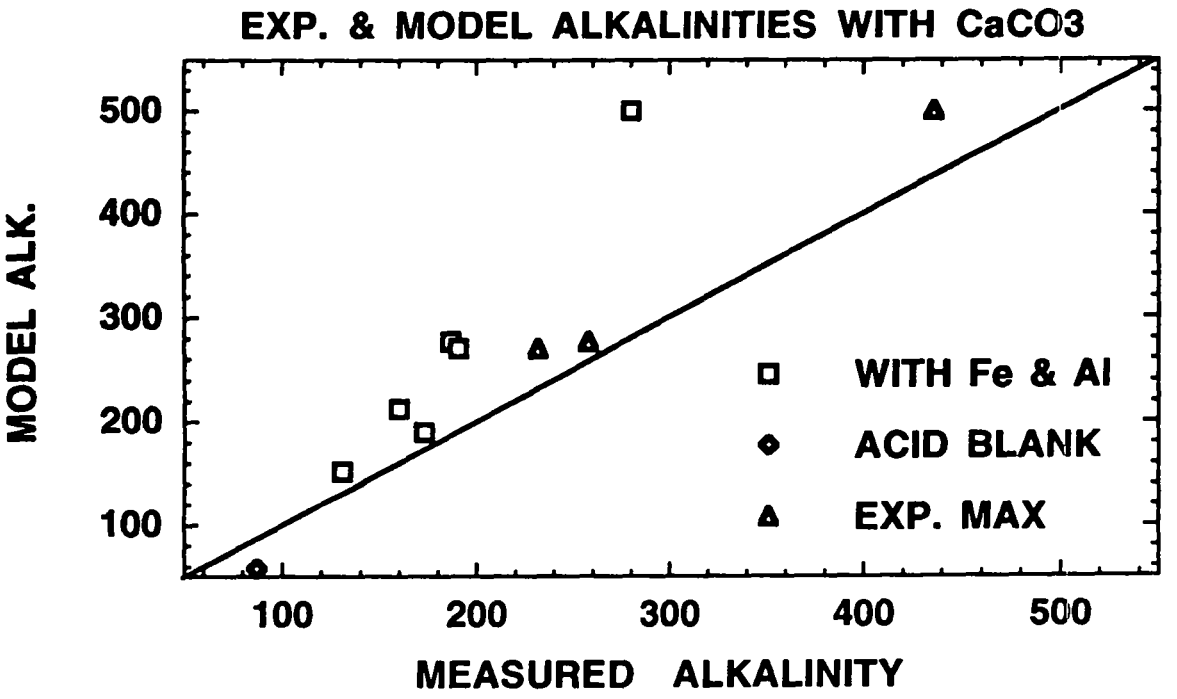

Figure 4. Model and experiment alkalinities from the reaction of acidic solutions of Fe(III) and Al(III) equilibrated with $CaCO_3$. In the modelling and experiments, the CO_2 generated was retained. The experimental maxima occurred between 12 and 24 hours. The final values were after 48 hours.

Another possibility for resolving the laboratory results with MINTEQA2 modelling is to determine the solubility product of $Fe(OH)_3$ and $Al(OH)_3$ based on the laboratory results and use this value in the model. For the 15 Fe precipitation experiments, the average solubility product on a log basis is -38.9 ± 0.3. For the 10 Al precipitation experiments the average solubility product on a log basis is -34.1 ± 0.5.

Literature values for the log Ksp for ferrihydrite range from -36 to -39 (Chapman, Jones, and Jung 1983). MINTEQA2 allows a range for ferrihydrite log Ksp of -37.00 to -40.44, and uses a default value of -37.1 if nothing is specified. Chapman, Jones, and Jung (1983) studied the processes controlling metal ion concentrations in AMD and determined that ferrihydrite was the most reasonable precipitate and a log Ksp of -39 produced the most consistent results. Based on the results from this and other studies, using ferrihydrite with a log Ksp of -39 as a possible solid phase is the most reasonable procedure for modelling the concentration of Fe(III) in AMD.

Literature values for the log Ksp for $Al(OH)_3$ range from -31.6 to -36.3 (Chapman, Jones, and Jung 1983). MINTEQA2 has maximum and minimum values for log Ksp of -32.56 to -33.51 for gibbsite, and uses a default value of -33.23 if nothing is specified. Chapman, Jones, and Jung (1983) determined that amorphous $Al(OH)_3$ was controlling the aluminum concentration, and a log Ksp of -31.6 produced the most consistent results. The value of log Ksp of -34.1 determined in this study is outside the values used in MINTEQA2 and in the Chapman, Jones, and Jung (1983) study. However it is within the range of literature values. Based on the results from this and other studies, using gibbsite as a possible solid phase is the most reasonable procedure for modelling the concentration of Al(III) in AMD. However, using the default value of log Ksp of -33.23 will produce an upper bound on aluminum concentration and a value of -34.1 will produce a lower bound.

The modelling and laboratory studies of the equilibration of acidic Fe(III) and Al(III) solutions with $CaCO_3$ do not completely parallel the reaction of AMD in an actual ALD. This is because Fe(III) and Al(III) hopefully are not present when AMD courses through an ALD, and consequently, precipitation of hydroxides is avoided. In these studies, complete precipitation of Fe(III) and Al(III) occurred. Nevertheless, certain aspects of this study apply to any time AMD reacts with limestone including the reaction of AMD in an ALD. In particular, the property of alkalinity increasing and pH decreasing with increasing mineral acidity applies whenever AMD reacts with limestone. This somewhat counter-intuitive property can be understood by considering the following equilibrium reaction:

$$CO_3^{=} + CO_2(g) + H_2O \text{ <-----> } 2\,HCO_3^{-}$$

The reactant CO_2 gas is generated from the reaction of AMD with $CaCO_3$ according to the first reaction in the **Introduction** and is retained in the system rather than being released. The reactant $CO_3^{=}$ comes from calcite dissolution. The greater the mineral acidity, the more CO_2 is generated to react with the calcite, and the more bicarbonate alkalinity is produced. If the water is allowed to degas and CO_2 escapes, this shifts the above reaction to the left and alkalinity is reduced. However, when the reaction shifts to the left, more $CO_3^{=}$ is produced and the pH increases.

With respect to the reaction of AMD with an ALD, the amount of alkalinity generated depends on the pH of the water entering the ALD. If the CO_2 is retained, the lower the pH, the greater the amount of alkalinity that theoretically can be generated. Greater alkalinity does not

necessarily mean a higher pH. In addition, if the CO_2 is retained and reaches pressures greater than the atmospheric pressure of $10^{-3.5}$, then this CO_2 has the possibility of degassing from the water when the AMD breaches the surface after coursing through the ALD. In such a situation, the pH would rise, but there would be less alkalinity available for precipitation of $Al(OH)_3$ and $Fe(OH)_3$.

Conclusions

Modelling and laboratory experiments studying the chemistry of acidic solutions of Al(III) and Fe(III) alone and in contact with $CaCO_3$ has lead to the following observations:

- The precipitation curve for $Fe(OH)_3$ occurs at about a pH of 3 so that by a pH of 3.5, the concentration of Fe(III) is down to 0.1 mg/l. For Al, the precipitation curve for $Al(OH)_3$ occurs above a pH of 4 and at a pH of 4.7, the concentration is approximately 0.20 mg/L.
- For modelling Fe(III) chemistry in AMD, ferrihydrite with a log Ksp of -39 is the best choice for a possible phase. For modelling Al(III) chemistry in AMD, gibbsite is the best choice for a possible phase. The value of log Ksp to use is a bit more uncertain. This study suggests a value of -34.1.
- In the reaction of AMD with $CaCO_3$, if the CO_2 is retained, the higher the mineral acidity, the higher the possible alkalinity that can be generated, and the lower will be the pH at equilibrium.

- If, when AMD reacts with an ALD, the CO_2 is retained and reaches pressures far above the atmospheric value of $10^{-3.5}$, the CO_2 can dissolve from the AMD and be lost to the atmosphere when it breaches the surface. If this occurs, the pH will rise and alkalinity will be lost.

From a practical viewpoint, aluminum is not always analyzed in AMD. Consequently, its presence in water may be overlooked in assessing the appropriateness of an ALD. In a 1995 survey of AMD from metal mines in Colorado, a significant number of the waters that had pH below 4 had concentrations of Al above 1 mg/L. In the opinion of the authors, a maximum concentration of Al of 1 mg/L is presently a conservative guideline based on the fact that people are unsure of how much dissolved aluminum will plug an ALD.

The other important feature that this study reveals is how important initial acidity of the water and retention of CO_2 are to the final alkalinity of the AMD. Loss of CO_2 or precipitation of iron within the ALD could account for the alkalinity differences that Hedin, Watzlaf, and Nairn (1994) found in their study. The high alkalinity values shown in Table 1 and in Figure 4, generated when all the Fe(III) and Al(III) were allowed to hydrolyze should be carefully considered. This was caused by the generation of more CO_2 which reacted with the calcite. In an actual ALD, no precipitation can occur because it would eventually plug the structure. In Table 1, the alkalinity values for the system where no Fe(III) and Al(III) are present may be closer to what should be expected in the field. In-field alkalinity values of 300 mg/L of $CaCO_3$ may look very desirable. However, such high values may only be an indication that Fe(III) or Al(III) are precipitating in the ALD.

Acknowledgement

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PYRITE MICROENCAPSULATION: POTENTIAL FOR ABATEMENT OF ACID MINE DRAINAGE[1]

by

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Abstract. Oxidation of pyrite in mining waste or overburden is the main source of acid mine drainage (AMD) production which causes major environmental pollution. Presently, the most common method of controling AMD problems is through the mixing alkaline substances, such as limestone, with the AMD producing materials. However, the effectiveness of this method is still questionable. The main reason for this is that the surface of pyrite particles in mining waste are still exposed to the atmospheric O_2 after treatment. Experimental evidence on novel pyrite microencapsulation technologies currently under development in our laboratory are presented. It was demonstrated that these technologies, which include ferric hydroxide-phosphate-coatings and ferric-hydroxide-silica coatings, could effectively protect pyrite from oxidation.

Introduction

Pyrite is a mineral commonly associated with coal and most metal ores. When exposed to the atmosphere, pyrite initially reacts with oxygen producing H^+, SO_4, and Fe^{2+} (Nordstrom, 1982). The Fe^{2+} produced can be further oxidized by O_2 into Fe^{3+}, which in turn hydrolyses into amorphous iron hydroxide and releases additional amounts of acid into the environment (Nordstrom, 1982; Fornasiero et al., 1992). During the initial stage of pyrite oxidation, the process is relatively slow. However, as acid production continues and pH in the vicinity of the pyritic surfaces drop below 3.5, the formation of ferric hydroxide is hindered and activity of free Fe^{3+} in solution increases (Lindsay, 1979). Under these conditions, oxidation of pyrite by Fe^{3+} becomes the main mechanism for acid production in coal waste because Fe^{3+} can oxidize pyrite at a much faster rate than O_2 (Singer and Stumm, 1970). In addition, at low pH, an acidophilic, chemoautotropic, iron-oxidixing bacterium, *Thiobacillus ferrooxidans*, can catalyze and accelerate the oxidation of Fe^{2+} into Fe^{3+} by a factor larger than 10^6 (Singer and Stumm, 1970).

The approaches currently used to prevent pyrite oxidation in the field are mainly aimed at eliminating Fe from pore waters and/or blocking the access of the atmospheric O_2 to pyritic surfaces. These approaches include the use of limestone or rock phosphate to precipitate Fe in the insoluble form as iron hydroxide/oxyhydroxide or $FePO_4$, respectively (Skousen et al., 1987 and references therein; Renton et al., 1988; Brown and Jurinak, 1989), and the application of bactericides to inhibit the oxidation of Fe^{2+} into Fe^{3+} (Kleinmann, 1980). Both approaches have shown a certain degree of success in preventing pyrite oxidation and acid production in pyritic waste. However, they both have a weakness, i.e., they have a

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short span of effectiveness. The main reason for this is that the surface of pyrite particles in mining waste are still exposed to the atmospheric O_2 after treatment. Consequently, as Fe^{2+} accumulates and *T. ferrooxidans* repopulates on pyritic surfaces, pyrite oxidation by Fe^{3+} and acid production initiates (Kleinmann and Crerar, 1979). In addition, recent findings suggest that limestone may actually accelerate, under certain conditions, the pyrite oxidation process (Evangelou and Huang, 1994a). Therefore, better understanding on the effects of environment on the pyrite oxidation mechanisms is needed to make sound recommendations on the use of appropriate technologies in controlling AMD. To completely prevent pyrite oxidation, it appears essential to block the access of the atmospheric O_2 to pyritic surfaces. Recently in our laboratory we developed a number of pyrite coating technologies to inhibit pyrite oxidation. These technologies include (a) iron-oxide coating, (b) iron-phosphate coating, and (c) iron-silica coating (Evangelou and Huang, 1994b; Evangelou, 1996). The objective of this study is to demonstrate the use of the novel pyrite microencapsulation technologies developed in our laboratory for abatement of acid mine drainage in field leaching columns.

Materials and Methods

Coating Technology Experiments

Three new microencapsulation (coating) methodologies for preventing pyrite oxidation and acid production in coal pyritic waste have been developed in our laboratory using small leaching columns. The first coating methodology is that of *iron-oxide-phosphate coating.* This was done by leaching coal waste with a solution composed of 0.01 *M* sodium acetate buffer (NaOAc) plus 0.106 *M* H_2O_2 and 0.001 *M* KH_2PO_4. During the leaching process, H_2O_2 oxidizes pyrite and produces Fe^{3+} so that iron phosphate precipitates as a coating on pyrite surfaces. The purpose of pH buffer (NaOAc) in the coating solution is to eliminate the inhibitory effect of the protons, produced during pyrite oxidation, on the precipitation of iron phosphate (Evangelou, 1995a,b).

The second coating methodology is that of an *iron-oxide coating.* This was done by leaching pyritic waste with a solution composed of 0.106 *M* H_2O_2 and 0.01 *M* NaOAc (Evangelou and Zhang, 1995).

The third coating methodology is that of an *iron-oxide-silica coating.* This was done by leaching pyritic waste with a solution composed of 0.145 *M* H_2O_2 and 50 mg L^{-1} of dissolved Si at pH 5 adjusted with 0.01 *M* NaOAc. In order to demonstrate that the coating on the surface of pyrite was acid resistant because it was composed of two distinct layers, an iron-oxide layer (acid sensitive) and a silicon oxide layer (silica) (acid resistant), an iron-oxide-silica coated pyrite sample was treated as follows: sample A was leached with water; sample B was oxidized with 0.145 *M* H_2O_2; sample C was leached with 50 ml of 4 *M* HCl and then it was oxidized with 0.145 M H_2O_2; sample D was treated with 50 ml of 4 *M* HF (remove the iron-oxyhydroxide coating as well as the silica coating) and then it was oxidized with 0.145 *M* H_2O_2. The purpose of these two treatments (HCl versus HF) was to demonstrate that silica coating on the surface of pyrite was produced and that this coating was resistant to acid attack.

Outdoor Leaching Column Experiments

In order to test the coating technology under natural environment for possible mass application in the field, an outdoor leaching column experiment was conducted. The set of leaching columns consisted of three treatments in duplicate: limestone, phosphate coating, and silica coating. The leaching columns were made from plexiglass tubing with a height of 18" and inside diameter of 6" equipped with a leachate collection system. All columns were fitted with drain tubes. Actual support for the columns were weighted 5 gallon plastic buckets, and collection vessels were one gallon polypropylene jugs.

The phosphate coating columns were constructed by filling the column with 0.5 kg mine tailing mixed with 9.5 kg sharp sand. Furthermore, 800 g of limestone covered with 107 g of $Ca(OCl)_2$ was added to the column to neutralize all of the potential acidity of the columns and 5 L of coating solution consisting of 0.1 *M* NaAC and 0.001 *M* KH_2PO_4 (P = 30 mg L^{-1}) at pH 5, was then leached down to the columns prior to setting up in the field. The same method was used for silica coating except that in this treatment 0.001 *M* Na_2SiO_3 (Si = 30 mg L^{-1}) was used instead of phosphate. The limestone treatment was constructed by filling the column with 0.5 kg mine tailing mixed with 9.5 kg sharp sand and 800 g limestone and were leached with deionized water only. After a 1-3 week incubation time the columns were set up in the field. Leachates were collected and analyzed periodically for pH, Fe, and SO_4-S concentration. Finally, after a 3 month period in the field, pyrite oxidixing bacteria were introduced to the columns to evaluate the pyrite coating technology in preventing microbial oxidation.

Results and Discussions

Coating Technologies Experiment

The phosphate coating process is shown schematically in Fig. 1. The dotted lines in Fig. 1. signify physical bonding between pyrite and $FePO_4$. When iron (Fe^{3+}) reacts with PO_4^{3-} it forms an acid resistant ferric phosphate ($FePO_4$) coating, which inhibits oxidation of pyrite as shown in Fig. 2A (see

also Evangelou, 1995a,b). It is important to note that the technology of coating pyrite as described above, is not to be confused with field application of rock phosphate (Flynn, 1969). Rock phosphate complexes dissolved iron (Fe(II)), thus, reducing the potential of Fe(III) production, and reducing the potential for pyrite oxidation as well (Stumm and Morgan, 1970). Rock phosphate does not coat pyrite; rather, it complexes released Fe(II) from the oxidizing pyrite (Evangelou et al., 1992). Instead, a rock phosphate surface coating with Fe(II) forms, reducing rock phosphate dissolution. Therefore, the effectiveness of rock phosphate in controlling pyrite oxidation is short lived.

During the leaching process, H_2O_2 oxidizes pyrite and produces an iron oxide coating on the surface of pyrite (Fig. 2C). The purpose of a pH-buffer in this case is to buffer the solution during coating formation at a pH between 5 and 7 where iron oxide formation is promoted.

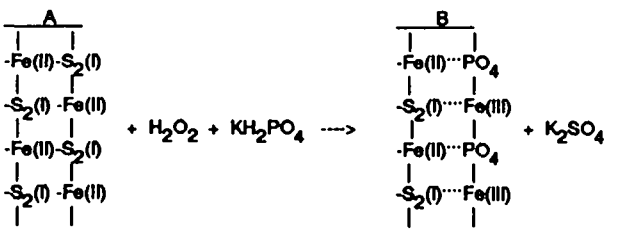

Figure 1. Schematic of H_2O_2 induced oxidation proof phosphate surface coating on iron sulfides (Evangelou and Huang, 1994b, U.S. Patent No. 5,286,522, Feb. 15, 1994).

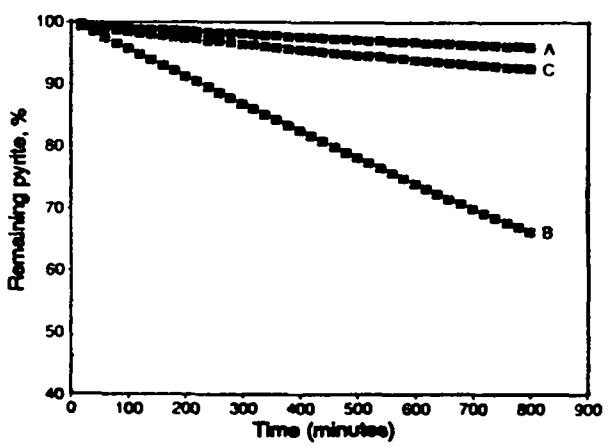

Figure 2. Oxidation kinetics of pyrite leached with the following three pH 6 solutions: A) 0.01 *M* sodium acetate (NaOAc) plus 0.106 *M* H_2O_2 and 0.001 *M* KH_2PO_4; B) 0.01 *M* NaOAc plus 0.106 *M* H_2O_2 and 0.013 *M* EDTA; C) 0.01 *M* NaOAc plus 0.106 *M* H_2O_2.

The data in Fig. 3 show the oxidation potential of framboidal pyrite by 0.145 *M* hydrogen peroxide (H_2O_2) in the presence and absence of 50 mg L^{-1} dissolved silica at pH 5 adjusted with 0.01 *M* sodium acetate. As can been seen, silica significantly suppressed the potential of H_2O_2 to oxidize pyrite. The explanation for this behavior is that oxidation of pyrite by H_2O_2 in the presence of Si and sodium acetate lead to the formation of an iron-oxide silica coating as shown schematically in Fig. 4. The resistance of silica coating to low pH or strong acid attack is demostrated in Fig. 5. The data from zero to 900 minutes represents the silica coating process of pyrite. After 900 minutes, the data labeled A represent leaching of iron-oxide-silica coated pyrite with oxygenated water alone. No pyrite oxidation was apparent. The data labeled **B** represent coated pyrite oxidation with 0.145 *M* H_2O_2, a strong pyrite oxidizer. These data show that the iron-oxide-silica coating protected pyrite from oxidizing by inhibiting H_2O_2 diffusion to the pyrite surface. The data in Fig. 5C (representing 4 *M* HCl treatment) show that oxidation of pyrite by 0.145 *M* H_2O_2 was greatly suppressed relative to that treated with 4 *M* HF (Fig. 5D). This strongly suggested that the silica part of the coating offers substantial protection to pyrite from H_2O_2 (a very strong oxidizer) attack due to the fact that silica is not soluble in acid.

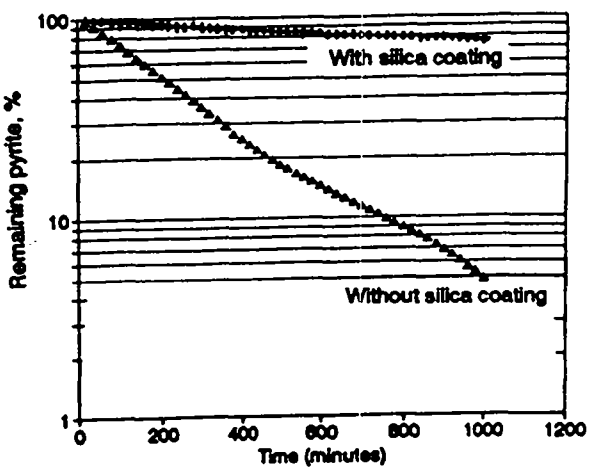

Figure 3. Pyrite leached with 0.145 *M* H_2O_2, with and without 50 mg L^{-1} silica (Si) (having as source sodium meta-silicate (Na_2SiO_3 $5H_2O$) at pH 5 buffered with 0.01 *M* sodium acetate at room temperature (Zhang and Evangelou, Unpublished data, 1994).

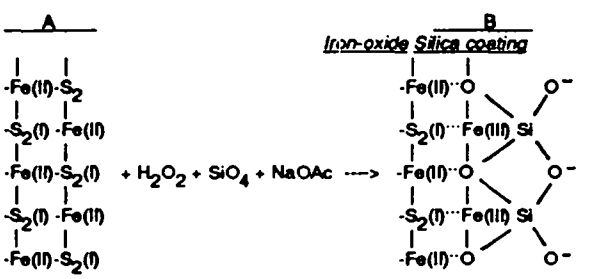

Figure 4. Schematic of H_2O_2 induced oxidation proof silica surface coating on iron sulfides (Evangelou, 1996, U.S. Patent No. 5,494,703, Feb. 27, 1996).

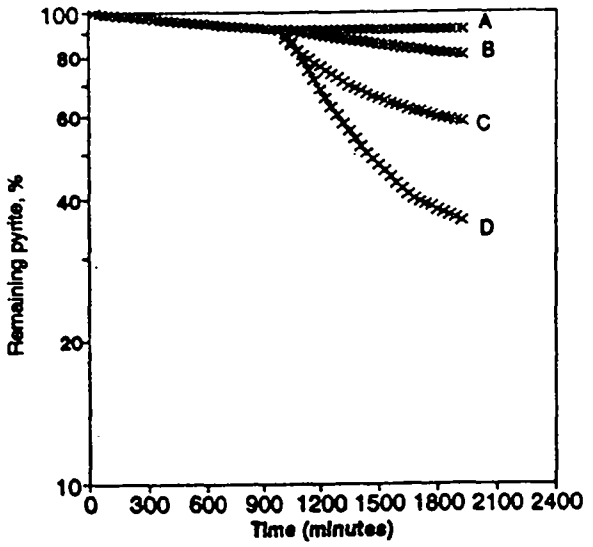

Figure 5. Various pyrite samples first (up to 900 min) leached with 0.145 *M* H_2O_2, 50 mg L^{-1} silica (Si) at pH 5 buffered with 0.01 *M* sodium acetate at room temperature. Each of the pyrite coated samples was treated as follows: Sample A was leached with water; sample B was leached with 0.145 *M* H_2O_2; sample C was first leached with 50 ml 4 *M* hydrochloric acid (HCl) and then leached with 0.145 *M* H_2O_2; sample D was first leached with 50 ml 4 *M* hydrofluoric acid (HF) and then leached with 0.145 *M* H_2O_2 (Zhang and Evangelou, Unpublished Data 1994).

Successful application of such coating methodologies in the field could mean long term solution (perhaps even permanent solution) to certain types of acid mine drainage problems. These coating methodologies are expected to be cost effective since they involve readily available materials and only cover the surface of pyrite particles. Furthermore, the coating solution could be applied to any permeable coal mine waste thus, little or no physical disturbance of coal mine waste during treatment would be necessary.

Outdoor Leaching Columns Experiments

To date, pH of the leachate samples collected from the limestone treatment is slightly lower than that from phosphate and silica treatments (Fig. 6). On the average, pH of the leachate samples from limestone, phosphate-coating, and silica-coating treatments are 6.75, 6.88, and 7.00, respectively. The lower pH of the limestone treatment may be due to higher pyrite oxidation from mine tailing as indicated by the higher Fe concentration of the leachate samples (Fig. 7). The average iron concentration of the leachate samples from limestone treatment is 148 while those from phosphate and silica-coating treatments are 11 and 8 mg L^{-1}, respectively. However, the sulfate concentration from all of the treatment seems to be similar (Fig. 8). We expect that during long-term column exposure to atmospheric conditions, leachate composition will differ between limestone treated column and coated columns if indeed coating has taken effect by the procedures employed.

As expected, the introduction of pyrite oxidizing bacteria (*Thiobacillus ferrooxidans*) temporary increased the rate of Fe (Fig. 7) and SO_4-S (Fig. 8) production in all of the treatments. However, introduction of pyrite oxidixing bacteria has more significant affect on the Fe production from the limestone treatment than phosphate and silica-coating treatments suggesting that, up to now, pyrite coating is protecting microbial pyrite oxidation. The fluctuation of all chemical concentrations of the leachate samples from sampling to sampling apparently is affected by the rainfall pattern through flusing and/or dilution effects.

The results of these outdoor column leaching experiments show that application of the coating technology seems to give new promise in the abatement of acid mine drainage. However, long term meticulous monitoring and evaluation are still needed. We will be monitoring these columns for several years.

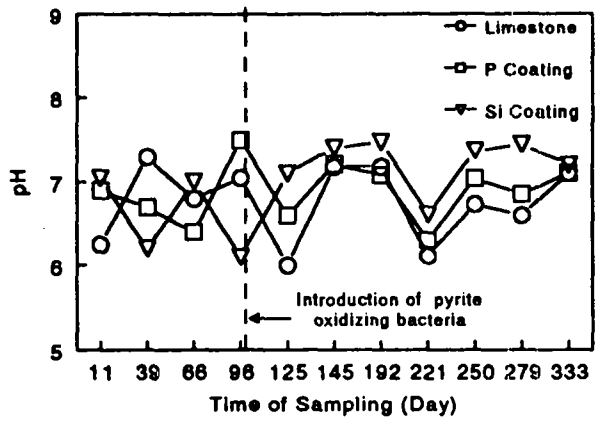

Figure 6. Leachate pH from outdoor columns taken at the ten sampling (unpublished data).

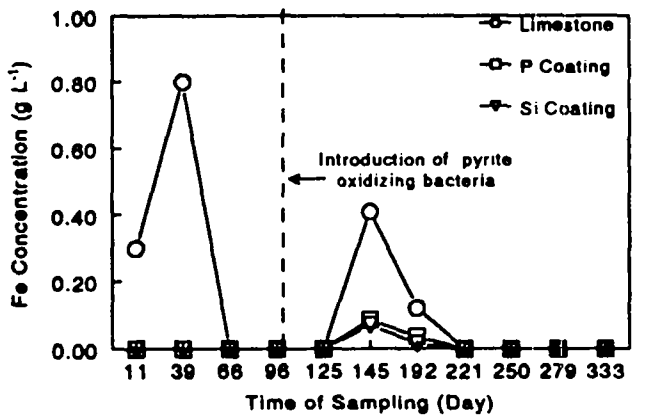

Figure 7. Iron (Fe) concentration of the leachate samples from outdoor columns taken at the ten sampling (unpublished data).

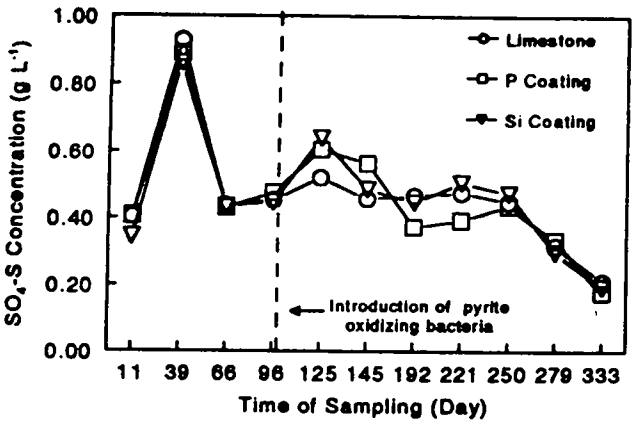

Figure 8. Sulfate (SO_4-S) concentration of the leachate samples from outdoor columns taken at the ten sampling (unpublished data).

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Disclaimer

The views and conclusions contained in this document are those of the authors and should not be interpreted as necessarily representing the official policies or recommendations of the U.S. Bureau of Mines, U.S. Geological Survey, and U.S. Department of Energy.

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LIMESTONE ADDITIONS TO AFFECT CHANGES IN LOADING TO REMEDIATE ACID MINE DRAINAGE[1]

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Gwendelyn Geidel[2]

Abstract. All things being equal, the acid potential of pyritiferrous rock naturally outstrips the rate of alkaline production due to the differences in solubilities and rates of pyrite reaction and bicarbonate dissolution. Thus, mixtures of acid and alkaline waters produced by the two chemical systems usually result in net acid conditions when the blending of these two water types takes place. The study area watershed was indirectly impacted by mineral mining when pyrite-rich tailings were transported downstream and deposited. The acidity produced by the tailings deposits varied from 50 to 200 mg/l and, based on detailed field traverses, the area of the deposits was approximately 1672 m^2. The strategy was to generate at least four times more flow through alkaline material than the acid horizons, thereby, adjusting alkaline loads to neutralize the acid drainage. A blanket application of imported limestone, 8 cm thick and covering approximately 6690 m^2, was placed during June and July, 1995, near the headwaters and removed from the acid producing areas. To date we have seen the acidity levels reduced to about 45 mg/l. As designed, the critical placement of the limestone blanket is intended to affect the quality of 1) run-off, 2) near surface interflow and 3) groundwater. The effects of acid rain, coupled with the time necessary for impacts to take place on the groundwater and vadose zones, suggests that at this time, only the run-off component is reporting to the stream and that more time is needed for the flow path to become sufficiently alkaline and effect the drainage quality.

Additional key words: acid mine drainage

Introduction

From a geochemical standpoint, the acid potential of pyritiferrous rock strata is not constrained as the rate of alkaline production due to differences in their solubilities. The acid potential is related to several factors and

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may vary from 100-5000 mg/l acidity as $CaCO_3$, while the maximum solubility of limestone, under ideal conditions, is limited to about 400 mg/l alkalinity as $CaCO_3$ under maximum pCO_2 ($pCO_2=10^{-1}$). Thus, because of the disparity, mixtures of acid and alkaline waters, all things being equal, often results in net acid conditions. In order to provide a suitable balance, acid and alkaline loads rather than concentrations should be used in determining the net acid potential.

In previous studies in two coal mines in Ohio and West Virginia, limestone was applied as a surface veneer to attempt to remediate the acid ground/spoil waters. In Ohio, coarse limestone was applied at a rate of 16 to 26 metric tons/hectare with minor effects. At this site, the backfill consisted of porous, acidic sandstone rubble and seeps had acidities varying from 75 to 5000 mg/l. As a result, with the exception of one sub-basin with low levels of acidity in which the sub-basin substantially decreased its acid load, the levels of alkalinity generated were too low to have a direct effect on the more highly acidic sub-basins of the study. While an overall decrease in the acidity of the system occurred with time, the decrease, with the exception of the sub-basin noted above, was similar to that of the control basins indicating that a natural decrease in acid loads was occurring. (Geidel and Caruccio, 1982).

Similarly at the Mercer site in West Virginia, the backfill was comprised of a porous shale/coal sequence and the application of limestone was applied at a rate of 3 to 15 cm thick (up to approximately 110 metric tons/ha) and produced no significant improvement in seep discharge quality (Caruccio, Geidel, and Williams, 1985). Our studies to date have shown that surface veneers of limestone applied on the surface of acid producing materials, do not generate significant amounts of alkalinity (as bicarbonate) and cannot be used to remediate moderate to high acid groundwater. Though it was shown that the amount of alkalinity generated under atmospheric conditions (60 mg/l) was not sufficient to impact the acidity in these high acid areas, there was merit, however, in using this technique at sites where acid loads were low. This study was designed to test this possibility and determine if this treatment technique had any merit.

Background

Within the watershed containing the current study area, less than 25 percent of the watershed was impacted by mining. On occasion, breaks in the tailings conveyance pipes caused pyritiferrous tailings to spill downstream and through the action of the meandering stream, were deposited as point bar, pyrite-rich tailings. At this mine site the acid producing zones have been identified as discrete deposits within the stream and their inaccessibility by heavy equipment to remove the deposits, precluded the removal of the pyrite-rich tailings. The acidity produced by the tailings, varied from 50 to 200 mg/l.

The area has been mined intermittently during the last 40

years and the last operation was completed approximately ten years ago. In the processing of the ore, fine grained tailings were generated and transported, via pipe, to tailings ponds. However, during the course of the years of operation, some tailings were deposited, presumably as the result of a spill, in the headwaters of an intermittently flowing stream. The tailings deposited are comprised of fine grained quartz, clay and minor amounts of pyrite (on the order of about 2%).

The quality of the stream flowing through the study area is mildly acidic (around 200 mg/l) which most likely results from the oxidation and leaching of the tailings from within the stream bed and, in some cases, from the stream banks. The distance from the headwaters to the sampling point (Sampling Station 1) is approximately 215 meters. As the stream flows through the area, the flow occurs intermittently at the surface, with segments of the stream showing signs of overland flow and irregular acid loads only during high run-off precipitation events. That high flows have taken place is evidenced by the surface expressions.

The primary sampling point is located approximately 215 m downstream from where the stream first appears as a defined channel. This location was selected for a sampling point because continuous flow has been noted at this site and for approximately 55 meters above this point, the stream flow is sub-surface. During the few times when the stream was sampled, the flow varied from a low of approximately 0.3 liters/sec to a very high flow during summer high intensity, short duration rainfall events. This sampling point (SS1) had a continuous flow and, therefore, was selected as our sampling point.

Methodology

In March 1995, a field survey determined the extent and occurrence of the tailings within the stream bed. Over the 215 m of stream, tailings were visible in about 96 m of stream bed and, while the remainder of the stream appeared to flow beneath the surface, it also contained tailings.

The surface area occupied by the zone of tailings or the potentially acidic material was determined to be 1672 m^2 based on field mapping of the site. Five samples were collected from various locations within the zone. The samples were analyzed for paste pH and the specific conductance of the resultant decanted leachate was measured. The results of these analyses showed paste pH values ranging from 3.85 to 2.55 and specific conductance values between 85 and 1030 μS with the lowest pH and highest specific conductance samples from mid-stream sample points, as opposed to stream overflow or dry stream bed samples.

The stream, at Sampling Station 1 (SS1), has been sampled since November 1993 and provides some background data. The background acidity values vary between 45 and 205 mg/l acidity and the pH values vary between 3.75 and 3.8. The specific conductivity data show a variation between 187 to 300 μS

with the lower value corresponding to the lower acidity value. The stream characteristics reflect the overall nature of the drainage basin and, as expected, average the variation in quality seen in individual soil samples. Using stream data, in conjunction with the rock sample analyses, the average acidity was established to be 200 mg/l over the area of influence.

Using the 200 mg/l acidity value, 200 mg/l of alkalinity which must be generated to neutralize the acidity. Limestone, by its nature and chemistry, can only produce approximately 60 mg/l alkalinity under normal atmospheric conditions of $pCO_2 = 10^{-3.5}$. Therefore, using a mass balance approach, there must be approximately three and one half times more alkalinity than acidity made available to the system. To neutralize each liter of acid water with an equivalent amount of alkalinity, and given that the maximum amount of alkalinity from limestone is 60 mg/l, increased "loads" of alkalinity are required such that the flow of the total system is at least 3.5 times greater in the alkaline zones than from the acid prone zones.

In a drainage basin where the zone of influence by the acidic material is 1672 m^2, acid rain contacting this material will produce 200 mg/l acidity for the 1672 m^2. To offset this acidity, rainwater must contact approximately 3.5 times more limestone and achieve equilibrium conditions so as to offset the acidity during a given rainfall event. Therefore, as a conservative measure, four times the area of the acid zone was determined to be necessary; 6688 m^2. In addition to the area of impact, the limestone must reach equilibrium conditions. Based on previous studies by Geidel, 1980 and Neuhaus, 1986, it was shown that a layer of coarse limestone (1 inch down to #30) which was 8 cm thick could provide the appropriate amount of alkalinity and at the same time not wash away during normal rainfall events.

The area scheduled for the limestone application was below the previously disturbed and reclaimed area of the mining operation and was within a southern hardwood forest. In consideration of minimizing land disturbance, the limestone was placed on the hillslopes with a minimum disturbance to the trees and shrubs. These areas with the alkaline addition will provide the alkalinity component and would take part in a balance against the acid production potential. Thus, the limestone (being removed from the acid producing material) would not become coated and should sustain an alkaline discharge. The areas for limestone application were staked out accordingly. During the first phase of the application, 2415 m^2 were delineated and limestone was stockpiled outside the area and brought to the various sites by a small front-end loader. The limestone was then manually spread in a layer 3 inches thick. Based on the ease of the first application, the second phase boundaries were established and the limestone application continued until the 6690 m^2 were covered with limestone. The total amount of limestone applied during June and July, 1995, was

972 metric tons.

Results to Date and Future Plans

The results of the monitoring of SS1 (Figure 1) indicate that while the acidity levels have been consistently low since August 1995, the acidity and specific conductance were also low in November 1993, suggesting that the impact, at this time, has not been significant. Monitoring, however, will continue on a monthly basis for at least the next year and we anticipate that a continued improvement in water quality will occur.

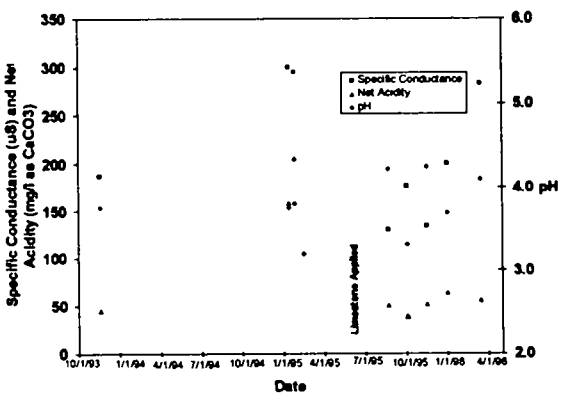

Figure 1. Net Acidity, pH and Specific Conductance for SS1.

There are several possible explanations for the short term (four month) lack of identifiable changes in the water quality. First, the rainfall in the area has been monitored during the past two years. The precipitation is acidic with pH values ranging between 3.3 and 4.2, specific conductance between 5 and 40 μS and net acidities between 7 and 16 mg/l as $CaCO_3$. The limestone, therefore, utilizes between 10 and 20 percent of its available alkalinity to neutralize the acidity of the rainfall. In addition, the clay rich soil over which the limestone was placed constitutes an acid reservoir which must be neutralized by the alkalinity generated by the limestone. Until the reservoir has been neutralized, the induced potential excess alkalinity generated by the limestone will not be available for neutralization of the acidity being generated by the tailings.

Secondly, during low intensity rainfall events, it has been observed that most of the rainfall infiltrates into the limestone and that there is little surface flow. Some surface flow is evident from the minor amounts of limestone that have washed or been transported from the application site. However, the wash has been minimal and in most areas, there appears to have been no movement of limestone. The significance of the infiltration is that the alkaline front is moving into the vadose zone via the rainfall and chemically interacting with the stream as a ground water discharge rather than as a surface water discharge. Long term, this is the preferred scenario because as the alkalinity increases in the groundwater, the alkalinity will become a portion of the base flow. This is important during low flow conditions because the acid components will continue to be affected even without a precipitation event.

The third reason for the lack of significant improvement in the water quality relates to the amount of alkalinity generated by the three inch layer of limestone. We are in the

process of installing a shallow monitoring network to evaluate the alkalinity immediately below the limestone layer and to evaluate the acid and alkaline potentials at shallow depths within the soil horizon. Although the studies by Neuhaus (1986) indicated that the limestone quality was not a significant factor related to equilibrium conditions, the field monitoring will indicate whether the flow-through time of the rainfall under field, rather than laboratory, conditions generates maximum alkalinity. The network will also provide information on any increases in alkalinity which may result from the increasing pCO^2 as a result of surficial leaf decay.

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SESSION 12: BYPRODUCTS AS SOIL AMENDMENTS

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SOIL DEVELOPMENT IN A COAL WASTE, SLAG AND SLUDGE ARTIFICIAL TOPSOIL MIX

by

Justine A. Cox, Rob J. Whelan & Stuart C. Thompson

Abstract. Waste disposal has become a major concern for industries that create a large amount of by-products due to the cost of landfill and environmental responsibility. Beneficial re-use of these by-products which contain resources for other industries, will solve some of these problems. An artificial soil mix has been created out of ironmaking and sewage wastes, for use in a revegetation program to improve the BHP Steelworks site in Australia. Coal washery refuse, blast furnace slag and sewage sludge have been combined to form a topsoil which is placed on waste mounds and revegetated. Field trials were established to evaluate the soil forming characteristics of this artificial media. The fine fraction (<2mm) of the particle size distribution decreased from 71% to 64% in one year, showing evidence of weathering and particle breakdown. Bulk density remained the same (1.0 g/cm^3) in a two year comparison. The pH of the soil mix declined from 7.6 to 7.1 in one year. Native tree growth in the soil mix was very good, although comparisons could not be made with real soil . The long term viability of the soil mix in relation to nutrient cycling could not be extrapolated with only two years of data. The potential is promising for the use of this soil substitute in urban landscaping, land restoration and revegetation, and opportunities for further research are available.

Introduction

Beneficial Waste Re-use

Industrial waste material has been typically dumped, used as landfill or otherwise disposed of and generally regarded as a problem, but there is an increasing shift towards the beneficial use of these by-products. Waste management strategies currently seek to minimise costs of disposal by the consideration of other methods of use besides dumping. Recycling by-product materials has helped improve the environment by reducing the amount going to landfill, but these methods still regard industrial by-products as wastes. The majority of by-products should be viewed as potential resources. Many 'waste' materials possess beneficial attributes for alternate industries or processes, which should then be fully utilised (see Joost *et al.* 1987, Logan & Harrison 1995). The physical and chemical characterisation of by-products enable these factors to be uncovered.

BHP (the Broken Hill Proprietary Company Limited) in Wollongong, on the east coast of Australia, has an integrated steelworks producing 5 million tonnes of steel a year. Along with this, approximately 2.5 million tonnes of solid wastes are produced, mainly coal washery refuse (CWR), blast furnace slag (BFS) and basic oxygen steel (BOS) slag. Some slag is re-used for sand, cement and road production,

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and coal wash has been used as structural fill in vast quantities in the local area (Thompson & Makin 1990). Waste emplacements of these by-products on site has occurred over many years with several off site locations also utilised. This has meant space for future material disposal has become increasingly scarce.

Revegetation of Waste Emplacements

Due to the urban location, environmental awareness and the company's decision to improve the conditions of the 800 ha BHP site, research into the revegetation of waste emplacements was initiated. Waste stabilisation and site rehabilitation methods of topsoiling and seeding with grass or cover crop were not undertaken. This was because the aim of the large scale project was to establish native vegetation in the waste material, and to eventually create a sustainable "urban forest" (Thompson & Makin 1990). There is no soil on site and very large areas are involved, so the cost and environmental impact of transporting native soil to the site was considered to be excessive. A soil substitute was required to provide the nutrients and growing conditions necessary for plant growth and survival. The abundance and availability of coal washery refuse and slag provided an opportunity to utilise waste products which contained elements beneficial for this purpose.

The BHP Soil Mix

While the major steelworks wastes provide an adequate physical structure for the basis of an artificial soil medium, there is a lack of organic matter, essential nutrients and micro-organism activity. Sewage sludge (biosolids) is a waste product of the sewage treatment system, and seems capable of providing these requirements. It is high in organic matter, contains organic nitrogen (which becomes available slowly), phosphorous and provides a suite of decomposer bacteria and fungi. An artificial growth medium has been created out of coal washery refuse, granulated blast furnace slag and sewage sludge in a ratio of 2:1:1. The materials are blended together with a front end loader to form a substitute topsoil, and stockpiled for various times, depending on the rate of utilisation. Several other wastes have been examined for growth potential, such as fly-ash, foundry sand and BOS slag and dust (Whelan *et al.* 1993) but are not used in the soil mix.

Constituents

The characterisation of this artificial growth medium can be estimated using soil analyses. However, as the material is man-made, unique and not a "real" soil it may be difficult to interpret some results. The properties of each component will also aid in understanding the soil's potential.

Coal washery refuse is the by-product of washing coal from underground mines in the local area for coke making. The majority of the shale and clay minerals are quartz and kaolinite. In addition, the material contains residual coal particles. The black waste is dewatered by belt press to 2-20 mm size fragments and is generally quite alkaline (Table I). The pyrite content is very low in this type of coal.

Blast furnace slag is the by-product of the iron-making process and is created at extremely high temperatures. It is granulated by a high pressure jet of water to a texture similar to coarse silica sand, and consists of calcium and aluminium silicates (especially wollastonite and mullite) (Thompson & Makin 1990). The size fraction is generally 0.2-2 mm and is also quite alkaline (Table I).

The sewage sludge obtained from several Wollongong sewage treatment plants is anaerobically digested after sedimentation (Table I). It's dewatered by either centrifuge or belt press. As well as contributing organic nutrients, there are also some heavy metals at low concentrations. The soil mix project uses all of the local area's sludge, and the volume of soil mix produced is constrained by this supply.

Table I. Physical and chemical characteristics of the constituents of the soil mix (from Whelan and Liangzhong 1991).

	particle size distribution (mm)	pH (H_2O) 1:1 w/v	Ec 1:5 w/v	N (%)
coal wash refuse	0.2 - 20	9.4	0.15	
blast furnace slag	0.2 - 2	9.2	0.21	
sewage sludge		6.4	6.19	5.24

Soil Mix Characteristics

The soil mix is a black, granular, relatively homogeneous mixture with an initially alkaline pH due to the slag and coal wash. The use of salt water to granulate the BFS has meant that some soils made in 1994 are reasonably sodic (Thompson & Makin 1990). Also, due to the granular nature of the mix, the water infiltration rate is very high. Given these attributes from the constituents analyses, no amelioration was attempted. Instead, native species tolerant of these factors were initially chosen. The plant response to this growing medium was exceptional. Many types of native trees, shrubs and ground covers grew quickly even though site conditions are often harsh. The adequate rainfall of the region (approx. 1000 mm/yr with a summer and winter maxima) could also contribute to this effect.

On-Site Revegetation of Waste Mounds

There are currently many types of revegetated areas around the industrial site, with more than 300,000 native plants growing in the artificial soil. A typical garden is comprised of a mound of waste coal washery refuse which ranges considerably in size, from small strips for car parks to boundary, screening mounds sometimes exceeding 1km in length. Soil mix is placed on top of the coal wash base (usually to a depth of 150mm), and bark chip type mulch added. Tree, shrub and ground cover seedlings from local nurseries are planted by hand and watered until established. Over the past eight years 20% of the steelworks site has been revegetated, using over one hundred Australian and exotic plant species growing in the soil mix. Detailed records and site history are kept for future reference. Weed control is carried out by mulching, using woodwaste or woven plastic matting, and herbicides. Lawn is grown in several areas where trees/shrubs are unacceptable (e.g. around heavy industry).

Soil Mix Development

The aim of the greening program was to create a self-sustaining forested ecosystem which required no maintenance, such as nutrient input, weed control or soil additives. It was also to create a value-added soil mix product from wastes for eventual use in the urban landscaping industry. Restoration of degraded sites, revegetation projects and other land rehabilitation efforts may also benefit from this type of media in the future. The long term aim of the soil mix project is for the artificial growing medium to function as a true soil whereby soil forming processes occur. Bradshaw (1987) recognised the importance of the soil substrate as the basis for successful revegetation and restoration attempts. Once physical, chemical and biological barriers to plant growth are ameliorated, there is

a much better (and quicker) chance for ecosystem development.

To create a self-sustaining ecosystem it is necessary once the plants (especially legumes) are established, for the litter layer to build up, decomposer organisms to be active and nutrient cycling to begin to operate. For the long term viability of the soil mix it is useful to assess the changes in the physical, chemical and biological characteristics over time which relate to soil development. Due to the nature of the material and the revegetation methods used, it is difficult to predict *a priori*, the rate and direction of these changes.

The factors that influence soil formation are: parent material, climate, slope (topography), organisms and time (Jenny 1941, Crocker 1952, Paton 1978). These universal forces would also act on the soil mix, changing its physical structure, chemical environment and biological activity with time. Many studies on the initial stages of soil development have identified important parameters of soil formation. These include particle weathering and horizon formation (Roberts *et al.* 1988), increases in nitrogen (especially), carbon, Cation Exchange Capacity and decreases in bulk density (Anderson 1977, Varela *et al.* 1993) and increased root biomass and microbial activity (Schafer *et al.* 1979). Revegetation of mine spoil with sewage sludge has shown that the organic source of nitrogen and phosphorous has dramatically improved the establishment of plants, due to the nutrients and the improvement in soil structure it provides (Topper & Sabey 1986, Seaker & Sopper 1988, Wong & Ho 1994).

The present study evaluates some physical, chemical and biological changes of the artificial soil mix in relation to soil formation over time. Several parameters were chosen to fully characterise the initial soil state and then to monitor the development of the mix in a field trial on a site at BHP Flat Products Division, Pt. Kembla for two years. Native tree/shrub species were used as indicator plants.

Methods

Field Trial Description

A very large coal washery refuse emplacement was the base of the trial site at the BHP Steelworks. This level area measuring approximately 60m x 20m, was situated on top of a hill overlooking a blast furnace. Twelve small plots (4m x 2m) were created for the control (no plants), oats and lucerne treatment (three replicates each) with four large plots (10m x 5m) created as the plots for native species. Each plot was lined with plastic sheeting to a depth of 1m of coal wash. The plastic surface sloped to direct leachate water into a sump of several 44 gallon drums welded together. Once backfilled and flattened, each coal wash surface was topsoiled with 300mm of BHP Soil Mix.

The four 'native' plots were planted with *Eucalyptus maculata* (Myrtaceae), *Acacia floribunda* (Mimosaceae) and *Callistemon salignus* (Myrtaceae). On each plot, fifteen individuals of each species were hand-planted (total 45). An irrigation system was established to water twice a week (sometimes three times in the first Summer), but this was stopped after the first year.

Soil Physical Parameters

Soil samples were taken approximately every three months for physical analysis. With a screw auger 20 cm long, ten random samples per plot were taken, and composited. Samples were spread thinly on newspaper then air dried for at least 48 hours. Pretreatment entailed light crushing to disperse the aggregates. Particle size distribution was measured by sieving the "soil" into fractions of: >16mm, 16-8, 8-4, 4-2, 2-1, 1-0.5, 0.5-0.25, 0.25-0.125, <0.125mm using a mechanical shaker. Not all samples were analysed, so only data collected over one year for the significant <2mm fraction is presented. Bulk density was determined at the end of the experiment for comparison with values examined by Whelan & Liangzhong (1991). This was

measured by the core method, where a known volume (29.5 cm^3) of soil was sampled, dried and weighed. Ten replicates were sampled from each plot.

Soil Chemical Parameters

Soil was sampled as before, with ten random auger samples composited and sub-sampled for individual analyses. Available nitrogen (nitrate/nitrite and ammonium) and total phosphorous concentration were measured three monthly for twelve months for all plots. The leachate water from each plot was also sampled after six and twelve months, for nitrate, ammonia and phosphate concentration. Each water sample was comprised of five aliquots from individual samples. The pH of the soil mix was measured at the beginning of the experiment and after one year. Ten replicate individual soil samples per plot were air dried, sieved and the <2mm fraction used in a 1:5 soil : $CaCl_2$ solution. Composite samples from the soil mix were also analysed for ten heavy metals (Pb, Cd, Zn, Cr, Cu, Ni, Mo, As, Hg and Se) at the beginning of the experiment.

Soil Biological Parameters

Native plant growth was monitored monthly for a year, then three monthly for another year. Tree height, and diameter of the stem at 20 cm above the ground were measured. This bioassay of the soil mix provides an integrated analysis of the physical structure (relating to rooting environment and water use), nutrient availability, potential toxicities and microbiological decomposer activity. Two pathogen indicators were also measured; enumeration of fecal coliforms and the presence of *Salmonella*.

Results

In the particle size analysis, the initial mean proportion of < 2mm fraction in the soil mix was 71% (± 0.7% SE) and had significantly declined to 64% (± 0.7%) over one year (Fig 1). This meant that the soil mix had

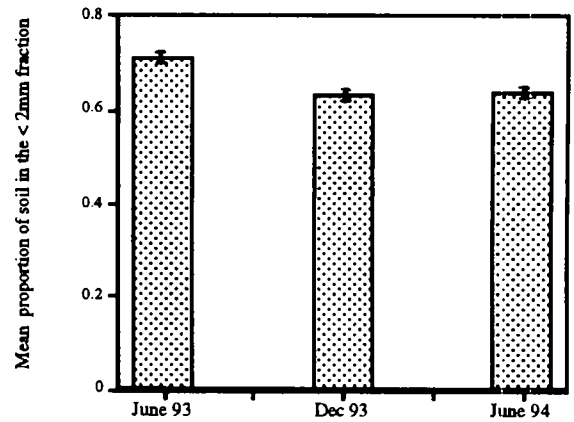

Figure 1. Mean proportion of soil in the <2mm fraction (after sieving) over one year. Standard error is shown.

undergone particle weathering and breakdown, and / or movement down the profile. This breakdown occurred quite rapidly after exposure to the environment, and the subsequent movement of particles has many consequences related to the leaching of ions into deeper layers. The bulk density of the soil mix initially was 1.0 g/cm^3, and after two years it had not changed with a mean value of 1.0 g/cm^3 (± 0.03).

The available nitrogen in the soil mix was initially high due to the sewage sludge, with a mean nitrate/nitrite content of 169 mg/kg (± 44) and ammonia of 50mg/kg (± 13). These values rapidly declined to a very low level in three months and remained there for another nine months (Fig 2). Total phosphorous concentration in the soil mix at the beginning of the experiment was variable with a mean of 740 mg/kg (± 115). Unfortunately there were some doubts about validity in all subsequent analyses, and so they are not reported. The nutrients in the leachate water were low. At six months, the mean nitrate concentration was 25.0 ug/ml (± 5.6), the phosphate

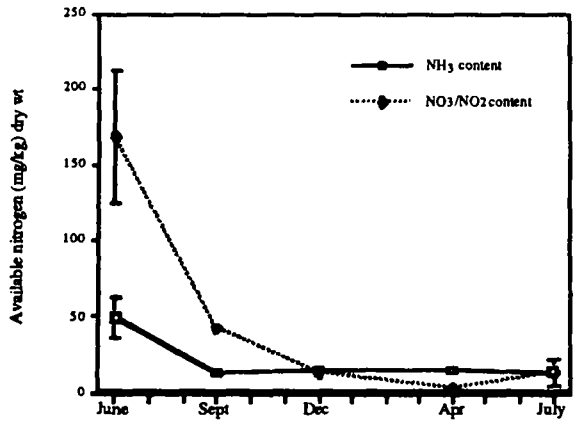

Figure 2. Mean available nitrogen (nitrate/nitrite and ammonia-N) content of the soil mix over one year (mg/kg dry weight).

concentration all <1mg/ml, and the ammonia concentration all <0.1 mg/L. After twelve months the mean nitrate concentration was a very low 0.95 mg/ml (± 0.23) with the phosphate and ammonia concentrations the same as before.

The initial mean pH ($CaCl_2$) of the soil material was 7.59 (± 0.04), and dropped to 7.10 (± 0.02) after one year. This meant that some acidification process had occurred. The heavy metal concentration of the soil mix are all quite low with mean values given in Table II. New draft guidelines on heavy metal concentrations of all biosolids products in Australia are presented for comparison (EPA 1995).

Plant growth was exceptionally good over the extent of the study duration. The native species grew rapidly after establishment and continued to develop. The *Eucalyptus* showed the most accelerated growth, and the *Acacia* also performed well (Fig 3). The *Callistemon* plants displayed relatively less growth which may be due to the advanced stages of the plants when the experiment began. Several plants of all three species died, with *Callistemon* most affected. Over the two years, weed invasion increased, but did not suppress native tree growth. This was mainly due to the delay in weed establishment, their annual life cycle and the large size of the trees when they eventually took hold. Insect attack was also noted. A number of *Acacia* plants flowered in the first spring, with most flowering in the second.

Table II. Heavy metal concentrations (mean and standard error) of the soil mix at the start of the field experiment (n=3), with the maximum metal concentration in any biosolids product from EPA Draft Environmental Management Guidelines (EPA 1995).

Heavy metal	Mean concentration in the initial soil mix (mg/kg)	Standard error	Max. conc. in unrestricted biosolids product (mg/kg)
Pb	4.77	0.19	150
Cd	1.30	0.06	3
Zn	31.0	1.20	200
Cr	7.60	0.50	100
Cu	24.67	2.33	100
Ni	5.50	0.21	60
Mo	2.77	0.32	-
As	0.22	0.12	20
Hg	0.99	0.13	1
Se	0.30	0.12	5

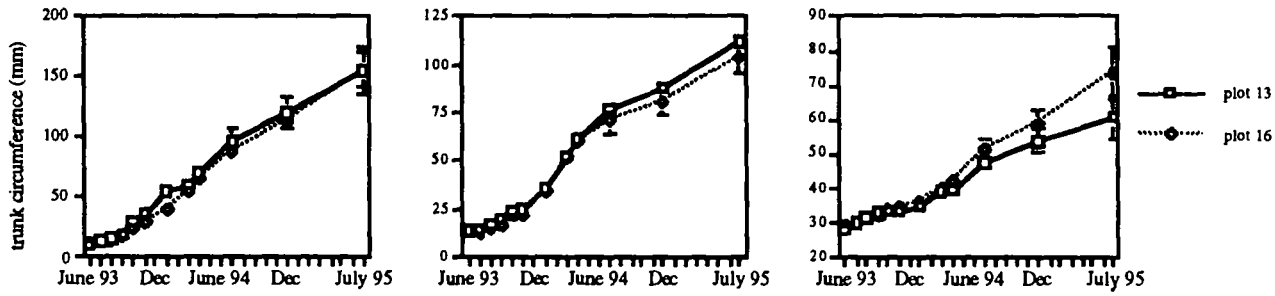

Figure 3. Native tree trunk circumference (mm) at 200mm above ground level for *Eucalyptus maculata*, *Acacia floribunda* and *Callistemon salignus*, over 2 years. Plot 13 and 16 are the 'native' plots.

There were never any *Salmonella* organisms detected in the soil mix in six months of monitoring. At the beginning of the experiment there was a mean of 220 CFU/g for fecal coliforms in the soil mix. After a month there were consistently <10 CFU/g, which remained at this level for six months. The new EPA guidelines for biosolids products have a limit of <1000 MPN per gram for fecal coliforms and no *Salmonella* sp detected in 50g, from a composite sample of five individual grab samples (EPA 1995)

Discussion

Physical soil development

The physical environment of the soil mix seems capable of supporting plant growth, by providing little restriction to root penetration, a range of particle sizes and adequate supply of available water. The results of this experiment suggest that weathering is occurring as there is a significant shift in the particle size distribution. The finer fraction of the soil mix has decreased over time which means that some larger particles have broken down to smaller components and moved down the profile. Coal washery refuse is the most likely component for this to occur as the clay minerals are readily broken down. Further weathering of the other components would also contribute to this increase. It was surprising to find the bulk density remaining the same after two years. An improvement in soil structure would have resulted from an increased value, as the loosely packed material becomes more aggregated. It is possible that the pozzolanic character of the soil mix has decreased with time which would counteract this trend. The sampling equipment may have influenced the accuracy, as the diameter of the corer was 20mm, and perhaps too small for this type of gravelly media.

Available nutrients

The nutrients available were provided by the conversion of the organic N and P source into readily available nitrate and phosphate. Plants did not show any signs of nutrient deficiencies, but clear indications of these have not been reported for the native species used. The initial flush of nitrate from the biosolids did not stay high for long. When the layer of soil mix was applied organic nutrients were exposed to oxygen and water. The aerobic organisms would be able to initiate their biochemical transformations under these optimal conditions (Alexander 1977).

The plants rapidly used the available nutrients, and any excess would have leached down the profile. As very little nitrate and virtually no phosphate or ammonia were detected in the leachate waters, we can assume that most of the available nutrients in the soil were used. This has great benefits, as the potential pollution problem from excess nutrients in the leachate may be minimised if there are enough plants to utilise the resource. The nitrogen cycle (transformations, additions and losses of N) of a community is of vital importance, and the establishment of one is often responsible for disturbed or degraded ecosystem recovery (Cundell 1977, Tate 1985, Sorensen & Fresquez 1991).

The pH of the soil mix decreased over one year which indicates that acidification processes had occurred. There are several ways in which this can happen, for example organic matter oxidation and leaching of carbonate materials. This has relevance to the increased availability of some metals to plants at lower pH values.

Plant growth

Plant growth and condition is a good indicator of soil fertility and potential toxicities. As the growth of the native species on this soil media was considered very good it was a positive sign for the short term success the soil mix. Of course these plants do not represent the vast range of conditions required for all growth, but observations in the established gardens on site suggest that limitations are not common. As the trees grew, shading / competition may have affected the growth rate of some trees, which would increase the variance within a plot. Insect attack may also have influenced relative growth rate. Several types of soil invertebrates were observed but not identified of quantified, these include springtails, Diplurans, Isopods, many ants and earthworms.

Comparisons of soil development

As there is no other artificial soil mix similar to this one, comparisons of rate and direction of change with other systems must be accomplished with caution. Some studies of soil genesis on mine spoil encounter similar conditions; for example, an initial homogeneous, gravelly mix, may be analogous. In these systems, in contrast to the artificial soil studied here, a lack of organic matter and nitrogen, severely limited plant establishment and development of a functioning ecosystem (Stroo & Jencks 1982, Sorensen & Fresquez 1991).

The revegetation attempts of coal mine spoil with sewage sludge in coal mining areas of the US, provides a more realistic comparison for soil formation. Joost *et al.* (1987) conducted a study on revegetation and minesoil development of coal waste, amended with sewage sludge and limestone. They found that after two years there was an improvement in soil structure, a reduction in bulk density and an increase in water holding capacity. Similar results may have occurred for the BHP soil mix had the same parameters been measured.

Another plant growth medium made of wastes, N-Viro Soil, is used in USA, Australia and England as a lime substitute, soil amendment and soil substitute. This material is composed out of dewatered sewage sludge and alkaline industrial by-products, such as lime kiln dust and cement kiln dust (Logan & Harrison 1995). There are many similarities with this media and the BHP soil mix, although the gravelly coal washery refuse would determine the difference in the water holding capacity, bulk density and other physical characteristics. The success of the N-Viro Soil in improving existing soil qualities over two years is recognised, and it would be interesting to determine the long term viability of the material.

Long term viability of the BHP soil mix can not be estimated by extrapolation from a two year study. A survey will be undertaken on site in the future of the many gardens that have been constructed in the last eight years. Many physical, chemical and

biological parameters will be measured to quantify trends in soil development over time. The biota of the soil system, i.e. plants, roots, microorganisms and invertebrates will be the focus of this study, due to their importance in nutrient cycling and ecosystem functioning in a new and developing native vegetation community.

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ACIDIC MINESPOIL RECLAMATION WITH AFBC BY-PRODUCT AND YARD-WASTE COMPOST[1]

by

R.C. Stehouwer[2], W.A. Dick, and R. Lal

Abstract. Economic and environmental incentives to reduce solid waste volumes have spurred interest in the development of beneficial uses for urban and industrial by-products. This project investigated the reclamation efficacy and impacts on soil and water quality of two such materials: atmospheric fluidized bed combustion (AFBC) by-product and yard-waste compost. Six 1-acre watersheds were constructed on acidic abandoned mined land spoil (pH range 3.5 to 4.5). Two watersheds each were then reclaimed with 8 in of borrow soil, 125 tons/acre of AFBC, or 125 tons/acre of AFBC and 50 tons/acre of compost, and planted with a grass-legume seed mix. Watersheds were instrumented to record hydrographs of storm-water runoff events, measure erosion, and collect samples of surface- and percolate-water flow. One year after reclamation the AFBC and AFBC+compost treatments compared favorably with the traditional resoil reclamation practice. Spoil pH in the 0 to 4 in depth was increased to the range 6 to 8 which was similar to the resoil pH, and complete vegetative cover was successfully established on all watersheds. However, plant biomass production was approximately 2 times larger on the resoiled watersheds than on the amended spoil. Consequently, erosion was smallest on the resoiled watersheds. All three reclamation treatments increased runoff water pH to >7 and decreased soluble Al. Concentrations of Ca and S were larger in runoff- and percolate-water samples from AFBC-treated watersheds than from the resoiled watersheds. Trace element concentrations in all water samples remained very low and showed almost no treatment effects.

Additional Key Words: runoff, water quality, soil quality

Introduction

Conventional reclamation of abandoned mined lands (AML) involves placement of a layer of topsoil over the graded minespoil. Because topsoil was generally not conserved when AML sites were mined, soil must be "borrowed" from adjacent land thereby creating another disturbed area. The cost of reclamation becomes prohibitive if sufficient topsoil is not available adjacent to the minesite. The need for borrow soil could be reduced or eliminated by direct revegetation of acidic minespoil to which sufficient alkaline and organic amendments have been added. The use suitable of by-product or waste product materials for amendment of acidic minespoil would further reduce reclamation costs and also remove these materials from the solid waste stream.

The scrubbing of SO_2 from the stack gasses of coal-fired boilers (flue gas desulfurization, FGD) produces large quantities of FGD by-products, which in many cases are highly alkaline (Carlson and Adriano, 1993). The industries which produce FGD by-products (primarily electric utilities) have strong economic and environmental incentive to beneficially use FGD by-products. Although there is considerable variation among FGD by-products produced by various scrubber technologies, most consist of 3

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components; calcium sulfite/sulfate (gypsum), residual alkalinity (lime/limestone), and coal ash. Because of their alkalinity, FGD by-products can be used to neutralize acidity in soils (Stehouwer et al., 1995a, Marsh and Grove, 1992, Korcak, 1980) and minespoils (Stehouwer et al., 1995b). Because of its mobility, the gypsum component of FGD by-products may help to alleviate toxic conditions below the depth to which they are incorporated (Sumner et al., 1986; Farina and Channon, 1988; Alva and Sumner, 1990; Alva et al., 1990).

Atmospheric fluidized bed combustion (AFBC) is a type of FGD in which pulverized coal is combusted in the presence of lime or limestone sorbent. The SO_2 reaction product is anhydrite ($CaSO_4$) and is collected, together with unspent sorbent, in both the bed and fly ash streams.

Similarly, bans on landfilling of organic wastes, and legislative mandates to reduce solid waste volumes, have provided strong incentives for composting organic waste materials. Organic amendments, primarily sewage sludge, have been shown to significantly improve long-term minespoil revegetation success (Sopper, 1992). Composted organic wastes would likely provide similar benefits, and large volumes could be utilized in minespoil reclamation.

Large amounts of these materials were required (>100 tons AFBC $acre^{-1}$, >50 tons compost $acre^{-1}$) in order to improve spoil quality parameters such as pH, base status, pH buffering, microbial activity, and mineralizable C. However, these materials also contain large quantities of soluble salts and some trace elements of environmental concern. These could negatively impact soil and water quality both directly and through mobilization of trace elements in the spoil. Greenhouse experiments showed that when used at the appropriate rate, FGD by-products could ameliorate phytotoxic conditions in the spoil without adversely affecting plant growth (Stehouwer et al., 1995b, c). Analysis of greenhouse pot leachates also showed that at agronomically appropriate application rates trace element concentrations remained at very low levels. When used at lower rates, pH amelioration was insufficient resulting in poor plant growth. When applied at higher rates excessively high pH and soluble salts limited plant growth, and solubilization of organic C increased transport of trace elements.

This watershed-scale field study was initiated to compare conventional reclamation of an acidic minespoil by topsoiling, with direct revegetation of the spoil amended with an AFBC by-product alone and AFBC combined with yard-waste compost. The study was designed to investigate the revegetation success and the impacts on soil, plant, and water quality of the three reclamation practices.

Materials and Methods

In the summer and fall of 1994, during reclamation of an abandoned coal mine site located near New Philadelphia in east central Ohio, six 1-acre watersheds were constructed for the study described in this paper (Fig. 1). Exposed underclay material on the site was graded to a 4% slope and recompacted to serve as an aquitard. Thickness of the clay pad ranged from 10 to greater than 30 ft in an area 670 by 450 ft. A 5 ft wide by 1 ft high clay berm was constructed on the clay surface to hydrologically separate each 1-acre watershed. Following preparation of the clay aquitard, 4 ft of acidic minespoil material was placed over the recompacted clay and graded to a 4% slope. Each watershed was hydrologically separated by building a 10 ft wide by 2 ft high berm directly above the underlying clay berm. Berms were similarly constructed at the lower end of each watershed to direct surface water flow into a 10 ft long approach. Water flowed through an H-flume instrumented with a stilling well and a water stage recorder. Approximately 0.3% of total surface water flow was automatically diverted to a holding tank to allow sampling of each flow event. A trench was dug 5 ft up slope from the lower end of each watershed to allow the placement of a perforated 4 in tile line in a shallow trench (6 in deep) on the clay

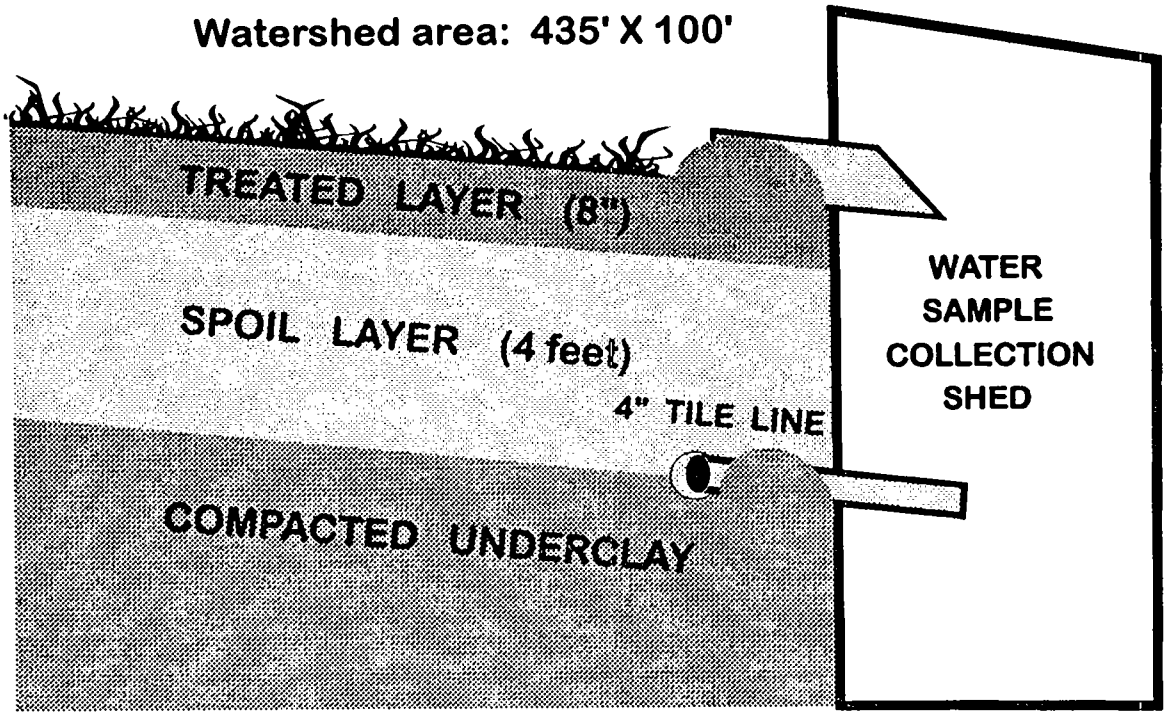

Fig. 1. Watershed design

surface. The tile line was covered with river-run gravel and the trench was backfilled. Any water which percolated through the spoil and flowed laterally over the clay was collected by the tile line and directed to an outlet pipe located below the surface water H-flume. All tile flow water was retained in a holding tank and sampled following each flow event.

Following completion of the construction phase, the three surface treatments listed below were applied to the spoil surface in each of 2 watersheds.

1. The conventional reclamation practice of application of 50 tons/acre of agricultural limestone on the spoil surface followed by placement of 8 in of borrowed topsoil material. Agricultural limestone was spread on the topsoil surface at a rate of 20 tons/acre and incorporated with a chisel plow and offset disc.
2. Application of 125 tons/acre of AFBC by-product followed by incorporation to a depth of approximately 8 in by multiple passes with a chisel plow and an offset disk.
3. Application of a mixture of AFBC by-product and yard waste compost applied at a rate of 125 tons/acre AFBC and 50 tons/acre compost followed by incorporation to a depth of approximately 8 in by multiple passes with a chisel plow and an offset disk.

Chemical characteristics of the minespoil and amendment materials are given in Table 1. The AFBC material used in this study was a blend of by-product collected from both the bed and fly ash streams. The by-product was conditioned with water at the power plant which resulted in the hydration of anhydrite to form gypsum, and of lime to form portlandite ($Ca(OH)_2$). The AFBC by-product had a neutralizing potential of 39% calcium carbonate equivalency. The source of the alkalinity in this material was predominantly $Ca(OH)_2$ with minor amounts of $CaCO_3$.

Watershed construction was completed in October, 1994 and plots were seeded to a grass-legume sward in

Table 1. Chemical characterization of minespoil, topsoil, yard-waste compost, and AFBC.

Parameter	Spoil	Topsoil	Compost	AFBC
Major Elements (%)				
Organic C	11.1	na	16.7	na
Total N	na	na	0.84	na
Aluminum	11.5	8.6	na	2.5
Calcium	0.04	0.07	1.69	26.1
Magnesium	0.50	0.53	0.35	3.65
Iron	5.57	3.96	1.77	5.9
Potassium	2.36	2.35	0.56	0.36
Phosphorus	0.07	0.04	0.15	0.03
Silicon	22.7	19.0	na	6.4
Sulfur	1.02	0.06	na	12.3
Trace Elements (μg g^{-1})				
Arsenic	46.3	5.5	11.47	71.5
Barium	701	503	na	204
Boron	na	na	39.11	418
Cadmium	0.8	3.3	<0.2	1.5
Chromium	94.4	95.6	284	42.2
Copper	26.8	62.8	69	49.5
Lead	78.0	15.9	26	17.4
Molybdenum	14.0	<0.2	27.79	22.4
Nickel	28.5	44.8	383	78.8
Selenium	4.5	<0.7	0.25	8.6
Sodium	150	150	207.9	1100
Zinc	<0.3	137.8	108	112
pH (1:1 water)	3.1	4.3	7.4	12.4

[2]Not analyzed.

November, 1994. The seeding mix included orchard grass (*Dactylis glomerata*), timothy (*Phleum pratense*), annual ryegrass (*Lolium multiflorum*), ladino clover (*Trifolium repense* Ladino), birdsfoot trefoil (*Lotus* sp.), and winter wheat (*Agropyron* sp.).

Baseline soil/spoil samples were collected in November, 1994 and again in June, 1995 by digging 3 cores (2 in diam) in each watershed. Cores were sampled in 2 in increments in the upper 12 in, and in 4 in increments from the 12 in depth down to the compacted clay layer. Soil samples were air-dried and ground to pass a 2-mm screen. Samples were analyzed for pH, electrical conductivity (EC), and for water soluble and Mehlich extractable concentrations of Al, Ag, As, B, Ba, Be, Ca, Cd, Co, Cr, Cu, Fe, K, Li, Mg, Mn, Mo, Na, Ni, P, Pb, S, Sb, Se, Si, Sr, V, and Zn by ICP.

Water sample collection began in the spring of 1995. Following each surface-flow event a hydrograph was constructed for the event, sediments deposited in the approaches and H-flumes were removed, weighed and sampled, and 2-L water and suspended sediment samples were collected from the holding tanks. Water samples were analyzed for pH, EC, acidity and alkalinity before and after filtering through a 0.45μm membrane filter. Sediment concentration was determined

by drying and weighing the filter cake. Filtrates were analyzed by ICP for dissolved concentrations of the same elements as listed above. Total element concentrations of water samples were determined by digesting unfiltered samples in hot perchloric and nitric acid. Digests were analyzed by ICP for the same elements listed above. Tile flow water samples were collected after each flow event and analyzed as described for the surface water samples. Water samples were also collected and similarly analyzed from a sediment pond which collected surface-water flow from the entire reclamation site. Surface water samples collected before reclamation (before pond construction) were obtained from intermittent stream flow draining the site.

Plant biomass production was determined by cutting three strips (2.5 x 70 ft) across each watershed in August, 1995. Plant material from each strip was dried for 48 h at 60°C and weighed.

Results and Discussion

All three treatments resulted in profile surface pHs near or slightly above neutral (Fig. 2). With all three surface treatments, profile pH decreased with increasing depth. The pH of topsoil and AFBC+compost profiles decreased to the range 4 to 4.5. In the AFBC profile, however, pH was somewhat higher, in the range 5 to 6. The reasons for the higher pH with AFBC are not clear. Although the gypsum component of AFBC is expected to be more mobile than agricultural limestone (Sumner et al., 1986), downward transport of Ca^{2+} and SO_4^{2-} would not be expected to have such a large influence on pH. Furthermore, if the pH increase was due to the AFBC, the AFBC+compost treatment would be expected to show a similar pH response with depth. It remains to be seen if this pH difference persists in subsequent sampling.

Soluble salt concentrations in the surface layer of topsoil were much lower than in the AFBC- or AFBC+compost-amended minespoil (Fig.

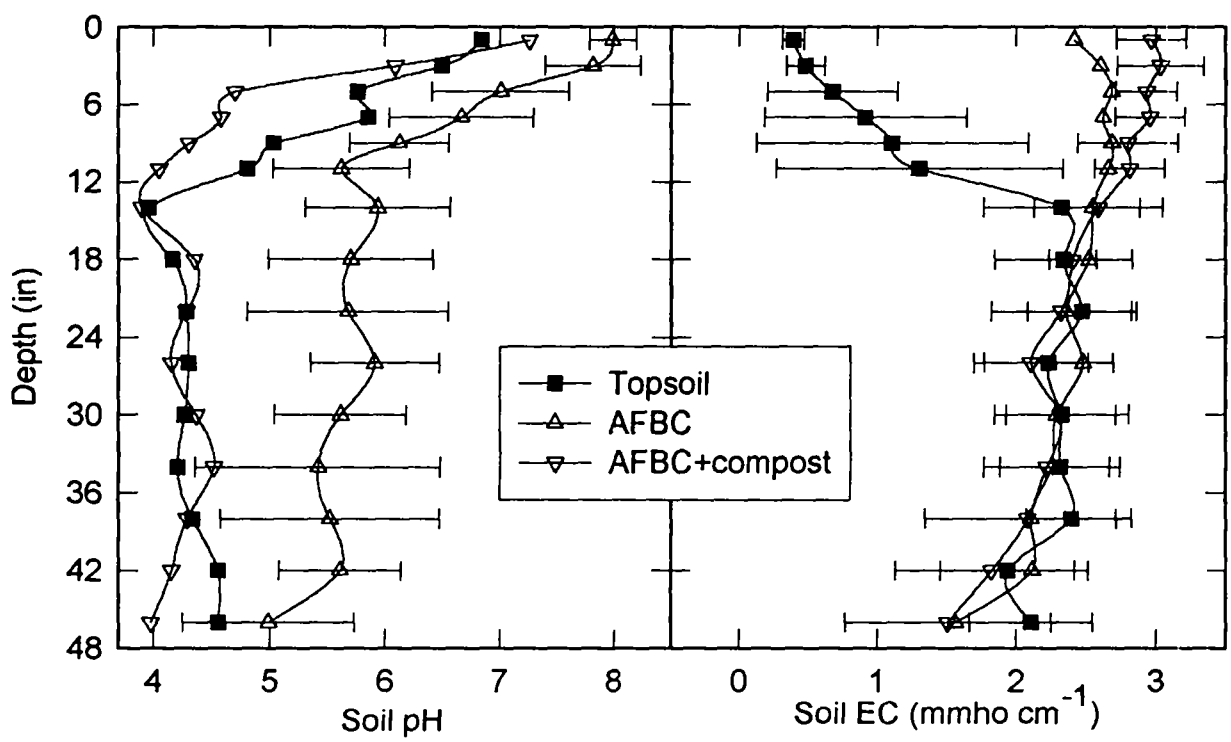

Fig. 2. Profile pH and electrical conductivity of acidic minespoil 9 mo after reclamation with topsoil, AFBC, or AFBC+compost surface treatments. (Error bars indicate plus and minus one standard deviation.)

2). Although AFBC alone did not increase spoil soluble salt concentrations, the addition of 50 tons acre^{-1} of yard-waste compost did increase salt concentrations slightly. Below the treated layers there were no differences among the three treatments with respect to soluble salts.

Establishment of vegetative cover was successful on all reclamation treatments, with 100% vegetative cover on almost all areas of all watersheds throughout the summer of 1995. Establishment was more rapid and growth was more vigorous on the topsoil treated watersheds. Topsoiled watersheds produced over twice as much plant biomass than did either the AFBC or the AFBC+compost treatments (Fig. 3). Although topsoil is clearly a better medium for plant growth than the minespoil material, amendment with AFBC did permit establishment of good vegetative cover on this otherwise toxic material. The high soluble salt concentration in the spoil material may account in large part for the less vigorous growth than on topsoil. It is expected that over time these salts will be leached downward. When this occurs, and if the spoil begins to develop other properties of a natural soil, plant growth may begin to approach that of the topsoil.

Prior to reclamation erosion rates from this abandoned mined land site were estimated at 420 tons acre^{-1} for a two-year storm. During the summer of 1995 no storm event produced greater than 2 tons acre^{-1} erosion from any watershed regardless of the surface reclamation practice. Thus both the conventional and the alternative reclamation practices were effective in greatly reducing erosion. However, both runoff volume (data not shown) and sediment losses (Fig. 4) tended to be smallest from topsoil watersheds and largest from the AFBC treated watersheds. Most of these differences appear to be directly related to the more vigorous vegetative growth on the topsoil watersheds.

All three reclamation treatments improved surface water quality. Prior

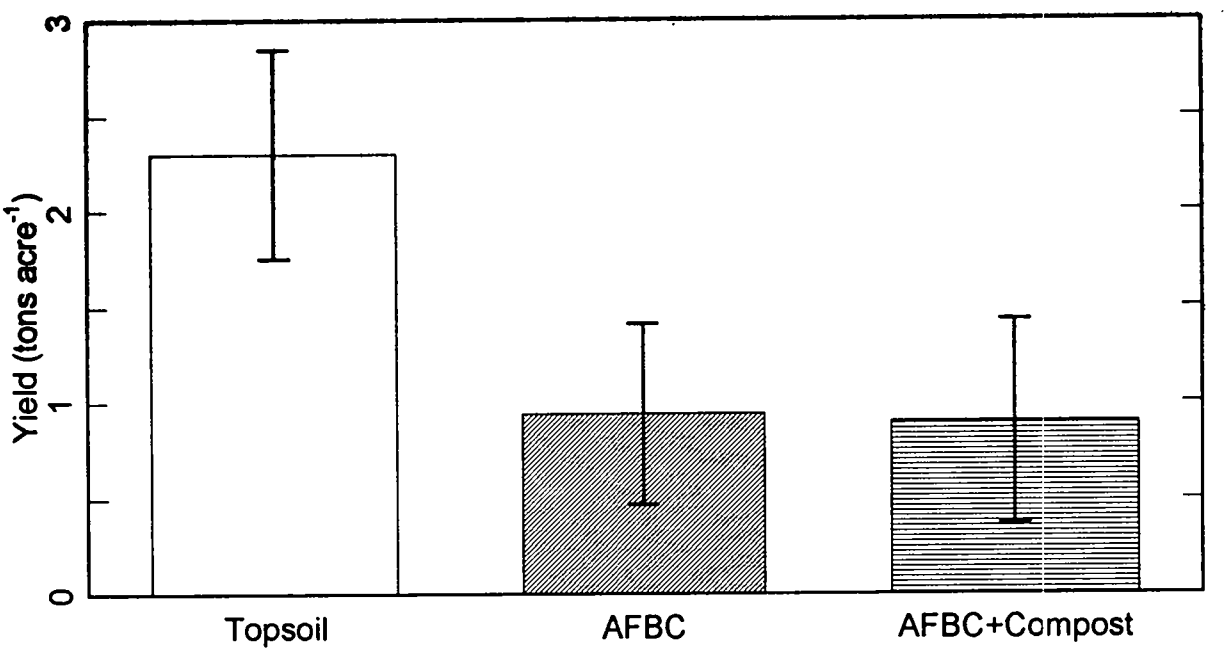

Fig. 3. Plant biomass yield in summer of 1995 on minespoil reclaimed with topsoil, AFBC, or AFBC+Compost. (Error bars indicate plus and minus one standard deviation.)

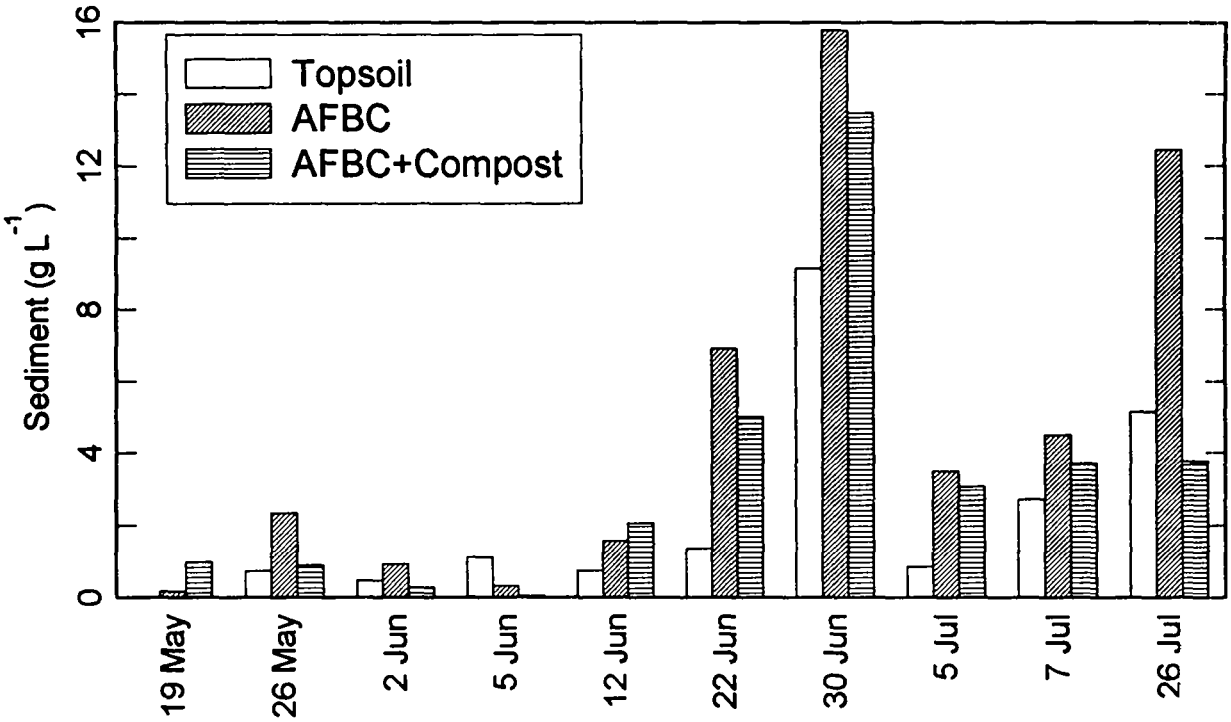

Fig. 4. Sediment concentration of 1995 summer storm runoff water from minespoil watersheds reclaimed with topsoil, AFBC, or AFBC+Compost.

to reclamation surface water pH was in the range of 2 to 3 with soluble Al concentrations as high as 120 mg L^{-1} (Figs. 5 and 6). All three reclamation treatments increased surface runoff water pH to the range 7 to 8, although the AFBC treatments tended to produce runoff water with slightly higher pH than did the topsoil. Changes in soluble Al showed just the opposite result. All three treatments decreased soluble Al to less than 0.3 mg L^{-1}. This result is expected given the increased pH and the low solubility of Al at near neutral pH.

Calcium concentrations in surface runoff water were clearly larger from the AFBC-treated watersheds than from the topsoiled watersheds (Fig. 7). This difference can be attributed to the much larger solubility of gypsum (a major component of AFBC) than limestone. As soluble gypsum is leached downward, and if vegetative cover becomes more vigorous, it is expected that the differences in soluble Ca between the AFBC and topsoil treatments will decrease. Similar results were observed with respect to soluble S (Fig. 8). Again the larger S concentrations with AFBC treatments can be attributed to the presence of relatively soluble gypsum in the AFBC. These differences are also expected to diminish gradually with time.

Several observations suggest that most of the water that reached the perforated tile line buried on the clay surface percolated through the spoil at the point where the trench was dug to install the tile rather than percolated through the spoil over the whole watershed area and flowed laterally over the clay surface. The spoil material was placed over the clay in several lifts using large earthmoving pans which resulted in significant compaction of the spoil. Surface runoff began very soon after the onset of a storm event and runoff volumes were large indicating there was little opportunity for percolation. Spoil core samples taken from just above the clay layer were quite dry even when the surface was wet. Finally, during surface flow

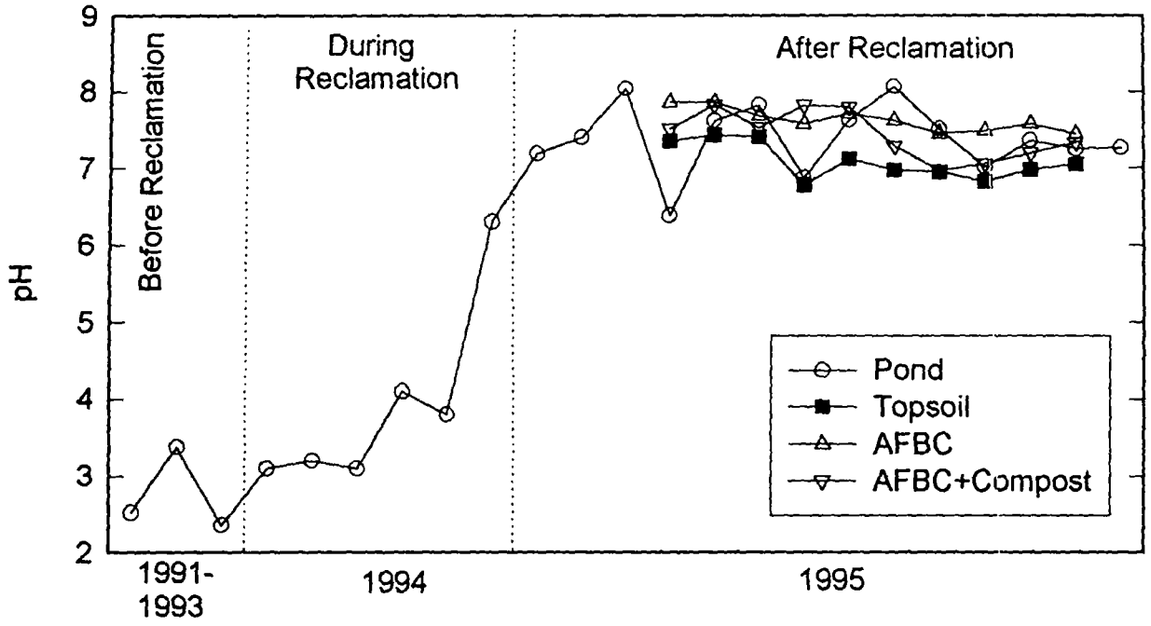

Fig. 5. pH of surface water runoff from minespoil reclaimed with topsoil, AFBC, and AFBC+Compost.

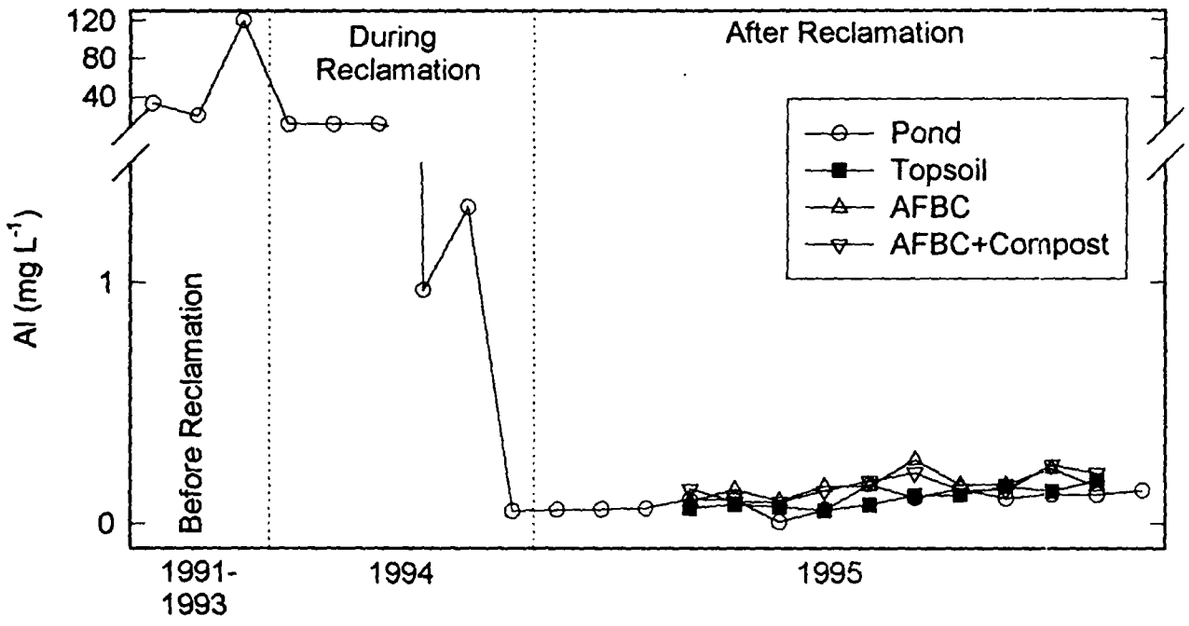

Fig. 6. Dissolved aluminum concentration in surface water runoff from minespoil reclaimed with topsoil, AFBC, and AFBC+compost.

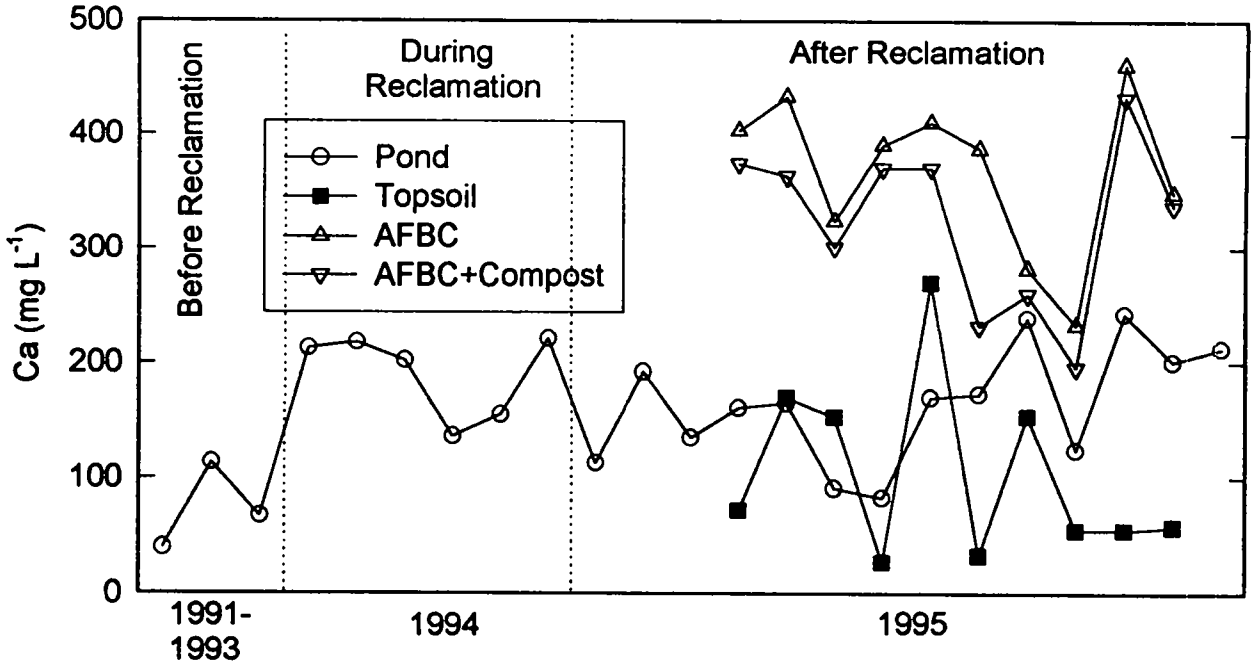

Fig. 7. Dissolved calcium concentration in surface water runoff from minespoil reclaimed with topsoil, AFBC, and AFBC+compost.

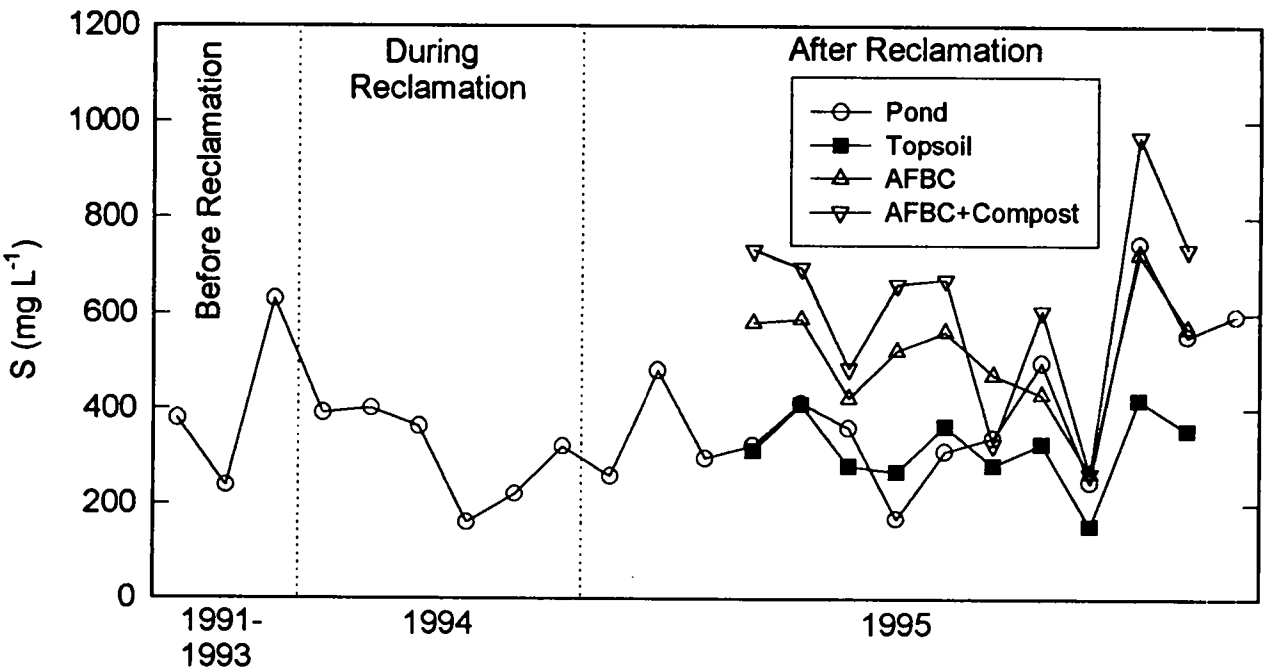

Fig. 8. Dissolved sulfur concentration in surface water runoff from minespoil reclaimed with topsoil, AFBC, and AFBC+Compost.

events water often puddled over the location of the trench. The tile water pH data also suggests limited interaction with subsurface acidic spoil and clay material (Fig. 9). Tile flow water from all three watersheds was only slightly lower than that of the surface water, and much higher than the pH of the subsurface spoil. Likewise soluble Al concentrations were in nearly the same range as that of the surface water (Fig. 10). Thus it appears that surface water is reaching the tile line via relatively rapid preferential flow paths and has limited interaction with the acidic spoil material.

The tile water is, however, interacting with the treated layers to a greater extent than the surface runoff water. Calcium concentrations in the tile water from all three watershed treatments were similar and in the range of 300 to 400 mg L^{-1} (Fig. 11). This was the same range as the surface water Ca for the AFBC treatments, but higher than for the topsoil treatment. Apparently the water percolating through the topsoil dissolved Ca from the ground agricultural limestone added to these watersheds. Reaction of the limestone with the acidic spoil released Ca^{2+} into the soil solution. Because of the large sulfate concentrations in the spoil (due to pyrite oxidation), the solubility of the Ca was controlled by gypsum, just as it was in the AFBC watersheds where gypsum was added.

For most of the summer, tile water S concentrations were larger in the AFBC watersheds than in the topsoil watersheds, reflecting the large addition of soluble sulfate in the gypsum component of the AFBC (Fig. 12). Toward the end of the summer, S concentrations from all three surface treatments and the pond appeared to be approaching a similar level. This may indicate that the system is approaching a new steady state condition where S concentrations are controlled by gypsum solubility.

Trace metal concentrations in surface water, tile water, and pond water generally remained very low and

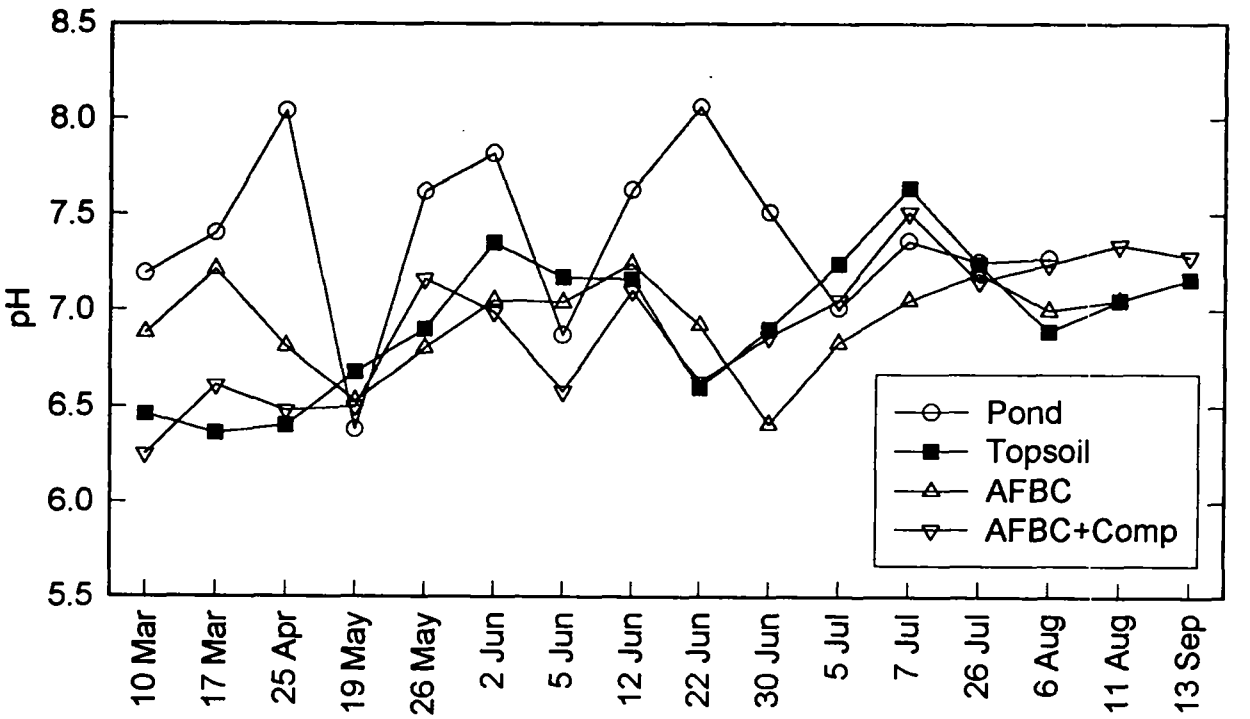

Fig. 9. pH of subsurface tile water flow from minespoil reclaimed topsoil, AFBC, and AFBC+compost.

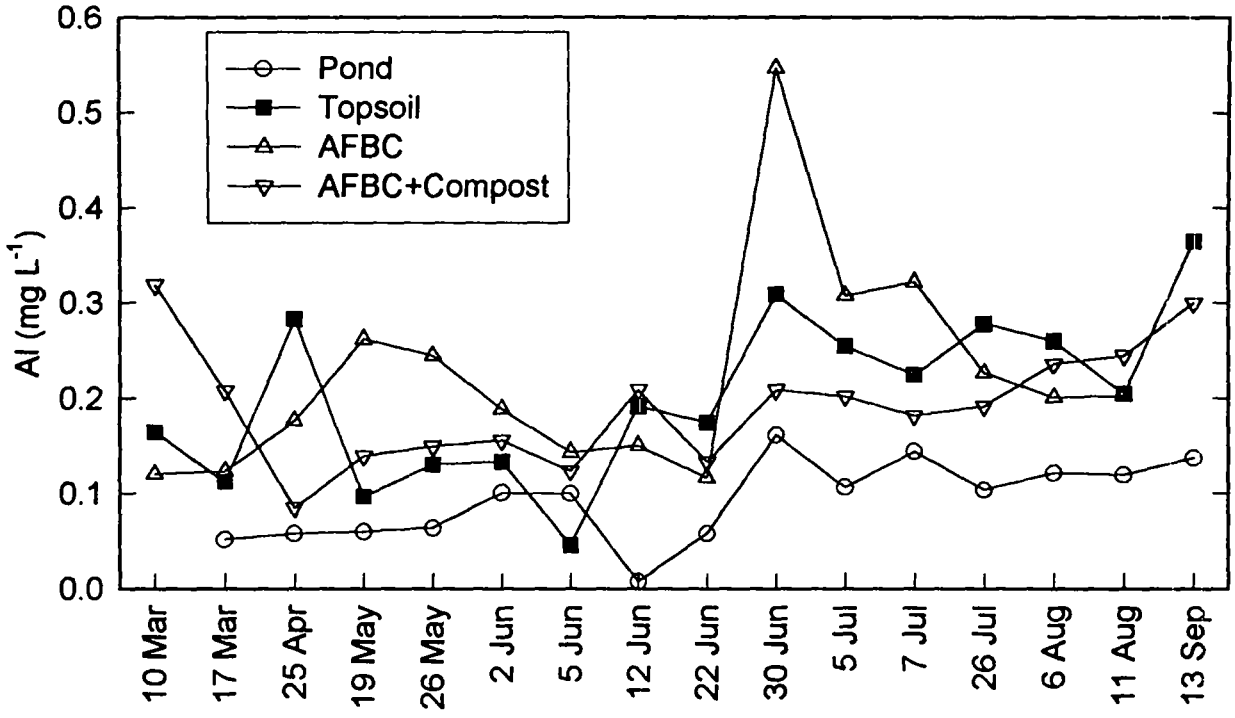

Fig. 10. Dissolved aluminum concentration in subsurface tile water flow from minespoil reclaimed with topsoil, AFBC, and AFBC+compost.

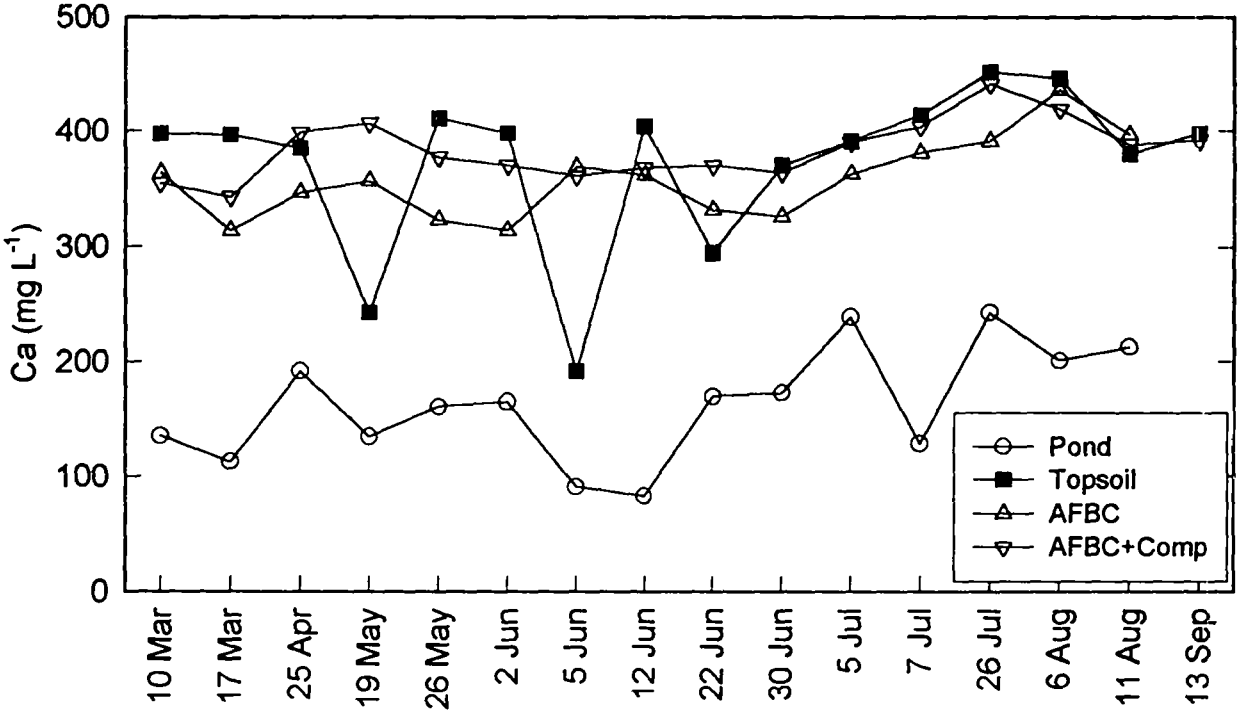

Fig. 11. Dissoved calcium concentration in subsurface tile water flow from minespoil reclaimed topsoil, AFBC, and AFBC+compost.

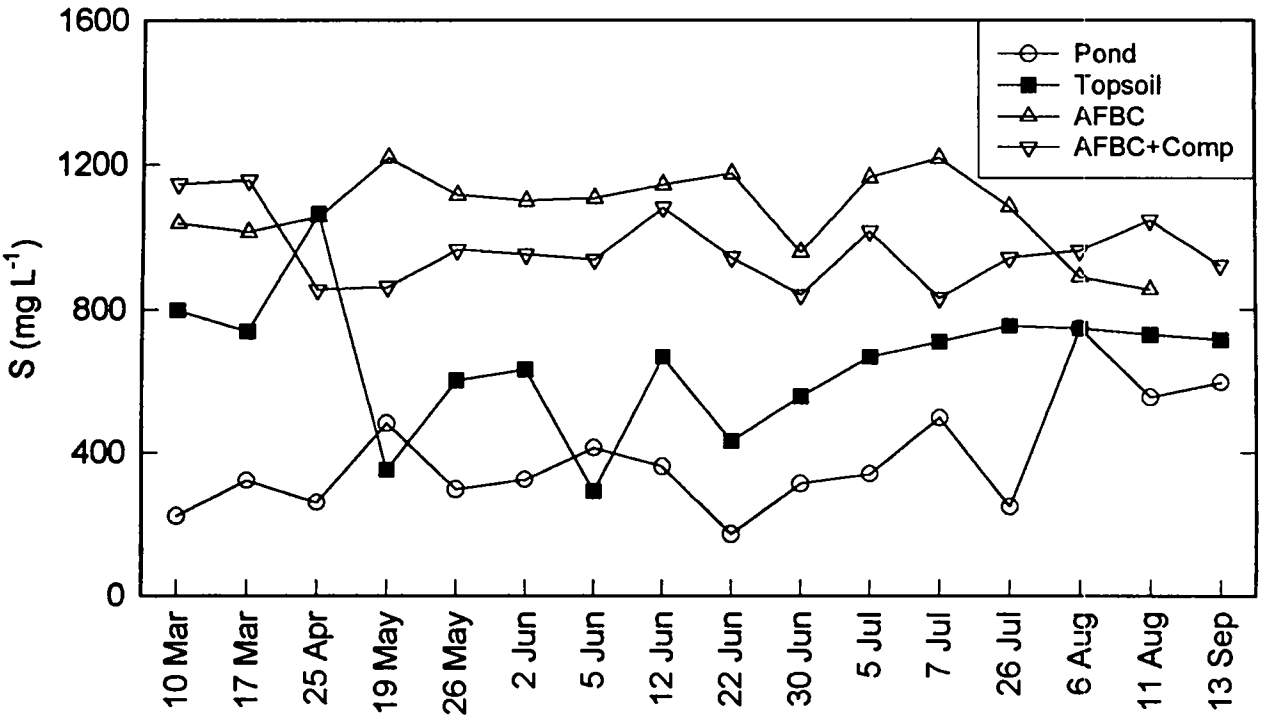

Fig. 12. Dissolved aluminum concentration in subsurface tile water flow from minespoil reclaimed with topsoil, AFBC, and AFBC+compost.

unaffected by the reclamation treatments (Table 2). Mean concentrations of As, Ba, Cd, Cr, Cu, and Se were below detection limits or below primary drinking water standards. Although occasional samples had higher concentrations of these regulated metals, these hits could not be ascribed to the AFBC or compost as all three reclamation practices gave values generally in the same range. This was generally the case for all measured trace elements. Mean Ni and Pb concentrations were above drinking water standards for the tile flow, but both mean and maximum concentrations were similar for all three reclamation treatments.

Boron and Mn concentrations both showed some response to the reclamation surface treatments (Table 2). These elements are of interest because of their potential for phytotoxicity at elevated concentrations. Boron was enriched in both surface and tile waters from the AFBC and AFBC+compost treated watersheds. This reflected the B added with the coal ash component of the AFBC by-product. It should be noted that B toxicity was not observed in the vegetation on these watersheds. Furthermore, the most phytotoxic B species are also highly water-soluble. Therefore the B enrichment indicated that the most phytotoxic B was being removed from the spoil rooting zone. Manganese concentrations in surface waters were similar for the three treatments, but in tile flow AFBC treatments had larger Mn concentrations than did the topsoil treatments. This indicated Mn was mobilized from the spoil in these watersheds. Mobilization of Mn following FBC by-product application has also been observed in acidic agricultural soil (Stehouwer et al., 1995a). The mechanism for this mobilization was not clear.

Conclusions

In the first year after reclamation, good vegetative growth on minespoil amended with AFBC or AFBC and yard-waste compost was achieved. Vegetative cover was nearly 100% over all areas of the 1-acre test

Table 2. Concentrations of trace elements in various water sources from minespoil reclaimed with topsoil, AFBC, and AFBC+compost. Values are means and maximum levels of all flow events in the spring and summer of 1995.

Element	Pond		Surface Runoff Water						Tile Flow Water					
			Topsoil		AFBC		AFBC+Compost		Topsoil		AFBC		AFBC+Compost	
	Mean	Max	Mean	Max	Mean	Max	Mean	Max	Mean	Max	Mean	Max	Mean	Max
	mg kg^{-1}													
Ag	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
As	<0.04	0.083	<0.04	0.088	<0.04	0.096	<0.04	0.069	<0.04	0.106	<0.04	0.162	<0.04	0.095
B	0.526	1.082	0.147	1.867	3.163	6.875	1.977	3.681	0.207	0.961	2.945	5.178	1.644	3.279
Ba	0.023	0.040	0.017	0.032	0.012	0.020	0.010	0.017	0.017	0.032	0.017	0.024	0.019	0.042
Be	<0.001	0.001	<0.001	<0.001	<0.001	<0.001	<0.001	0.001	<0.001	0.001	<0.001	0.002	<0.001	0.001
Cd	<0.001	0.001	<0.001	0.001	<0.001	0.001	<0.001	0.002	<0.001	0.010	0.001	0.011	<0.001	0.004
Co	0.008	0.024	0.008	0.025	0.010	0.034	0.014	0.036	0.223	1.216	0.416	2.258	0.341	1.266
Cr	0.003	0.009	0.003	0.010	0.005	0.012	0.005	0.011	0.009	0.019	0.010	0.031	0.010	0.019
Cu	0.009	0.023	0.010	0.030	0.006	0.029	0.013	0.027	0.005	0.048	0.002	0.022	0.006	0.035
Mn	1.33	3.527	1.63	7.330	1.06	3.722	2.68	8.042	23.94	124.7	32.80	198.4	34.27	122.4
Mo	<0.011	0.030	<0.011	0.020	0.028	0.065	0.018	0.043	<0.011	0.017	<0.011	0.033	<0.011	0.018
Ni	0.023	0.053	0.014	0.075	0.015	0.101	0.035	0.091	0.738	2.935	1.195	4.246	0.940	2.986
Pb	<0.02	0.048	<0.02	0.055	<0.02	0.050	<0.02	0.055	0.026	0.087	0.027	0.105	0.023	0.050
Se	<0.09	<0.09	<0.09	<0.09	<0.09	<0.09	<0.09	<0.09	<0.09	0.100	<0.09	<0.09	<0.09	<0.09
Zn	0.065	0.144	0.065	0.135	0.021	0.139	0.002	0.113	0.139	0.192	0.382	0.204	0.131	0.217

watersheds. The establishment of vegetative cover directly on amended minespoil reduced erosion rates to less than 2 tons acre^{-1} for a 2-year storm. Although vegetative growth was more vigorous, biomass production was greater, and erosion was less on topsoiled watersheds than on watersheds with direct revegetation of minespoil, it is expected that these differences will become less as the spoil material begins to develop more soil-like properties. The successful revegetation of the minespoil can be attributed to increased pH and decreased concentrations of soluble Al and Fe in the minespoil. The less vigorous plant growth in the minespoil may be attributed in part to the large soluble salt concentrations in the spoil. It is expected that these concentrations will decrease over time as salts are leached from the profile surface.

No detrimental environmental effects were observed or measured as a result of using AFBC or compost in reclamation of acidic minespoil. Similar improvements in surface runoff and tile flow water quality were observed with topsoil, AFBC, or AFBC+compost. All three reclamation practices increased water pH to neutral levels and greatly decreased soluble Al. Concentrations of trace elements of environmental concern remained at very low concentrations in waters from all three reclamation practices. In those cases where measurable concentrations were present, there were no differences among the three reclamation treatments.

Results during the first year following reclamation show that the use of AFBC or AFBC and yard-waste compost will allow successful direct revegetation of acidic minespoil without adverse environmental effects. This site will continue to be monitored for several years to determine the long-term revegetation success and environmental impact of direct reclamation of minespoil with AFBC and yard-waste compost amendments.

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MUNICIPAL SEWAGE SLUDGE AND PHOSPHORUS AS AMENDMENTS FOR MINESOILS ON THE CUMBERLAND PLATEAU[1].

by

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Abstract. A study was conducted to evaluate the benefit of applying municipal sewage sludge and/or phosphorus on surface mined land on the Cumberland Plateau in Fentress County, Tennessee. The objective of the study was to evaluate minesoil fertility and plant response after treatment. The site was prepared according to surface mine regulations and sampled before and after treatment. The experiment was conducted as a split, split, split block arranged in a randomized complete block design with four replications. Each replication contained a sludge treatment, a phosphorus treatment, a sludge/phosphorus combination treatment and a control. Minesoils amended with sludge alone had the greatest increase in fertility and plant response. Phosphorus alone had a comparable, though slightly lower response of plant growth on the sludge. Legume species responded particularly well on phosphorus. A combination of sludge and phosphorus had a slightly suppressive effect on some plant growth responses compared to sludge alone.

Additional Key Words: Minesoil Fertility, Sludge Application, Revegetation.

Introduction

The impact of surface mining for coal on the land is a major environmental concern. Surface mined land has been reclaimed to the standards of the Surface Mine Reclamation Act of 1977 (Wagner, 1979) since its inception. New soils created after the surface mining processes range from highly acid to neutral pH values. Soils that are highly acid have high concentrations of Al and Mn. Additionally, these soils contain low levels of N, P and organic matter (OM). Researchers have reported that these conditions are detrimental to plant growth and corrective measures should be employed for successful establishment of vegetation (Fribourg *et al.*, 1981; Lyle, 1987; Sopper and Seaker, 1990). Spoil from the "high-sulfur" coal of the eastern U. S. containing various forms of pyrite (FeS_2) must be limed sufficiently to counteract the continuing formation of sulfuric acid. Within two years spoil will become too acid to support a satisfactory vegetative cover if not limed sufficiently (Sopper and Seaker, 1990).

An alternative to standard fertilization is to use municipal sewage wastes. Halderson and Zenz (1978) reported that typical plant nutrient analyses of municipal sewage wastes is as follows; N = 5% ; P = 2.5%; and K = 0.4% (on % dry matter basis). Advantages of reclamation with sewage sludge are the rapid incorporation of OM and plant nutrients into the minesoil (Loehr *et*

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al., 1979).

Studies in Pennsylvania showed that applying sludge to minesoils enhanced seed germination and vegetative growth (Loehr *et al.*, 1979). A study of sludge application on revegetated minesoil in West Virginia indicated that grass density and biomass increased with increased application of sludge, while numbers and biomass of legume species decreased (Skousen, 1988). Data and observations on the Fentress County mine site (Walker, 1993) confirmed this observation.

Sludge provides an immediate addition of OM and microorganisms to the soil. Natural accumulation of organics usually takes many years to complete. Municipal sludges also contain varying quantities of the needed P from detergents and other sources and could supply up to 27% of the P needed in agriculture (Sommers and Sutton, 1980; Suss, 1981). The P available to plants in sludge varies from 20-100% compared with 100% in monocalcium phosphate (Haan, 1981). However, the larger quantities of P, N and OM in municipal sludge could counteract the chemical binding of P that takes place due to poor minesoil conditions.

Moisture stress causes P to be unavailable to plants (Ozanne, 1980). The OM in sludge helps the soil retain moisture, which could aid P availability. Sludge stabilized by heavy lime treatment can have value as a liming agent as well as by adding organics and N (Jacobs, 1981). Application of limed sludge must be carefully controlled as excessive quantities of limed sludge can create pH levels that are too high and conditions detrimental to plant growth (Morel, 1981; Walker, 1993).

Minesoils are often moderately to extremely deficient in P which is an important plant nutrient for establishing most plant species (Fribourg *et al.*, 1981; Buck and Houston, 1986). The low pH of some minesoils binds P to Al and Fe in chemically insoluble forms unavailable for plant use. Phosphate dissolved in H_2SO_4 (a by-product of pyrite oxidation in acid minesoil) is too strongly bound to Fe and Al oxides for plant uptake (Hani *et al.*, 1981). The addition of P to the soil is particularly helpful in the initial establishment of vegetation, especially when legumes are desired.

The objective of this study was to evaluate the benefit to plant growth and soil fertility of applying municipal sewage sludge and/or P on surface mined land in Tennessee.

Materials and Methods

Site Location

The experiment was located on the Cumberland Plateau in Fentress County approximately 2 miles south of Jamestown, TN and west of state highway 127 at an approximate elevation of 1600 ft. above mean sea level (Fig. 1). The study site was a 3.5 acre portion of an area that had been surface mined for bituminous coal by the Meta-Elkhorn mining company. Approximately 25-30 ft. of overburden was removed to reach the Nemo coal seam. The minesoils were dominately a fine sandy loam texture containing weathered sandstone, coal, and shale fragments (Branson *et al.*, 1991).

Geology and Soils

The study area was located on the Cumberland Plateau overthrust containing Pennsylvanian age sandstone and shale (Stearns, 1954). The Crab Orchard Mountain group (Rockcastle conglomerate) was identified as the sandstone removed during the mining. The Rockcastle formation is characterized as a sandstone with a shaly unit and coal horizon which occurs approximately in the middle of the formation.The native soils on the Cumberland Plateau are Ultisols and Inceptisols that have formed in residuum in the Pennsylvanian sandstones and shales, or in colluvium from sandstone and shale. The dominant soils are 2 to 4 feet deep over rock (Springer and Elder, 1980) well drained, loamy, strongly acid, and low in natural fertility especially with respect to phosphorus (Jared, 1972).

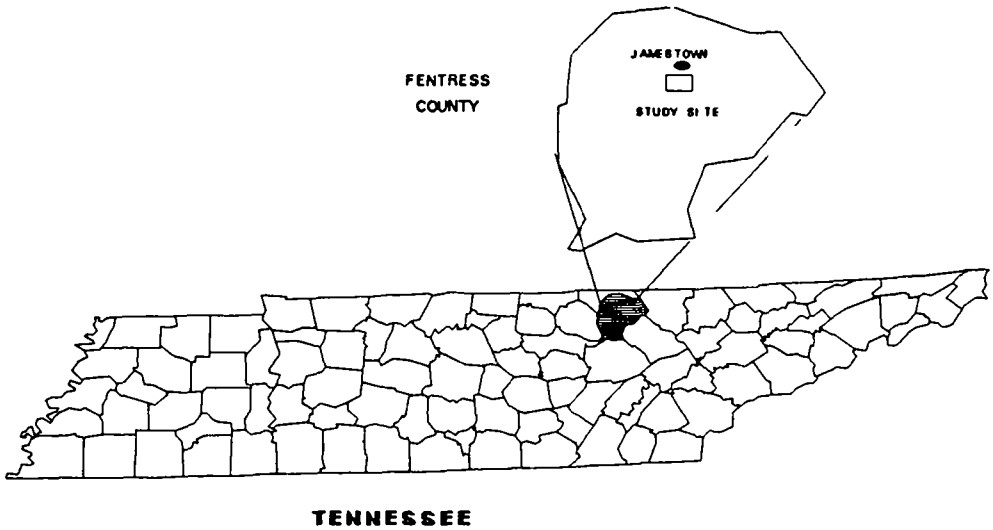

Figure 1. A map showing the study site, located in Fentress County, Tennessee.

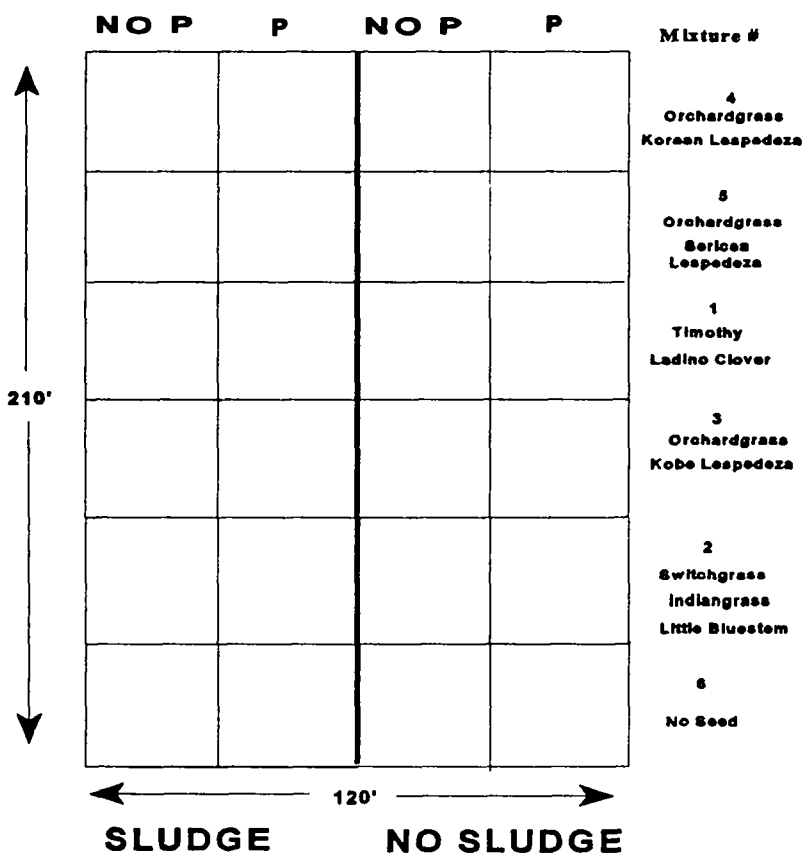

Figure 2. Schematic design of a representative experimental block containing the randomly located sludge and P amendments and seed mixtures.

Field Methods

The study site was prepared and limed by the mine company according to surface mine reclamation regulations prior to installation of the research plots and soil amendments (Wagner, 1979). The research area was divided into four blocks, each measuring 210' long and 120' wide, for the experiment. Figure 2 shows a representative experimental block. Eight soil samples were collected at 6 inch depths from within each split block and combined into bulk samples prior to soil amendment. One-half of three blocks was amended with open-air, composted sludge from Rockwood, Tennessee. One-half of one block was amended with limed, dewatered, municipal sewage sludge from Knoxville, Tennessee. Acquisition difficulties necessitated the use of sludges from two sources. One-half of each block was left unamended by sludge. Bulk samples were collected from each load of sludge before it was applied to the site. The sludge was dumped at rates varying from 8 to 16 tons per acre depending on the amount in the truck (1 truck load was applied to each block), spread with a box grader, and incorporated into the minesoil to a depth of 6 to 8" by discing. It was not possible to incorporate uniform quantities of sludge throughout the sludge blocks. After the sludge was applied, triple superphosphate $Ca(H_2PO_4)_2$ was applied by a cyclone hand spreader to half of each sludge strip and half of each unamended strip at a rate of 50 lbs P/acre .

The statistical design was a split, split, split block with 4 replications. Each replication contained 4 different soil amendments and 6 seed treatments in 30' by 35' plots (Fig. 2). Each replication contained a sludge amendment, a P amendment, a sludge + P combination amendment and an unamended-limed control applied in randomly selected strips. Plots were then seeded in randomly selected strips across all amendments with one of the following five seed mixtures.

1) Ladino clover, *Trifolium repens*, timothy, *Phleum pratense* (LDT)
2) Switchgrass, *Panicum virgatum*, indiangrass, *Sorghastrum nutans*, little bluestem, *Schizachyrium scoparium* (WSG)
3) Kobe lespedeza, *Lespedeza striata*, orchardgrass, *Dactylis glomerata* (KBO)
4) Korean lespedeza, *Lespedeza stipulacea*, orchardgrass (KRO)
5) Sericea lespedeza (A. U. Lotan), *Lespedeza cuneata*, orchardgrass (AUO)
6) No Seed Control (NS)

Vegetation was identified and percent cover was estimated visually in five one square foot quadrats within each plot post-amendment in 1989 and 1990. Soil samples were collected to 6 inch depths and combined as a bulk sample from each half block preamendment in 1989. A soil sample was collected from each seed plot and combined as a bulk sample from within each amended and unamended strip in each block in 1990.

Sample Analysis

The blocks were mapped for areas of light to heavy sludge content and this information was considered in the plant statistical analysis. Chemical analyses were performed on the soil samples and statistical analyses (SAS General Linear Models and Least Squares Means) were performed on the soil and vegetation data (SAS, 1989).

Soil and Sludge pH. The pH was measured in the laboratory on representative samples of the sludges applied to the minesoil. The sludge from Rockwood, TN had an average pH of 6.2. The sludge from Knoxville, TN had an average pH of 11.0. The pH was measured on each of the soil samples taken at the site (Southern Cooperative Series, 1974).

Cation Analysis. Soil samples were extracted using the Mehlich I soil test (Baker and Amacher, 1982; Southern Cooperative Series 1974). The extracts were analyzed for P, Ca, K, Mg, Cu, Zn, Mn, and Fe on an Inductively-Coupled Plasma Spectrophotometer (Shugar and Dean, 1990).

Results and Discussion

Observation of the study site in

June 1989, (one month after treatment and seeding) revealed dramatic differences in vegetative response among amendments. Seeds on unamended control areas had germinated, but plants were very sparse and severely stunted. Plants on P amended plots had germinated well and showed fair to good growth depending on the species. Sludge treated areas showed dense, luxuriant growth, in part due to heavy weed and tomato seed contamination of the open-air composted Rockwood sludge. The lime-stabilized, dewatered Knoxville sludge had no weed seed contamination, possibly due to high pH, and the applied experimental seeds germinated well. Excessive applications of lime-stabilized sludge were apparently toxic to the plants as no seeded species germinated in isolated spots of excessively concentrated Knoxville sludge. It appeared that P application greatly enhanced performance of all the legume species (Fig. 3). Grass species such as timothy and orchardgrass grew much more vigorously on sludge amended areas than on the other amendments. Unseeded controls had very sparse growth on all amendments, but volunteer species were present even on unamended plots. The plots amended with the combination sludge + P had slightly less cover than those amended with sludge. All treatments except the unamended plots resulted in 60 to 90% cover in 1990 (Fig.4). Grasses were more vigorous on sludge treated soils while legumes were predominant on P treatments. The percent cover of all seed mixtures on the combination (sludge + P) amendment remained slightly lower than the same mixtures on sludge alone, perhaps due to chemical toxicity or binding of needed phosphorus or other nutrients.

Table 1 shows minesoil fertility analyses on experimental plots before and after treatment. Values for pH on preamendment plots ranged narrowly between 5.1 to 5.2 for the plots. Ca averaged 228 lbs/a. Phosphorus concentrations ranged from 0.5 to 3.4 lbs/a, near the lower end of detection limits. K averaged 67 lbs/a and Mg 50 lbs/a.

Table 1 shows a dramatic increase in the Ca values between the preamendment analysis in 1989 (228 lbs/a) and the unamended control for 1990 (2169 lbs/a) which was attributed to the addition of lime during site preparation. Lime increased the pH of the unamended control plots to an average of 6.3. P and K levels showed little change from the preamendment levels after lime application. K levels were low for legumes and plants that germinated on limed, unamended plots were severely stunted. Mg levels were an average of 47 lbs/a higher after the addition of lime.

Effect of amendment on soil fertility

All soil fertility levels measured were higher on minesoil amended with P than on the limed, unamended control (Table 1). However, these changes were not significant at the $p \leq 0.05$ level. Statistical analysis of the differences between the preamendment (prior to lime and soil amendment) fertility levels of 1989 and the limed, P-amended fertility levels of 1990 for all nutrients showed no significance (Table 2). Minesoils amended with sludge had higher P levels by 244 lbs/a compared to the unamended control and 215 lbs/a compared to the P amendment (Table 1). These differences were significant at the $p \leq 0.05$ level. K levels on sludge plots were similar to levels on the unamended control and on P amendments. Ca and Mg levels were not significantly higher on sludge than on the other soil amendments. Statistical analysis of the differences in the preamendment (prior to lime and soil amendment) fertility levels from 1989 and the limed, sludge-amended levels from 1990 show P levels to be significantly higher on the sludge amendment at the $p \leq 0.05$ level (Table 2).

The combination sludge + P amendment exhibited a reduction of P and Mg, when compared to the sludge amendment, possibly due to inconsistent sludge application (Table 1). However, the percent cover of vegetation was lower on the combination amendment than the sludge with the application error considered. Statistical analysis of soil fertility showed no significant effect of the combination amendment on any nutrients

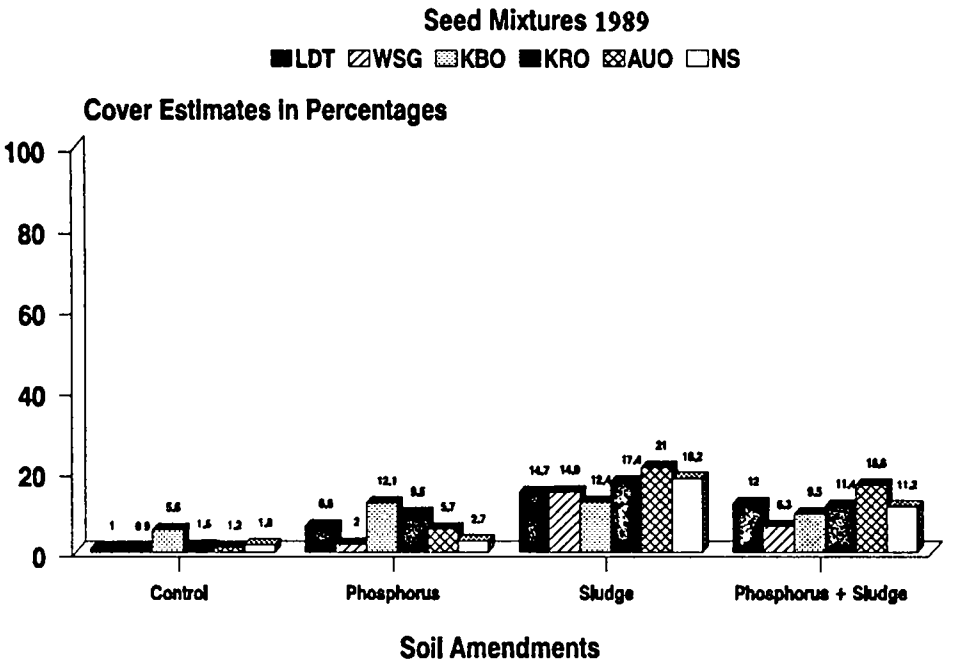

Figure 3. Mean percent cover estimated visually for each wildlife seed mixture relative to soil amendments during the first growing season in 1989. Abbreviations defined in Field Methods.

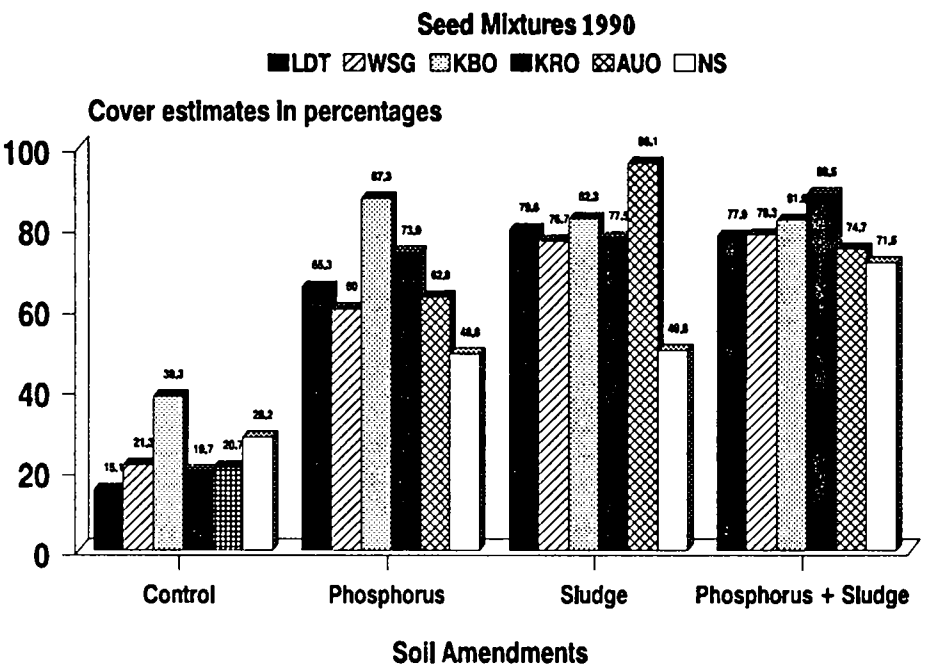

Figure 4.Mean percent cover estimated visually for each wildlife seed mixture relative to soil amendment during the second growing season in 1990. Abbreviations defined in Field Methods.

Table 1: Minesoil fertility analysis in 1989 before liming and soil amendments and in 1990 after liming and soil amendments. N=4.Least squares means ± standard error are shown.

	Lbs/A				
Amendment	Ca	P	K	Mg	pH
Preamend[θ]	228 ± 50	2 ± 0.5	67 ± 2	50 ± 3	5.2 ± 0.02
Control[1]	2169 ± 632	3 ± 46	77 ± 8	97 ± 18	6.3 ± 0.16
P	3675 ± 706	31 ± 52	90 ± 9	144 ± 20	6.5 ± 0.18
Sludge	4280 ± 815	246 ± 60*	86 ± 11	173 ± 23	6.5 ± 0.21
S + P	4007 ± 706	99 ± 52	79 ± 9	138 ± 20	6.7 ± 0.18

*Significant differences in soil fertility due to soil amendment based on Type III SS PR>F ≤0.05
[θ]Nutrients before application of lime and soil amendments.
[1] Unamended, limed control. Statistics do not include preamendment values.

Table 2: Differences in minesoil fertility of plots between 1989 and 1990 (Amended minus unamended). N=4. Least squares means + standard error are shown.

	Lbs/A				
Amendment	Ca	P	K	Mg	pH
Control	1919 ± 606	1 ± 46	11 ± 12	47 ± 18	1.1 ± 0.19
P	3457 ± 677	28 ± 52	23 ± 13	98 ± 20	1.4 ± 0.21
Sludge	4088 ± 782	243 ± 59*	18 ± 15	122 ± 23	1.3 ± 0.24
S + P	3769 + 677	97 + 52	12 + 13	85 + 20*	1.6 + 0.21

*Significant differences in soil fertility due to soil amendment based on Type III SS PR>F ≤ 0.05.

compared to the limed control. Statistical analysis of the differences in the preamendment (prior to lime and soil amendment) fertility levels from 1989 and the limed, combination amended levels from 1990 showed Mg levels significantly higher on the combination amendment at the $p \leq 0.05$ level (Table 2). Differences in the other nutrient levels were not significantly affected by the combination amendment.

Effect of soil amendment on heavy metals

Heavy metals were present in small quantities prior to soil amendment in 1989 (Table 3). Preamendment Cu levels averaged 0.75 lbs/a, Fe averaged 64 lbs/a, Mn averaged 27 lbs/a, and Zn averaged 4 lbs/a. Pb and Cd levels were too low to detect in all samples analyzed. The post amendment data for the 1990 limed, unamended control show heavy metal levels on all control plots decreasing slightly from the preamendment levels (Table 3). None of the changes in heavy metal concentration between the limed control and the P amendment were statistically significant (Table 3). Differences in heavy metal concentrations from preamendment (prior to lime and soil amendment) levels in 1989 to post-amendment levels in 1990 did not change significantly with the application of P (Table 4). Minesoils amended with sludge had Cu levels 9.0 lbs/a higher than the unamended control ($p \leq 0.05$) and 8.8 lbs/a higher than the phosphorus amendment ($p \leq 0.05$) (Table 3).

Sludge amendment had no significant impact on the levels of Fe. Sludge had significantly higher levels of Mn and Zn compared with the unamended control and the P amendment. Copper and Zn levels were significantly higher on sludge plots after lime and sludge were applied ($p \leq 0.05$) (Table 3). Mn levels were significantly lower on sludge ($p \leq 0.05$) than the preamendment levels. Application of sludge and lime did not significantly change Fe levels.

Minesoils amended with the combination sludge + P amendment contained lower levels of Cu and Mn and much lower levels of Zn compared to the sludge amendment, possibly due to inconsistent sludge application (Table 3). There was no significant effect of the combination amendment on any heavy metal concentrations compared to the unamended control. Statistical analysis of the differences between preamendment (prior to lime and soil amendment) metal levels in 1989 and post-amendment levels in 1990 indicated no significant differences on the limed, combination amendment from preamendment levels (Table 4).

Phosphorus as a Soil Amendment

P application in this study enhanced the establishment of vegetation. Plots amended with P in Fentress County were visibly higher in vegetative cover and the plants were more vigorous than plants on unamended control plots in both growing seasons (Figures 3 and 4). The P amendment was sufficient to establish vigorous stands of all the species of legumes in the legume/grass seed mixtures planted. P application in this study resulted in higher initial vegetative response. Other studies have shown that continued applications of P are required to maintain long term stands (Barnhisel *et al.* 1979).

The legumes on P amended plots will add N and, eventually, OM to the minesoil. However, this will occur more slowly than with organics supplied through the addition of municipal sludge. Ca, Mg and K fertility levels were increased with the application of triple superphosphate $Ca(H_2PO_4)_2$. Triple super phosphate contains 20% Ca, which explains the increase in Ca levels seen on P amended plots (Lyle, 1987). K levels were similar on all three soil amendments ,though slightly lower on the sludge + P amendment.

Municipal Sludge as a Soil Amendment

Previous studies of fertilization with sewage sludge have indicated that equal or better long

Table 3: Minesoil heavy metal analysis in 1989 before liming and soil amendment and in 1990 after liming and soil amendments. n=4.Least squares means ± standard error are shown.

	Lbs/A			
Amendment	Cu	Fe	Mn	Zn
Preamendment[e]	0.76 ± 0.03	63 ± 4	27 ± 1	4.2 ± 0.3
Control[1]	0.68 ± 1.41	52 ± 11	14 ± 4	3.3 ± 16
Phosphorus	0.82 ± 1.58	48 ± 12	19 ± 4	3.9 ± 18
Sludge	9.66 ± 1.82*	46 ± 14	30 ± 5*	100.3 ± 21*
Sludge + P	3.97 + 1.58	50 + 12	26 + 4	31.1 + 18

*Significant differences in heavy metal concentration due to soil amendment based on type III SS PR>F ≤0.05.
[e]Nutrients before application of lime and soil amentments.
[1]Unamended, limed control. Statistics do not include preamendment values.

Table 4 Changes in minesoil heavy metal concentration between 1989 and 1990 (Amended minus Unamended). N=4. Least squares means ± standard error are shown.

	Lbs/A			
Amendment	Cu	Fe	Mn	Zn
Control	-0.09 ± 1.42	-16.4 ± 14.6	-12.4 ± 5.4	-0.7 ± 16
Phosphorus	0.04 ± 1.58	-19.0 ± 16.3	-9.0 ± 6.0	-0.5 ± 13
Sludge	8.90 ± 1.83*	-9.3 ± 18.9	-1.3 ± 6.9*	95.8 ± 21*
Sludge + P	3.23 + 1.58	-10.8 + 16.3	-1.8 + 6.0	27.2 + 13

*Significant differences in heavy metal concentration due to soil amendment based on Type III SS PR>F ≤ 0.05.

term benefits may be derived from the incorporation of sludge than from traditional fertilizers (Hani *et al.*, 1981; Larsen, 1981; Vogel, 1981; Wolt, 1985). Analyses of percent cover in this study support this contention (Figures 3 and 4). Vegetative cover was consistently greater on sludge and combination sludge + P plots than on P or unamended plots. Orchardgrass was more competitive than lespedeza species on the sludge and sludge + P combination amendments than on P. Timothy densities were greater than those of ladino clover on sludge amended plots. Vegetative growth and cover were improved on sludge plots relative to the other amendments with all seed mixtures, including the unseeded control (amendments and seed mixtures defined in Field Methods). The relatively high cover values on the unseeded sludge plots were due to high concentrations of weed seeds present in the Rockwood sludge. Unseeded controls on sludge and combination plots had greater vegetative cover than unseeded controls on unamended plots and P plots. Improved soil conditions of higher fertility, organics, and better moisture retention on sludge and combination plots encouraged the appearance and growth of volunteer plant species, including tree seedlings such as tulip poplar, maple, and cedar.

Conclusions

1. Minesoil acidity was reduced from preamendment levels by the application of lime. The P, sludge, and combination amendments further reduced acidity.
2. All nutrient levels were higher on the P amendment than on the limed control.
3. A greater quantity of nutrients was detected on minesoil amended with sludge than on the other amendments.
4. The combination sludge + P amendment exhibited a lower level of all nutrients and slightly less vegetative cover compared to the sludge amendment.
5. Lime application reduced heavy metal levels from preamendment levels.
6. A single application of 16 tons/a of municipal sludge did not add unacceptable levels of heavy metals.
7. The combination sludge + P amendment had lower heavy metal levels than the sludge amendment with the exception of iron.
8. The sludge amendment had greater vegetative cover than all other amendments in the first growing season.
9. Percent cover on the P, sludge and combination amendments increased, with less difference between amendments, in the second growing season. All amendments had greater cover than the control.

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A RECLAMATION APPROACH FOR MINED PRIME FARMLAND BY ADDING ORGANIC WASTES AND LIME TO THE SUBSOIL[1]

by

Qiang Zhai and Richard I. Barnhisel[2]

Abstract. Surface mined prime farmland may be reclaimed by adding organic wastes and lime to subsoil thus improving conditions in root zone. In this study, sewage sludge, poultry manure, horse bedding, and lime were applied to subsoil (15-30 cm) during reclamation. Soil properties and plant growth were measured over two years. All organic amendments tended to lower the subsoil bulk density and increase organic matter and total nitrogen. Liming raised exchangeable calcium, slightly increased pH, but decreased exchangeable magnesium and potassium. Corn ear-leaf and forage tissue nitrogen, yields, and nitrogen removal increased in treatments amended with sewage sludge and poultry manure, but not horse bedding. Subsoil application of sewage sludge or poultry manure seems like a promising method in the reclamation of surface mined prime farmland based on the improvements observed in the root zone environment.

Additional Key Words: sewage sludge, poultry manure, horse bedding, liming, corn, forage.

Introduction

The use of organic amendments for mine spoil reclamation has been extremely successful (Seaker and Sopper, 1984) because of its immediate improvement of soil chemical, physical, and biological conditions, acceleration of plant establishment and growth, and achievement of long-term productivity. However, the results may be different for prime farmland reclamation because the soil physical, chemical, and biological properties of prime farmland are different from mine spoils. Currently, there are few studies that have quantitatively measured the effect of organic waste application to reclaimed prime farmland on soil properties and crop response.

Amending subsoil with organic wastes, such as sewage sludge, and lime may eliminate major factors limiting plant growth. Low amounts of available essential nutrients and extreme acidity (Dancer and Jansen, 1987), have been implicated in limiting post-mine plant growth. Subsoil acidity may be and important yield limiting factor (Sumner et al., 1986). It restricts roots from penetrating deeply into such soils (Adams and Moore, 1983). Organic amendments to subsoil may also decrease bulk density. Subsoil compaction often occurs during soil replacement by heavy equipment. Studies on the rooting behavior of row crops in mined soils of southern Illinois (Grandt, 1988) showed the plant root system was confined to the topsoil layer as a result of the adverse physical and structural

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properties of the subsoil. This, in turn, resulted in greater weather sensitivity for corn, resulting in significant yield reductions. Numerous studies have been reported which indicated that subsoil compaction during soil reconstruction by heavy equipment was an important factor limiting the postmine productivity for certain crops, especially row crops (Barnhisel, 1988).

The objectives of organic wastes and lime applications to subsoil for prime farmland reclamation are: (1) to enhance soil organic matter content and nutrient cycling in disturbed subsoil; and (2) to eliminate major factors limiting plant growth by improving subsoil properties.

Materials and Methods

Field procedures

This prime farmland was mined by TDC Coal Co. in Webster County, western Kentucky. The topsoil and subsoil were stored as separate stockpiles for two years. The soil series was the Loring (fine-silty, mixed, thermic Typic Fragiudalfs). The original topsoil and subsoil were replaced separately in the summer of 1990.

Prior to the application of the organic wastes to subsoil, the area was divided into two equal blocks (121.6 x 36.6 m) and agricultural grade limestone was applied at rate of 22.4 Mg ha^{-1} to one block. This lime rate was needed to raise the pH of 30 cm of subsoil to 6.5. Sewage sludge (33.6 Mg ha^{-1}), poultry manure (22.4, Mg ha^{-1}), and horse bedding (112 Mg ha^{-1}) were applied as separate treatments to the subsoil (plot size 30.4 x 36.6 m) prior to topsoil replacement. An control plot with the same size was also included for comparison. The rate of 33.6 Mg ha^{-1} for sewage sludge was the upper limit based on current regulations of N applied for "land farming" under the specifications associated with "permit by rule". The rate of poultry manure was based on recommendations of N need for corn when added to the soil surface (Steele, et al., 1982). The application rate for horse bedding was calculated to provide the same amount of total N as the poultry manure. All organic wastes and lime were incorporated into 0-15 cm of the subsoil with a chisel plow. Then topsoil (15 cm) was placed on the prepared subsoil. The entire area was broadcast with a mixture of alfalfa (*Medicago sativa* L.), red clover (*Trifolium hybridum* L.), and tall fescue (*Festuca arundinacea* Schreb.) on 17 August 1990 at seeding rates of 10.2, 10.2, and 18.2 kg ha^{-1}, respectively.

In May 1991, the limed and unlimed blocks were each divided into two equal parts. One part was sprayed with a mixture of atrazine (4WDL) and 'Roundup' at rates of 3.0 and 1.8 L ha^{-1}, respectively, then corn (*Zea mays* L.) was planted at seeding rate of 59,000 kernels ha^{-1}, Corn was also planted the following year on 25 May. In 1993, the forage was killed by atrazine and 'Roundup' and entire area was planted to corn. Fertilizer was applied uniformly to corn each year. Nitrogen, P, and K fertilizers were surface broadcast at rates of 165, 120, and 120 kg ha^{-1}. These blocks were further divided into four subblocks (4.75 x 30.4 m) for multiple yield measurements and observations of soil preperties (not true replications). Subsoil (15-30 cm) was sampled using a tractor-mounted Giddings coring machine for determination of physical and chemical properties on 15 June of 1991, 8 July 1992, and 20 November 1993 for biological properties.

Soil analyses

Soil bulk density was determined by Blake and Hartge, except that the core size was 4.25 cm (1986). Water-holding capacity was determined using the method developed by Klute (1986).

Four separate samples were collected from the subblocks (4.75 x 36.6 m) within each waste treatment. The soil samples for chemical analyses were air-dried, and ground to pass a 2-mm sieve. Soil pH was determined with a pH meter on a 1:1 deionized water to soil mixture (McLean, 1982). Exchangeable Ca, Mg, K, and Na and CEC were determined with the neutral 1 M NH_4OAc method (Thomas, 1982). Available P was determined using the Bray-I (Bray and Kurtz, 1945) which has been commonly used in Kentucky for disturbed land. Organic matter (Nelson and Sommers, 1982) and total nitrogen (Bremner and Mulvaney, 1982) were performed by the University of Kentucky Soil Testing Laboratory, Division of Regulatory Services.

Soil microbiological analyses were performed using moist soils. Total heterotrophic and facultative anaerobic organisms were enumerated by the Most Probable Number method (Alexander, 1982). Microbial biomass (C) was determined using a chloroform fumigation procedure (Parkinson and Paul, 1982). Soil respiration rate (CO_2 evolution rate) in the laboratory was measured by gas chromatography on a Varian 3700 gas chromatograph equipped with Porapak Q column operated at 80°C (Rice and Smith, 1982).

Plant analyses

Forage tissue (five samples collected for each treatment) and corn ear-leaf (four samples collected for each treatment) were sampled and dried under vacuum at 65°C. These samples were ground with a Thomas-Wiley mill, pass a 0.425 mm sieve and stored in plastic bags. The samples were then analyzed for N, P, K, Ca, and Mg. Nitrogen and P in plant tissue were determined based on the micro-Kjeldahl method as presented by Jones and Case (1990). Potassium, Ca, and Mg were determined using the nitric-perchloric wet-ashing procedure (Jones and Case, 1990). Element concentrations were determined using an Instrumentation Laboratory S 11 Atomic Absorption Spectrometer.

Statistical analyses

Statistical analyses were performed using the Statistical Analysis System (SAS Institute, 1985). The General linear Model (GLM) procedure was used to obtain the analyses of variance.

Results and Discussion

Soil response

Selected physical and chemical properties of subsoil (15-30 cm) are shown in Table 1 for 1991 and Table 2 for 1992. Subsoil bulk densities were reduced by all organic treatments compared to check plots in 1991. The horse bedding treatment had a lower bulk density than the control in 1992 (two years after waste application) compared to other organic treatments. All three organic amendments increased the recent organic matter, total N, and Bray-P in 1991 and 1992. However, organic amendments generally did not affect water holding capacity, CEC, and exchangeable bases (Ca, Mg, K, and Na). Liming significantly increased exchangeable Ca, but pH and CEC were not significantly increased. Subsoil liming also tended to decrease exchangeable Mg and K, but there were no significant differences between limed and unlimed treatments.

Total soil heterotrophic and facultative anaerobic organisms, microbial biomass C, and respiration rates are given in Table 3. Total heterotrophic and facultative anaerobic organism population, biomass C, and respiration rate were still higher than the check after three years following waste application. The horse bedding treatment had significantly high values among these microbiological parameters. This was probably due to the high amount of carbon added to the subsoil

Table 1. Physical and chemical properties affected by organic wastes and lime application to subsoil (15-30 cm) on 15 June 1991.

Trt	BD† (g/cm^3)	WHC (%)	pH	OM (%)	TN (---mg/kg--)	Bray-P	CEC (--------	Ca	Mg	K	Na
							coml(+)/kg	---	------)		
SS	1.35^b	22.0^a	4.83^a	1.10^b	740^b	16.0^a	11.6^a	6.01^a	1.18^a	0.16^a	0.09^a
PM	1.30^b	20.7^a	4.99^a	1.41^b	766^b	21.6^a	12.1^a	6.19^a	1.38^a	0.20^a	0.11^a
HB	1.19^c	23.4^a	6.07^a	2.31^a	933^b	15.6^a	11.2^a	6.59^a	1.52^a	0.35^a	0.14^a
CK	1.55^a	24.5^a	4.51^a	0.84^b	466^b	3.4^b	11.8^a	5.08^a	1.03^a	0.17^a	0.09^a
UL	1.34^a	23.8^a	4.67^a	1.52^a	747^b	13.3^a	11.5^a	5.22^a	1.48^a	0.23^a	0.12^a
L	1.35^a	21.5^a	5.53^a	1.29^a	705^b	15.0^b	11.8^a	6.71^a	1.07^a	0.21^a	0.09^a

† BD, bulk density; WHC, water holding capacity; OM, organic matter content; TN, total nitrogen content; SS, sewage sludge; PM, poultry manure; HB, horse bedding; CK, check plot; UL, unlimed; and L, limed treatment. Comparison within the same column at $p<0.1$.

Table 2. Physical and chemical properties affected by organic wastes and lime application to subsoil (15-30 cm) on 8 June 1992.

Trt	BD† (g/cm^3)	WHC (%)	pH	OM (%)	TN (---mg/kg--)	Bray-P	CEC (--------	Ca	Mg	K	Na
							coml(+)/kg	---	------)		
SS	1.48^a	24.6^a	6.02^a	1.21^b	852^b	7.8^a	11.5^a	5.73^a	1.40^a	0.18^a	0.10^a
PM	1.51^a	23.4^a	6.84^a	1.37^b	930^b	14.2^a	12.0^a	6.74^a	1.51^a	0.22^a	0.14^a
HB	1.37^b	21.6^a	6.40^a	2.32^a	1052^b	13.2^a	11.2^a	6.38^a	1.73^a	0.22^a	0.12^a
CK	1.51^a	23.4^a	6.28^a	0.90^b	685^b	3.7^b	11.0^a	6.10^a	1.50^a	0.19^a	0.08^a
UL	1.43^a	22.5^a	6.21^a	1.64^a	997^a	13.1^a	11.2^a	5.73^b	1.71^a	0.28^a	0.07^a
L	1.51^a	23.9^a	6.56^a	1.26^a	762^a	13.4^b	11.7^a	6.74^a	1.36^a	0.22^a	0.14^a

† BD, bulk density; WHC, water holding capacity; OM, organic matter content; TN, total nitrogen content; SS, sewage sludge; PM, poultry manure; HB, horse bedding; CK, check plot; UL, unlimed; and L, limed treatment. Comparison within the same column at $p<0.1$.

Table 3. Total heterotrophic and facultative anaerobic organisms, microbial biomass, and respiration rates after organic wastes and lime application to subsoil.

Trt	Total Heterotrophs (10^5/g)	Facultative Anaerobes (10^5/g)	Microbial Biomass (C) (mg/kg)	Respiration Rate (mg CO_2/kg h)
Sewage sludge	2.4 b†	0.59 ab	117 b	15.7 b
Poultry manure	3.0 b	0.41 b	103 b	14.1 bc
Horse bedding	11.6 a	0.92 a	187 a	20.1 a
Check	0.1 b	0.06 b	75 b	12.4 c
Unlimed	3.2 a	0.46 a	105 a	15.5 a
Limedb	5.3 a	0.53 a	136 a	15.6 a

† Comparison within the same column at $p<0.1$.

that could have enhanced microbial activity and development. Second attributing factor may be the higher application rate (112 Mg ha^{-1}) compared to the rates used for sewage sludge (33.4 Mg ha^{-1}) and poultry manure (22.4 Mg ha^{-1}). Another possibility is that lower microbial activity in sewage sludge and poultry manure treatments is due to effect of time, these observations were made three years after the wastes were applied. The results would be different if the microbiological activities were observed soon after application of organic wastes. Decomposition and mineralization rates are usually faster, particularly for poultry manure than horse bedding, as most of the sewage sludge and poultry manure materials were mineralized after three years (Zhai et al., 1993). Subsoil liming did not significantly affect microbial activity. This is probably due to that microbial activity and development is usually confined below pH 4.5 (Tate, 1985).

Corn response

Results for corn grain yield, N removal, ear-leaf element concentrations are presented in Tables 4, 5, and 6 for 1991, 1992, and 1993, respectively. The grain yield and N removal were significantly affected by the different subsoil amendments. Yields were generally higher for sewage sludge and poultry manure treatments than for the check. The horse bedding treatment had the lowest yield over all three years. This is likely due to the high C:N ratio of horse bedding reduced N availability (Zhai et al., 1993). Corn plants in the horse bedding plot were more chlorotic, especially during the grain filling period. Reduced N availability in horse bedding treatment was also shown in an N mineralization-immobilization study in which net immobilization was observed (Zhai et al., 1993).

Average corn yields for all treatments were higher in 1992 than 1991 or 1993. The yields of all treatments in 1991 (Table 4) were far below 6.70 Mg ha^{-1} which is the critical value for Phase III bond release of a reconstructed prime farmland. This was due to heavy rain in planting time which reduced seed germination. The late planted corn in 1991 (two weeks later than normal) suffered moisture stress in the growing season, particularly at the grain filling stage during late July and August.

Table 4. Ear-leaf composition, grain yield, and N removal in 1991 as affected by organic waste application and liming of subsoil.

Trt	Ear-leaf N	Ear-leaf P	Ear-leaf K	Ear-leaf Ca	Ear-leaf Mg	Grain Yield	N Removal
	(g/kg)					(Mg/ha)	(kg/ha)
SS†	27.6a‡	2.69a	21.6a	5.89a	2.98a	3.91a	99.2a
PM	25.0a	2.39a	20.1a	5.96a	3.22a	3.35ab	77.1b
HB	20.4b	1.83a	20.3a	5.88a	2.56a	2.31c	58.7b
CK	22.7b	2.03a	19.1a	5.81a	2.97a	3.02bc	58.8b
L	24.4a	2.35a	21.1a	6.05a	3.14a	2.58a	63.1b
UL	23.5a	2.11a	19.5a	5.71b	2.72a	3.72a	83.8a

† SS, sewage sludge; PM, poultry manure; HB, horse bedding; CK, check; L, limed; UL, unlimed.
‡ Comparison within the same column at $p<0.1$.

The ear-leaf N concentration has been repeatedly used as an indicator of the N nutrition of the corn crop (Dirks and Bolton, 1980). Ear-leaf N concentrations found in this study were different, depending on organic amendment. Higher ear-leaf N concentrations were observed for sewage sludge and poultry manure treatments than for the unamended check and horse bedding treatments (Tables 4 and 5). The ear-leaf N concentrations in all organic amendment treatments were generally not much different from that of the check in 1993 (Table 6). Subsoil liming consistently improved ear-leaf N, though not significantly.

Table 5. Ear-leaf composition, grain yield, and N emoval in 1992 as affected by organic waste application and liming of subsoil.

Trt	Ear-leaf N	P	K	Ca	Mg	Grain Yield	N Removal
	(---------- g/kg				----------)	(Mg/ha)	(kg/ha)
SS†	25.7a‡	3.02a	20.2a	3.51a	2.13a	9.57a	ND
PM	23.6b	2.81a	23.7a	3.31a	1.63a	8.92ab	ND
HB	15.9c	2.20a	19.3a	3.03a	1.68a	7.19b	ND
CK	22.8b	2.55a	20.1a	3.14a	1.85a	8.11ab	ND
L	22.8a	2.84a	21.7a	3.40a	1.84a	8.30a	ND
UL	21.2a	2.46a	20.1a	3.09b	1.80a	8.68a	ND

† SS, sewage sludge; PM, poultry manure; HB, horse bedding; CK, check; L, limed; UL, unlimed; ND, not determined.

‡ Comparison within the same column at $p<0.1$.

Table 6. Ear-leaf composition, grain yield, and N removal in 1993 as affected by organic waste application and liming of subsoil.

Trt	Ear-leaf N	P	K	Ca	Mg	Grain Yield	N Removal
	(---------- g/kg				----------)	(Mg/ha)	(kg/ha)
SS†	21.8a‡	2.71a	20.2a	4.23a	2.10a	4.81a	74.0a
PM	21.7a	2.60a	19.5a	3.60a	1.19a	3.40b	57.1b
HB	19.7a	2.50a	19.0a	3.93a	1.95a	2.67b	45.3c
CK	22.0a	2.79a	19.5a	4.23a	1.68a	1.89c	25.8c
L	23.6a	2.87a	19.7a	4.55a	1.97a	3.18a	50.6a
UL	19.0a	2.44a	19.4a	3.45a	1.85a	3.21a	50.4a

† SS, sewage sludge; PM, poultry manure; HB, horse bedding; CK, check; L, limed; UL, unlimed.

‡ Comparison within the same column at $p<0.1$.

Average values for ear-leaf N were similar. Steele et al. (1982) reported critical values of 28.2 g kg^{-1} for ear-leaf N concentration. Dirks and Bolton (1980) found the level of leaf N was a good predictor of corn grain yields, and that the ear-leaf N concentration of adequately N fertilized corn had values in excess of 23 g kg^{-1}. Asghari and Hanson (1984) found higher critical values of ear-leaf N (29.3 g kg^{-1}). In this study, ear-leaf N for horse bedding ranged from 15.9 to 20.4 over three years. This suggests that a N shortage was probably a major yield limiting factor in this treatment.

Potassium, Ca, and Mg were not consistently affected by organic amendments. Subsoil liming increased P and Ca nutrition throughout the three-year period, though not significant. Potassium and Mg were essentially unaffected by liming.

Forage response

Results for forage yield, N removal, tissue element concentrations for first cutting in 1991 and 1992 are presented in Tables 7 and 8, respectively. Because the results of chemical composition were similar to those of second and third cuttings, only first cutting data was presented each year. Yields were generally higher for the sewage sludge and poultry manure treatments than that for the check, but the horse bedding treatment was usually not significantly different from the check across harvests in both 1991 and 1992. Subsoil liming did not significantly affect forage yield and N removal although limed treatments had slightly higher yield and N removal.

In 1991, higher tissue N concentrations were observed for sewage sludge and poultry manure treatments than check plots (Table 7). There was not much difference between the horse bedding treatment and the check plot. Similar results were also observed for the harvest in 1992 (Table 8). Phosphorus concentrations increased in poultry manure and sewage sludge treatments as compared to the check in the first year, but were not as significant as for N. There was not

much difference in P, K, Ca, and Mg concentrations between the horse bedding and the check treatments. Potassium, Ca, and Mg were not consistently affected by organic amendments throughout the study period.

Table 7. Forage tissue, yield, and N removal of first harvest in 1991 as affected by organic waste application and liming of subsoil.

Trt	Forage Tissue N	P	K	Ca	Mg	Forage Yield	N Removal
	(---------- g/kg	----------)				(Mg/ha)	(kg/ha)
SS†	29.7a‡	2.77a	22.7a	2.70a	2.81a	6.36a	189.7a
PM	27.3b	2.34a	19.4a	2.72a	2.73a	5.99a	160.7b
HB	24.9c	2.23b	21.1a	2.70a	2.66a	4.77b	120.1c
CK	25.2c	2.17b	20.2a	2.68a	2.65a	5.22b	130.1c
L	26.8a	2.87a	21.1a	2.83a	2.59a	5.72a	153.8a
UL	26.7a	2.44a	20.5a	2.55a	2.84a	5.45a	146.5a

† SS, sewage sludge; PM, poultry manure; HB, horse bedding; CK, check; L, limed; UL, unlimed.
‡ Comparison within the same column at $p<0.1$.

Table 8. Forage tissue, yield, and N removal of first harvest in 1992 as affected by organic waste application and liming of subsoil.

Trt	Forage Tissue N	P	K	Ca	Mg	Forage Yield	N Removal
	(---------- g/kg	----------)				(Mg/ha)	(kg/ha)
SS†	18.1a‡	3.02a	24.5a	3.19a	2.25a	6.41a	118.3a
PM	20.2ab	3.07a	25.1a	3.80a	2.11a	6.66a	135.1a
HB	13.3b	2.97a	25.9a	3.35a	2.00a	5.66b	75.2 b
CK	13.8b	2.85a	23.8a	3.70a	2.20a	5.97b	82.8 c
L	17.2a	3.11a	24.4a	3.73a	2.10a	6.32a	110.3a
UL	15.5b	2.84a	25.3a	3.34a	2.18a	6.03a	95.2 a

† SS, sewage sludge; PM, poultry manure; HB, horse bedding; CK, check; L, limed; UL, unlimed.
‡ Comparison within the same column at $p<0.1$.

Subsoil liming slightly increased forage tissue N concentrations in 1991 (Table 7). However, liming effects were not significant. Subsoil liming always increased Ca, but did not consistently affect P, K, and Mg.

Overall, the sewage sludge, poultry manure, and horse bedding applications to subsoil tended to lower the subsoil bulk density and increase the soil organic matter and total nitrogen, as well as promoted microbial activities. Surprisingly, subsoil liming produced small increases in subsoil pH and exchangeable Ca, decreased exchangeable Mg and K, and these effects usually were not significant as reported by Sumner et al. (1988). Sewage sludge and poultry manure treatments showed higher N nutrition and yield for corn and forage. Results from this study suggest that subsoil amendments of sewage sludge and poultry manure to reclaim prime farmlands are a beneficial practice in terms of improving some soil properties, particularly N and P status, as well as plant N nutrition and yields.

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AMERICAN SOCIETY • SURFACE MINING and RECLAMATION •

APPLYING GEOMORPHOLOGIC PRINCIPLES TO RESTORE STREAMS IMPACTED BY SURFACE MINING[1]

by

Michael S. Ellison[2]

ABSTRACT. The combination of geomorphic principles and native material restoration techniques provides a viable alternative to traditional engineering approaches to restore rivers and streams affected by surface mining. Channels can be designed to reflect ranges of stability known to occur in natural streams for measurable parameters such as bankfull width, depth, gradient, meander radius, sinuosity and entrenchment. Stable channel geometry reduces stresses on the stream bed and banks and eliminates the need for channel lining. Methods to utilize native materials have been developed and refined to stabilize stream channels constructed to appropriate dimensions until planted riparian vegetation develops mature root systems. These native materials include root wads, willow bundles, and boulders. These methods result in improved wildlife habitat in and around channels that maintain equilibria between sediment supply and sediment transport, and between erosional and depositional rates and patterns. Two streams in Baltimore County, Maryland were disturbed during mining operations and are being restored using this approach. Goodwin Run had been channelized to allow quarrying of the Cockeysville Marble. Approximately 1100 feet of stream were restored in the fall of 1992. White Marsh Run has been channelized and relocated several times to facilitate sand and gravel mining between an urbanized area and sensitive habitats of the Chesapeake Bay. The design of the White Marsh Run Restoration Project incorporated refinements to techniques used at Goodwin Run, and entails the restoration of over 5000 feet of stream and adjacent wetland habitat.

Additional Key Words: Stream Classification, Channel Geometry, Native Materials.

Introduction

Stream channel morphology is a sum of function and process that reflects the influence of physical laws. The organized nature of stream morphology is expressed by natural stream channels, and the consistent relationships among measurable variables allow streams to be classified based on delineative criteria. The classification system developed by Rosgen (1994) allows predictions of a river's response to changes in its watershed that may result from surface mining.

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Prior to passage of the Surface Mining and Reclamation Control Act of 1977 (SMCRA), rivers were frequently relocated to acquire access to mineral deposits. While the current regulatory climate discourages this practice, streams continue to be impacted by surface mining operations. The cumulative effects of increased runoff, decreased infiltration, and increased sediment load, disturb the dynamic equilibrium of stable streams, resulting in adjustments of channel geometry and pattern.

Recognition of the potential effects to streams from increased discharge and sediment load requires an understanding of the normal characteristics of channels, fluvial processes, and probable reactions to imposed changes (Dunne and Leopold, 1978). Given such an understanding, measures can be prescribed to restore river function and process. By applying proven geomorphologic principles, the river environment can be returned to a more natural state.

English units of measure are used herein for two reasons. First, all of the river measurements published by the U.S. Geological Survey, and therefore the vast majority of the literature, are published in English units. Second, as a result, most of the hydrologists involved in stream restoration are more accustomed to this system.

The Nature of Streams

A river or stream functions as a conveyance system for water and sediment. The system is powered by the process of energy transformation in which the potential energy of elevation is transformed into the kinetic energy of flowing water and sediment. In a natural state, a condition of dynamic equilibrium exists in which the sediment supplied by the watershed is efficiently carried by the stream, and erosion rates within the active channel are balanced by approximately equal rates of deposition. Owing to the universality of physics, channel morphology is highly organized and similar for streams of the same size in comparable climates (Dunne and Leopold, 1978).

Bankfull flow corresponds to the stage that shapes and maintains channels, and has been shown to have a recurrence interval of 1.5 years (Leopold et al, 1964). It is the most effective discharge in that it transports the largest amount of sediment. Larger flood events may move more material, but they occur so infrequently that, over time, they do not accomplish as much as bankfull flows (Leopold, 1994). It is the bankfull flow that is most responsible for forming or removing bars, forming or changing bends and meanders, and generally doing the work that results in the average morphologic characteristics of rivers (Dunne and Leopold, 1978). These average morphologic features, which can be identified and measured, are the manifestations of the physical processes which must be understood in order to understand of a given reach of stream.

Channel geometry describes the physical size, shape, and characteristics in relation to the hydraulic factors of velocity, roughness, slope, and flow frequency (Dunne and Leopold, 1978).

Shear stress is the eroding force per unit area exerted by streamflow. The form assumed by a stable channel is such that shear stress at every point on the wetted perimeter is approximately balanced by the resisting forces of the stream bed or bank (Leopold, 1994).

Stream pattern morphology refers to the form taken by a river in its valley and is directly influenced by eight variables: channel width,

depth, velocity, discharge, slope, roughness of channel materials, sediment load, and sediment size. The independent variables are discharge and sediment load. A change in any one of these variables sets up a series of channel adjustments which lead to a change in the others, and the result is an alteration in stream pattern morphology (Rosgen, 1994). The response among the dependent variables to changes in discharge or load is highly consistent from one location to another in a given river system, and from one river to another, and are all factors related to the dissipation or conservation of energy, and to the distribution of energy expenditure (Leopold, 1994).

One of the most obvious morphological features is the stream meander, and the lateral migration of stream meanders is a very important and readily observed process. Meandering represents the condition that channels approach as erosion and deposition proceed. Stable channels maintain a constant cross section as lateral migration occurs, with the quantity of material eroded from the cut bank being approximately equal to the amount of deposition on the next point bar downstream on the same side of the channel (Leopold, 1994).

Meanders are a good example of the predictability of river behavior that is so important in the restoration of disturbed systems. The relation of bankfull width and radius of curvature to meander wavelength is linear through five orders of magnitude (Leopold, 1994). Rosgen (1993) has demonstrated that the meander radius of stable streams is 2.2 to 2.8 times the bankfull width. Leopold (1994) states that the meander radius of 2.3 times the bankfull width is the ratio of least friction loss, and manifests the conservation of energy or tendency toward minimum work. Having once established a meandering pattern, a stream will not change to a straight pattern as long as the climate does not change (Dunne and Leopold, 1978). Therefore, the relationships between bankfull width and meander radius and wavelength must be maintained or re-established for effective restoration.

Stream Classification

Scientists have long sought to define a categorical order for streams, and a multitude of descriptive and theoretical schemes are documented in the literature. One of the more recent classification systems is that of David Rosgen, who analyzed data from over 450 rivers in the United States, Canada, and New Zealand and developed a methodology based on the concept that current channel morphology is governed by the laws of physics through observable stream channel features and related fluvial processes (Rosgen, 1994). Because Rosgen brought together quantitative relationships into a readily understood, meaningful framework, his system has gained rapid acceptance across a variety of disciplines. The Rosgen Classification System is now the most widely accepted manner of describing a channel (Leopold, 1994).

Rosgen provides a staged classification scheme with four levels of detail that can be applied commensurately with the objectives and skills of the user. The system organizes data from measurable stream features into combinations of delineative criteria (Figure 1), while at the same time recognizing a continuum within and between the various stream types. The end result is 42 major stream types determined by (in order) entrenchment ratio, width/depth ratio, sinuosity, slope range and channel material particle size (Figure 2).

The entrenchment ratio provides a quantitative means of describing the interrelationship of a stream to its

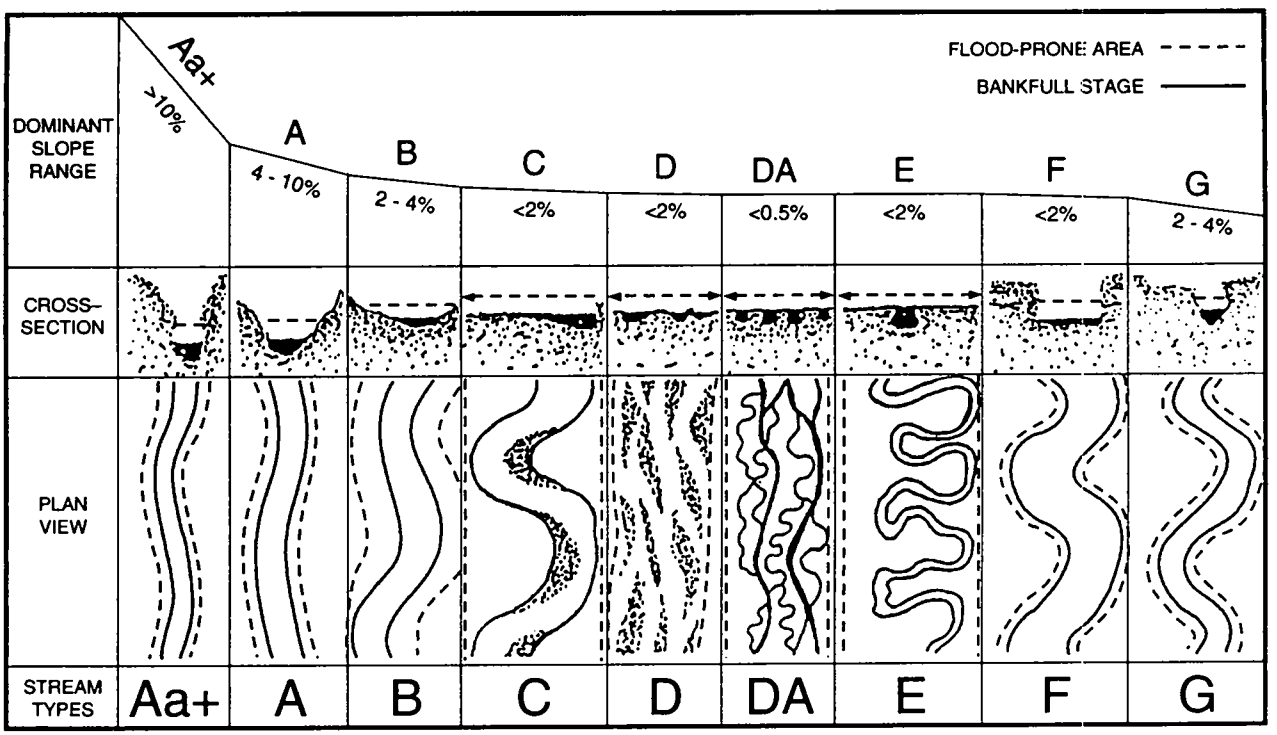
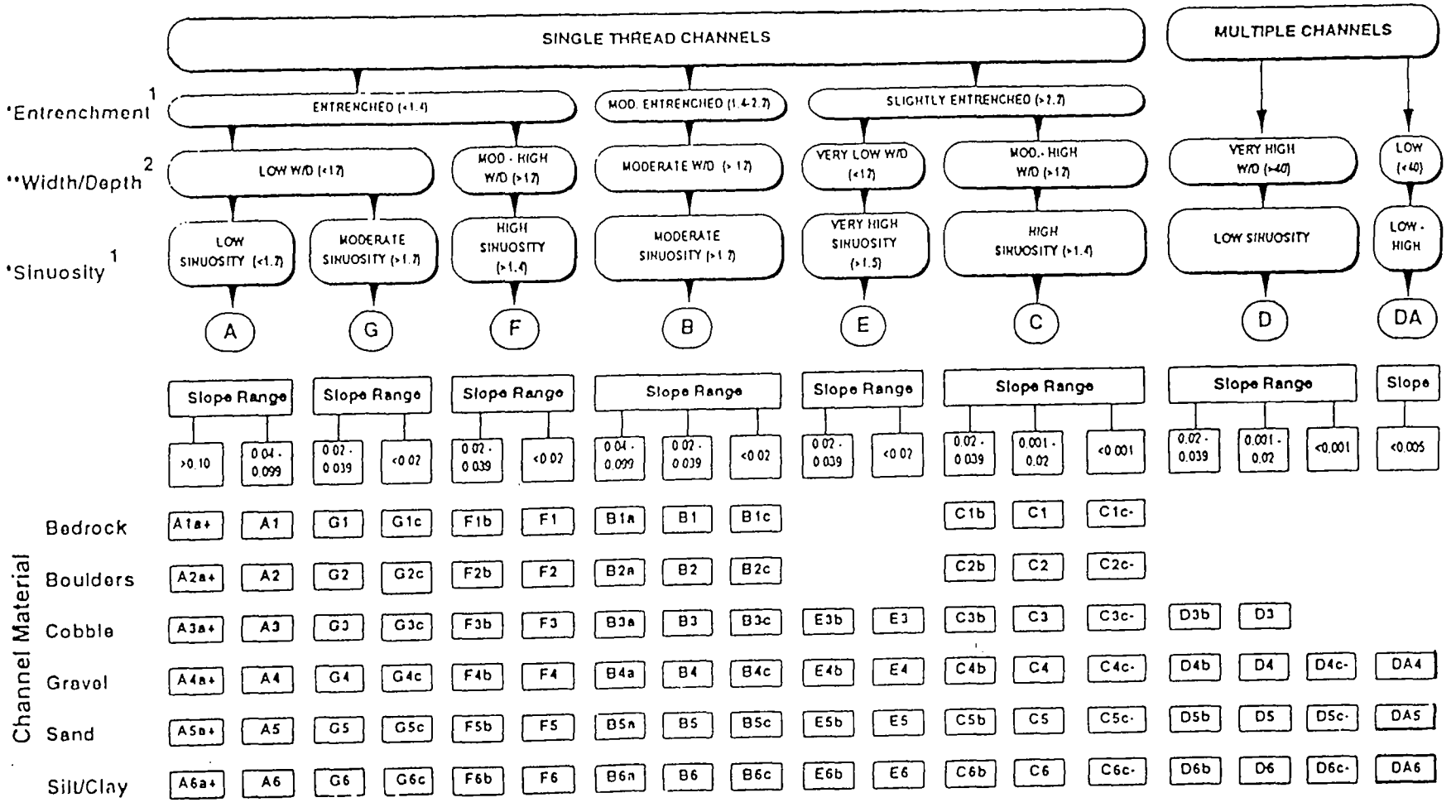


Figure 1. Longitudinal, cross-sectional, and plan views of major stream types (from Rosgen, 1994).

valley and/or adjacent landforms. It is the ratio of the width of the floodprone area to the bankfull width of the channel (Rosgen, 1994). Rosgen defines the floodprone area as the width measured at an elevation which is determined at twice the maximum bankfull depth (Figure 3). This is a relatively frequent flood elevation (50 year return period or less) and helps distinguish whether the flat adjacent to a channel is active floodplain, terrace, or outside a floodprone area.

The width/depth ratio is simply the bankfull width divided by the average bankfull depth, and is usually calculated from field measurements. Sinuosity is the quotient of channel length divided by valley length. Slope refers to the water surface slope in the center of the channel at the bankfull stage, and should be measured over a length of at least twenty channel widths. Channel material particle size is the median intermediate diameter of particles in the bed and banks of the active



1 Values can vary by ± 0.2 units as a function of the continuum of physical variables within stream reaches.
2 Values can vary by ± 2.0 units as a function of the continuum of physical variables within stream reaches.

Figure 2. Streamtype delineation key (from Rosgen, 1994).

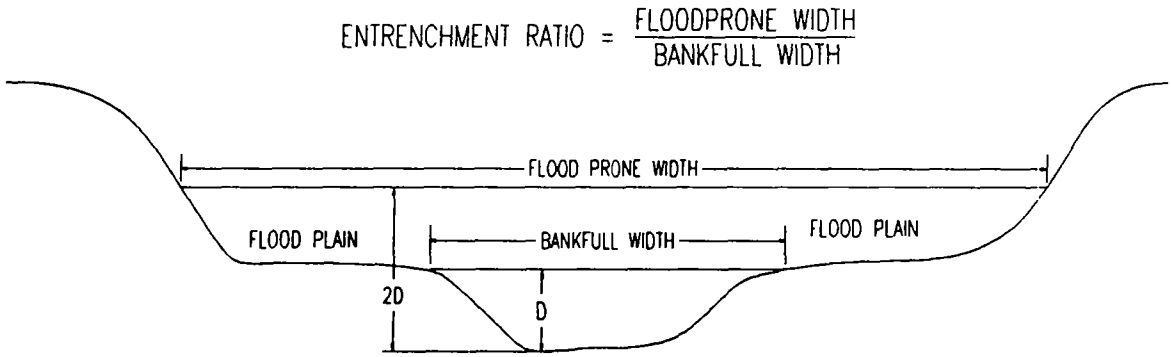


Figure 3. Determination of Entrenchment Ratio.

channel.

One of the primary objectives of this classification system was to be able to predict a river's behavior from its appearance, with a major application being the restoration of disturbed streams (Rosgen, 1994). The system was designed to allow interpretations regarding modes of adjustment-- either vertical, lateral, or both-- and energy distribution to be inferred through stream types (Rosgen, 1994). Rosgen and others have used this system to evaluate changes in channel geometry that have resulted, or would most probably result, from hydrologic and sediment load changes within watersheds.

Since its adoption by the USDA Forest Service, U.S. Fish and Wildlife Service, and many state and local government agencies, the Rosgen system has been successfully applied to many hundreds of streams across North America and found equally valuable for the evaluation of ephemeral, intermittent and perennial streams. Rosgen's system is a Classification of *Natural* Rivers (emphasis added) intended for use on individual reaches rather than entire stream systems or watersheds. Under natural conditions a given stream may vary in character, and therefore class, even through short distances downstream, as a result of passage from one lithologic type to another, tributary entrance, or change in landscape character (Leopold, 1994).

Stream Disturbance from Surface Mining

The overwhelming consensus in the literature is that surface mining increases the runoff and sediment supply to receiving streams, which, as discussed above, will initiate alterations in channel pattern. Changes in vegetative cover are a primary cause of the changes in hydrologic and sediment regimes that destabilize streams. In addition, alterations to soils, geology, and topography all contribute to erosion and affect the amount, timing and location of water reaching streams. Channel relocations to facilitate mining operations are another major cause of river instability, and will be discussed in a subsequent section.

Clearing vegetation increases runoff volume and decreases lag time, thereby increasing peak flows from a given storm event (Dunne and Leopold, 1978). Drainage diversions around active mine sites can result in the premature channelization of runoff from upslope undisturbed areas, increasing times of concentration. Toy and Hadley (1987) describe higher peak flows in mined watersheds compared to unmined watersheds, and discuss increases in volume, depth, and velocity of runoff, as well as increased rates of delivery to receiving streams as a result of reduced soil infiltration capacity.

Elliott (1990) and others have correlated increased bulk densities in reclaimed soils with reduced infiltration rates. Toy and Hadley also provide a good synopsis of several studies that compared erosion rates and sediment yield from surface mined lands to natural sites.

Elliott (1990) studied areas reclaimed under current SMCRA regulations that were still affected by relatively rapid erosion rates several years after conclusion of reclamation activities. He found erosion rates can remain high after establishment of vegetation due to steep slopes or narrow valley floors. Elliott identified two geomorphologically distinct areas of erosion: rills with occasional gullies on hillslopes, and gullies/unstable stream channels on valley floors. While both conditions can provide excess sediment to streams, gullies on reclaimed valley-floors have the potential to rapidly mobilize substantial volumes of soil and spoil, thereby inducing channel instability downstream. In addition, local base-level lowering can propagate headcuts upstream and into nearby tributaries (Elliott, 1990).

Surface mining also tends to remove lithologic and structural controls on the drainage network as consolidated lithologic units are fragmented, removed, and replaced when operations proceed across valleys (Elliott, 1990). The destruction of bedrock also results in decreased material strength and resistance to erosion and mass wasting (Toy and Hadley, 1987). Because geomorphic process rates are accelerated during and for some time after mining operations, valley geometry and hillslope stability become important factors in long term stream stability.

In evaluating stream impacts from mining, the elements of proximity and direction need consideration. Due to the interconnected nature of drainage networks, many effects may not be fully observed for some great distance from where change occurred (Dunne and Leopold, 1978). Different stream types adjust to stress in different ways, and some, such as most "A" type and some "B" type channels, are more forgiving of hydrologic changes, and significant channel disturbance may not be manifest until there occurs a change in stream type down valley. Additionally, the concept of headward erosion implies that all impacts may not be downstream of the mine site.

Traditional Remedies: The Engineering Approach

Historically, streams destabilized as a result of mining operations have been neglected. Dredging often occurs when mass wasting or excessive sedimentation cause flooding problems, and various methods of armoring have been applied to protect bridges or other structures threatened by erosion, but, in general, the restoration of river function and habitat is a relatively new concept. This is probably due, at least in part, to the lack of understanding of fluvial processes and the downstream ramifications of disturbance in the watershed.

In addition, the physical relocation of rivers has typically resulted in straight ditches designed for maximum efficiency of water movement and covering the shortest distance possible (Thompson, 1985). This practice, generally referred to as channelization, usually raises the severity of erosion and deposition problems. Without the appropriate channel geometry, a river will not remain passively in its newly assigned place, but will instead respond immediately with alterations among hydraulic variables to return to its pre-disturbed pattern (Dunne and Leopold, 1978).

In addition to the instability

caused by adjustment to imposed conditions, channelization typically has downstream effects, especially increased bank erosion and aggradation or degradation of the bed (Dunne and Leopold, 1978). One example, though unrelated to mining, frequently mentioned in the literature (Dunne and Leopold, 1978; Toy and Hadley, 1987; Leopold, 1994) is the Blackwater River in Missouri. The river was shortened by the elimination of meanders, which increased the slope of the channel and has caused over 60 years of continuous downcutting.

Artificial channels are often sized to contain flood events well in excess of the bankfull flow, and are typically designed to have a wide, relatively flat cross section. These conditions typically cause aggradation because the increase in width/depth ratio reduces the shear stress and entrainment capacity of the stream at any flow (National Research Council, 1992). In addition, the bed and banks are routinely stabilized with rip-rap, gabion baskets, or concrete, which limit the development of aquatic and riparian habitats and are devoid of aesthetic value.

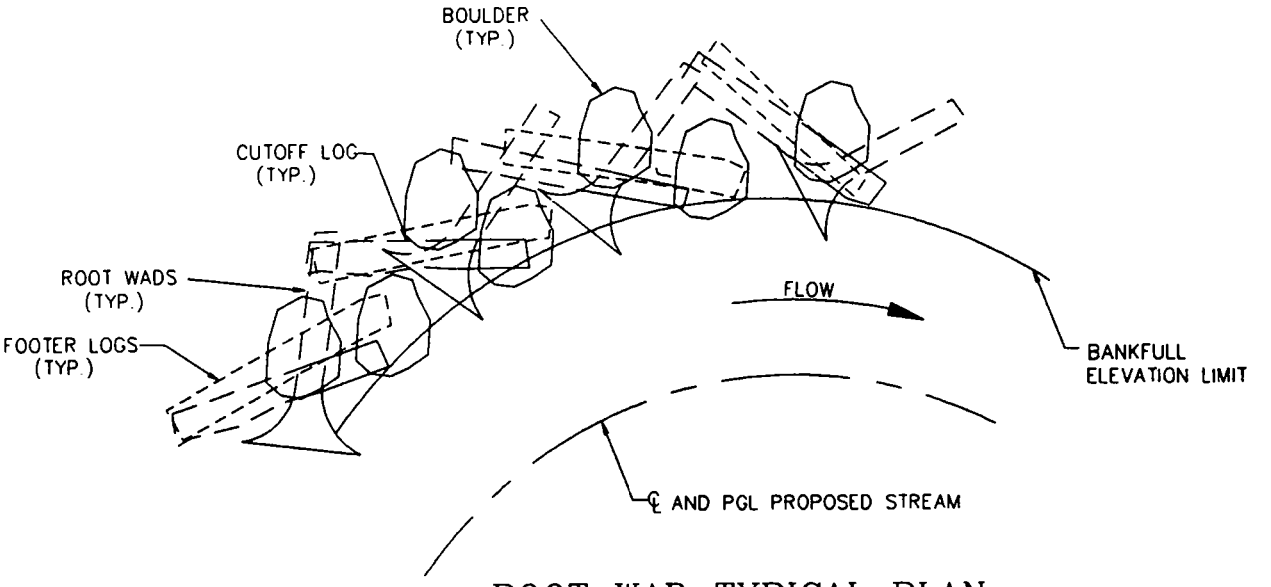
Geomorphology and Native Materials: An Alternative Approach

The objective of stream restoration is to restore the dynamic equilibrium of the physical system, which will establish and support the biological components (National Research Council, 1992). This is accomplished by establishing the channel geometry and pattern that are appropriate for the bankfull discharge and sediment load, and consistent with the morphological features of the valley.

The Rosgen classification system provides a framework for determining the most probable form of a stream. The delineative criteria reflect ranges of stability that can be used to calculate design channel dimensions based on streamtype. The morphology of the valley is the primary determinant of the appropriate streamtype because of the empirical relationships between a stream and its valley. Narrow valleys will dictate moderate to high entrenchment, while broader valleys allow greater sinuosity and a more developed floodplain. In addition, the valley slope divided by sinuosity equals the channel slope, which allows the integration of potential vertical and lateral constraints in the restoration design.

Other relationships between the valley-floor width, valley slope and drainage area are described by Elliott (1990). Elliott suggests that stream channel stability can be significantly improved if these parameters are replaced to appropriate values during the reclamation process.

If the channel geometry of a river is restored to a stable form, shear stress will be minimized and excessive armoring of bed and banks is unnecessary. This eliminates the expense of importing large quantities of rock or concrete. Instead, stream banks can be effectively stabilized using native material revetments such as rootwads and willow bundles (Figures 4 and 5, respectively), which can often be salvaged close to the site. These materials allow the construction of vertical banks on the concave side of meander bends, unlike rip-rap, which must be placed at slopes of 1:1 or less. In addition, they provide organic matter for benthic organisms and can be installed to provide cover for fish. Willow cuttings are typically planted in and around rootwads to provide permanent bank stabilization after the rootwads decay. Native materials facilitate this establishment of riparian habitat to the edge of the active channel, with the end result being a more natural stream in terms of aesthetics and function.



ROOT WAD TYPICAL PLAN

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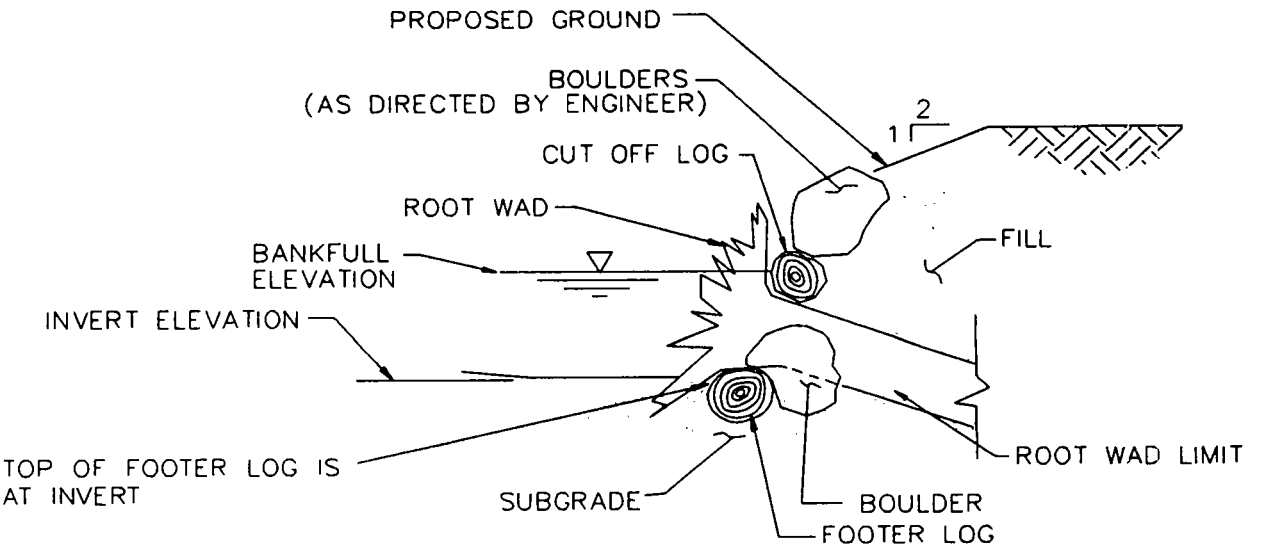

ROOT WAD REVETMENT TYPICAL SECTION

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Figure 4. Root wad revetment for concave side of restored meander.

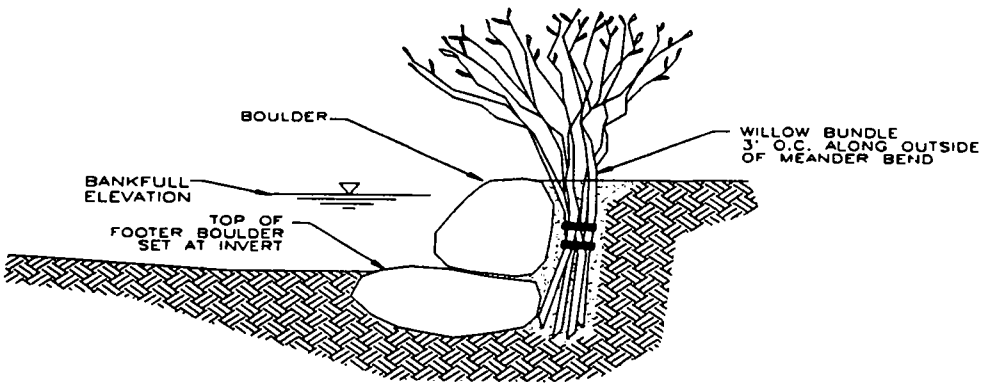


Figure 5. Willow Bundle revetment for concave side of restored meander.

Vortex rock weir structures (Figure 6) were developed and refined by David Rosgen to provide habitat enhancement and grade control without inducing backwater conditions. In addition, the upstream pointing "V" configuration directs flow into the thalweg, thereby reducing stresses on the near bank regions. Footer rocks are set at the channel invert elevation behind the vortex rocks to break the hydraulic jump of flow over the vortex rocks. These structures are typically placed at intervals of about every two to three channel widths, and at the points of curvature in and out of meander bends.

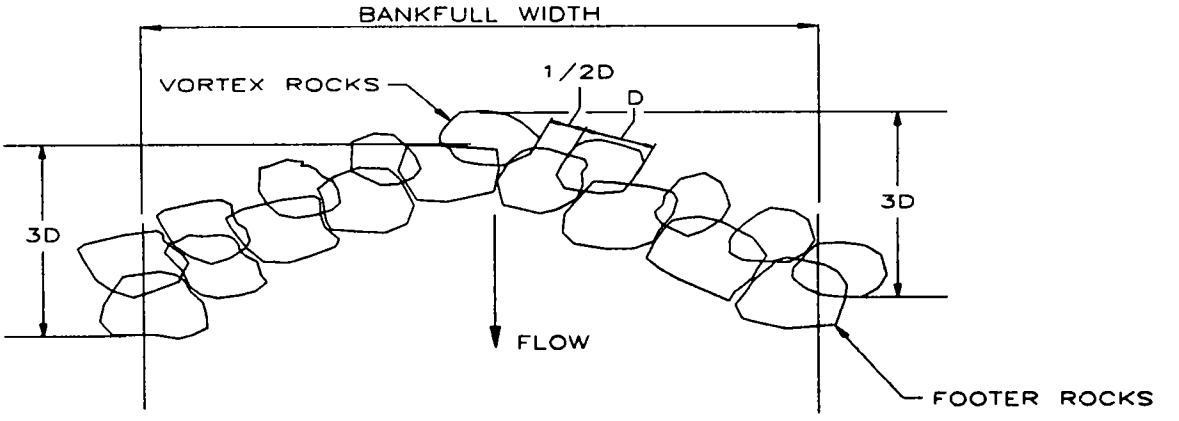
Examples

Goodwin Run

Goodwin Run is a second order tributary to Beaverdam Run, a significant trout fishery in the Piedmont physiographic province of north-central Baltimore County, Maryland. Near the confluence of its two headwater tributaries, Goodwin Run flows through a large quarrying operation where the Cockeysville marble has been extracted for decades. Despite several channel relocations over the years, one of the tributaries sustains a good brook trout population, and brown trout have been known to inhabit the main stem. The stream had been targetted for restoration as part of a county-wide stream improvement program, and funding was aquired when the Maryland State Highway Administration required compensatory mitigation for stream and wetland impacts related to a road project that would cross the stream.

Prior to restoration, the channelized stream was classified using the Rosgen system as predominantly a "G" type channel with some reaches of the "F" streamtype (Gracie, 1995). In addition, some sections of the stream had been lined with rip-rap. Severely eroding banks were providing excess sediment to downstream reaches and to Beaverdam Run, with substantial aggradation occurring near a railroad crossing just upstream of the confluence. The objectives of the restoration effort were to arrest the bank erosion problems by installing natural channel geometry, and to improve fish habitat by using native material revetments.

Measurements of existing conditions were collected to allow determination of the bankfull discharge and other design criteria. The design of the new stream was based on a C4 stream type in the Rosgen classification system (McGill and



VORTEX ROCK WEIR (PLAN)

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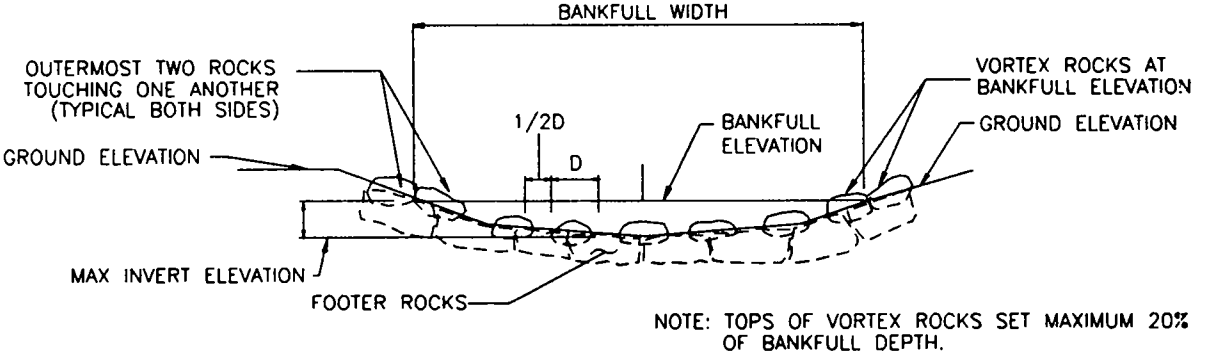

VORTEX ROCK WEIR (CROSS SECTION)

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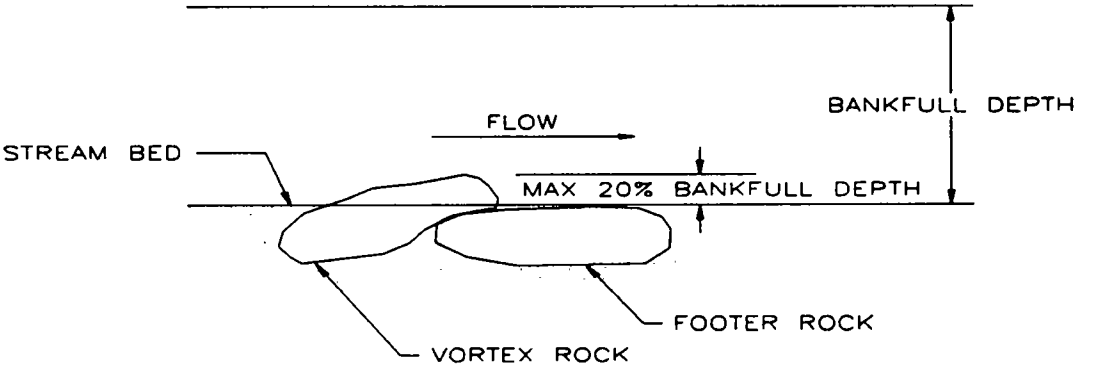

VORTEX ROCK WEIR (PROFILE)

NOT TO SCALE

Figure 6. Vortex Rock Weir structure for grade control and habitat enhancement.

Gracie, 1993). The design bankfull width was 28 feet and the average bankfull depth was 2 feet. The maximum depth in pools was 3.6 feet. Rootwads were used to reinforce meander bends and vortex rock weirs to provide grade control and direct flow into the center of the channel. Approximately 1100 feet of river were restored.

Construction was completed in September of 1992, and within a week a tropical storm caused flows to exceed the bankfull stage. The stream has since received several bankfull events and has generally held up well. Monumented cross sections allow consistency in annual monitoring of channel geometry, and sample data are provided in Figure 7.

Since construction, there have been some minor adjustments in the stream that are instructive. First, there has been some erosion of point bars on the inside of meander bends. The primary cause of this was a directive from a regultory authority that required the application of a stabilization seed mix on the point bars within the the limits of the active channel. After the grass was established, channel capacity was reduced and roughness was increased. The river adjusted by eroding enough material from unvegetated portions of the point bars to regain its capacity. The stream was designed to have a width/depth ratio of 14. If a width/depth ratio at the higher end of the stable range for this streamtype had been used, the amount of this erosion would most likely have been less.

Another problem has been aggradation in and around the rock weirs. There probably was not enough displacement to the apex of the weirs; in other words, the distance in the center of the channel from the wide end of the `V' to the upstream-most rock at the point was insufficient. Design modifications by Rosgen now indicate that this displacement should be 2.5 to 3 times the average intermediate diameter of the boulders. In addition, the vortex rocks supplied by the contractor were too big. Although correctly installed with a 0.3 to 0.5 diameter space between rocks, the large gap appears to have inhibited proper functioning of these devices.

The last problem was with the rootwad revetments. Rootwads are installed such that the bole of the tree is oriented parallel with the incoming flow, which usually sets the root fan normal to flow (Figure 4). The cut-off log is placed on top of the bole and oriented downstream and out toward the next rootwad. The cutoff log is secured with a single boulder and backfill material. The purpose of the cutoff log is to break up eddy currents that can develop when bankfull flows come around the meander. To function effectively, it is critical that the elevation of the tops of the cutoff logs be above the bankfull elevation. At Goodwin Run, most of the cutoff logs were only about twelve feet long. The result was that some of the logs were not long enough to be adequately buried in the bank. Of over 100 cutoff logs installed, 3 have been torn out and 4 or 5 others show evidence of buoyancy during high flows.

<u>White Marsh Run</u>

White Marsh Run has been channelized and relocated several times to facilitate extensive sand and gravel mining between a heavily urbanized area and sensitive habitats of the Chesapeake Bay. Due to the mitigation requirements of another highway project, approximately forty acres of wetlands will be created and over 5000 feet of stream will be restored. The existing stream is an F4 streamtype that is fully entrenched by steep banks.

The discharges used for the

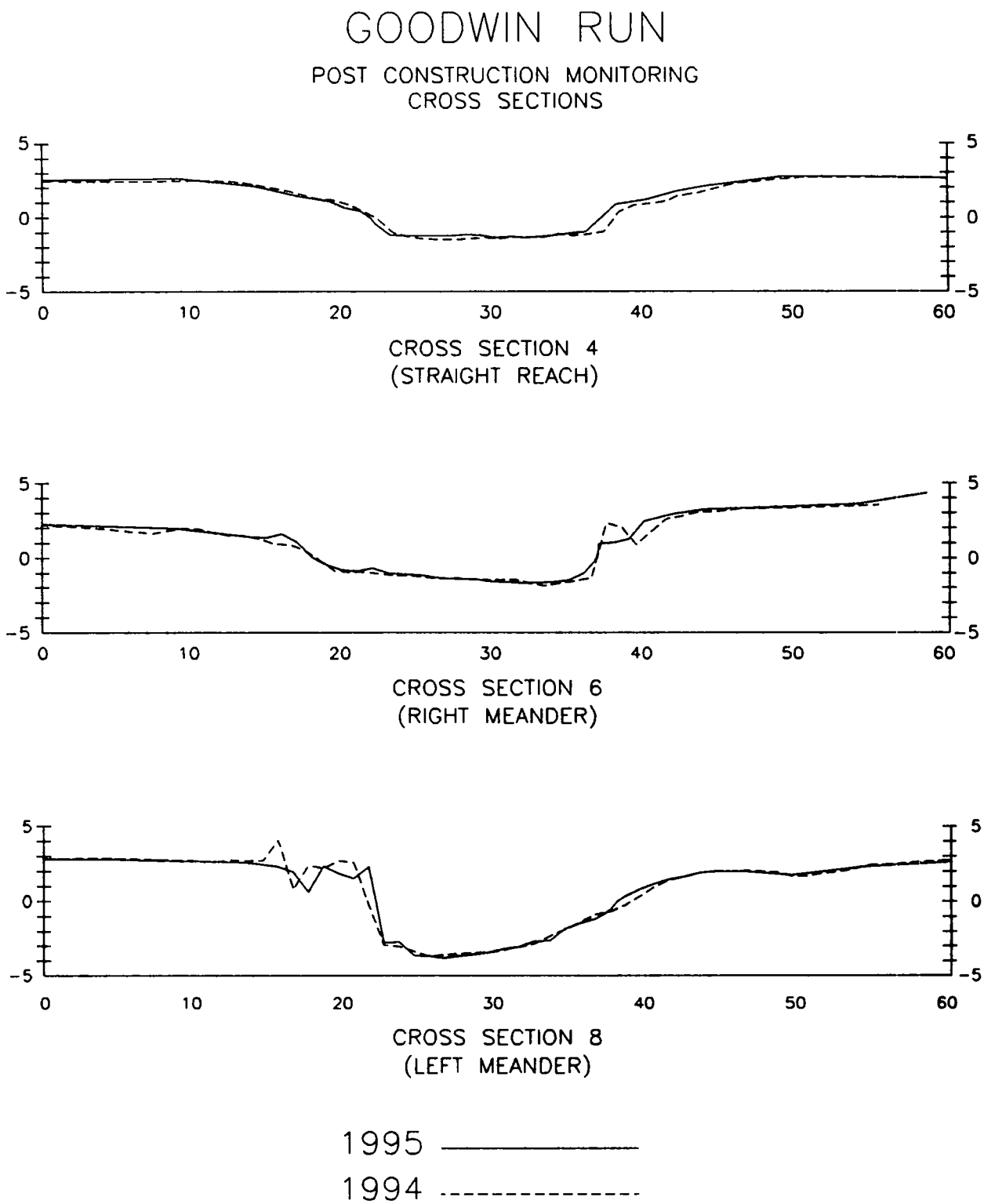

Figure 7. Goodwin Run sample cross section monitoring data.

restoration design were determined from field measurements and checked by stage discharge relations from gage station data. The upstream portion of the project was designed for a bankfull flow of 250 ft^3/second, with 300 ft/second used for the downstream section due to tributary entrance and increased drainage area. The restoration plans again call for a C4 streamtype with rootwad revetments on meander bends. The width/depth ratio will be 22. The cutoff logs are specified to have a minimum length of 16 feet. The plans also include the revised vortex rock weir design shown in Figure 6. Construction was started in March of 1996 and is expected to take 6 months to complete.

Summary

Geomorphologic principles can be applied to streams disturbed by surface mining to restore channel geometry to a stable form. By integrating knowledge of fluvial process, stream pattern morphology, channel and meander geometry, and the natural tendencies of adjustment toward stability, the most effective design for long-term stability and function can be predicted. The Rosgen Classification System provides a methodology for the determination of the design parameters needed for a natural channel design. By defining design parameters within the ranges of stability for natural streams, and by respecting the relationships between the eight variables, streams be returned to a self-maintaining condition of dynamic equilibrium.

By restoring this stable condition, a stream will tend toward uniformity in the rate of energy expenditure and minimum work, therefore exerting the least stress on channel bed and banks. The minimization of stress allows native materials to support and complete restoration to a natural condition.

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REINTERPRETING SMCRA: "PERMITTING" PHASED POSTMINING LAND USE

by

Zina R. Merkin and Thomas J. Nieman

Abstract: The coal producing area of Appalachian Kentucky has a shortage of developable land. The majority of mined land in this region has been reclaimed to pastureland or hayland, while narrow interpretation of the Surface Mining Control and Reclamation Act of 1977 (SMCRA) and regulations, especially regarding bond release, has limited alternative postmining land uses which could support economic development. A study of Federal and State of Kentucky laws and regulations shows that postmining land use regulations and their implementation have focussed on preventing and minimizing environmental damage. Land use and land use planning concepts are not well understood, thus permit applications inadequately address land use needs and the "highest and best use" of a site. Required information about pre-mining conditions is not collected and analyzed in a way useful for determining appropriate postmining land use. More comprehensive, higher quality land use information, with information about regional factors such as transportation, utilities, labor market, etc., should be included in the permit application to identify sites with strong development potential. This, combined with a broader interpretation of the law recognizing the validity of a phased implementation of postmined land use, would continue environmental protection while preparing reclaimed land to meet potential future land use needs. The mining plan can be designed so that appropriate areas are prepared and laid out for future buildings or roads, yet are conducive to interim use for pasture, wildlife or recreation. Reclamation to the interim use, sufficient to protect the public and allow bond release, maintains the potential for later development. Land later can be made available in response to development demands, contributing to a more diversified economy.

Additional Key Words: Surface Coal Mining, Regional Planning, Mining Regulation

Introduction

The Surface Mining Control and Reclamation Act of 1977 (SMCRA) was enacted both to protect the public from the negative impacts of coal mining and to assure the continuation of an industry seen as vital to the nation's interest. The law and regulations as finally promulgated reflect a contentious history, and are ambiguous, confusing, and sometimes unrealistic. Overall the effect of the law has been positive; it is responsible for improvements in reclamation and a significant reduction in the most notorious negative impacts of mining including acid mine drainage, erosion, landslides, and damage to both surface and groundwater. It is a big step, however, between mitigating these negative impacts and returning the land to its "highest and best use."

In Appalachian Kentucky, past farming, logging, and mining practices have left scars on the land. (Caudill 1963) The land's productive capacity is diminished throughout much of the region. The economy has followed the boom and

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bust cycles of extractive industries, and other industries are needed badly. The rugged topography has few relatively level areas which are not in the floodplain, and these tend to be just a few acres in size. (Nieman, and others 1989) Land suitable for commercial, industrial or even residential development is in short supply, a major obstacle to encouraging new industries or businesses in Appalachia. Scarcity is evident by the price that developable acreage commands.[4] Kentucky's strategy for economic development includes better utilization of reclaimed surface mined land to meet the need for commercial or industrial sites. (Kentucky Cabinet for Economic Development 1994) However, current interpretation and implementation of the mining law and regulations appear to be constraining designation of commercial and industrial postmining uses. Lack of clarity in the land use regulations is exacerbated by the lack of planning support or guidance from local or state agencies. The current interpretation of the regulations requires establishing a commercial/ industrial postmining use within a tightly specific time frame. Financing, politics, and other complexities of commercial/industrial development, combined with the logistics involved in mining, especially with large permit areas, make it infeasible for a mining company to commit to such timing. A mechanism for considering reclamation and land capability over a longer term, beyond bond release, and reclaiming to provide an immediate use with a potential for a later, more intensive use, can bridge this feasibility gap.

In searching for this mechanism within a reinterpretation of the law, the questions to be asked are 1) is it the intent of SMCRA to support reclamation for industrial, residential and commercial uses; 2) do the regulations promulgated to implement SMCRA support these postmining land uses, in theory and in practice; and 3) what are the obstacles to more productive postmining land use and how can they be overcome? The implementation of the law occurs largely through the permitting and bonding processes, which set the enforcement framework. A brief analysis of law and regulations introduces a more detailed discussion of the treatment of land use issues in the permit application. An analysis of the State of Kentucky permit application process, including examples from permit files, illustrates how a lack of understanding of land use principles has made the permitting process less effective than it could be in supporting a variety of productive land uses. The concept of phased postmining land use is examined as a solution to aspects of the permitting process which do not fulfill the intent of the law, and in fact work against productive postmining land use.

The Surface Mining and Reclamation Act of 1977

Background

The passage of SMCRA in 1977 created the Office of Surface Mining Reclamation and Enforcement (OSMRE) to promulgate and enforce regulations to control surface mining and reclamation operations. The law requires surface mining operations to follow a complicated process of permit application, bonding to cover expected reclamation costs, ongoing inspection, compliance with performance standards, and an extended period of liability during which reclamation efforts must be proved successful before bond monies are released. The complicated process, its focus on lengthy and detailed engineering specifications, and the lack of clarity with which land use issues are addressed, have restricted the perception of what reclamation can accomplish and inhibited consideration or adoption of reclamation approaches which might better fulfill the intent of the law. (Desai 1993) (figure 1)

[4]Whayne Supply paid approximately $600,000 for 5 acres in Hazard in 1991.

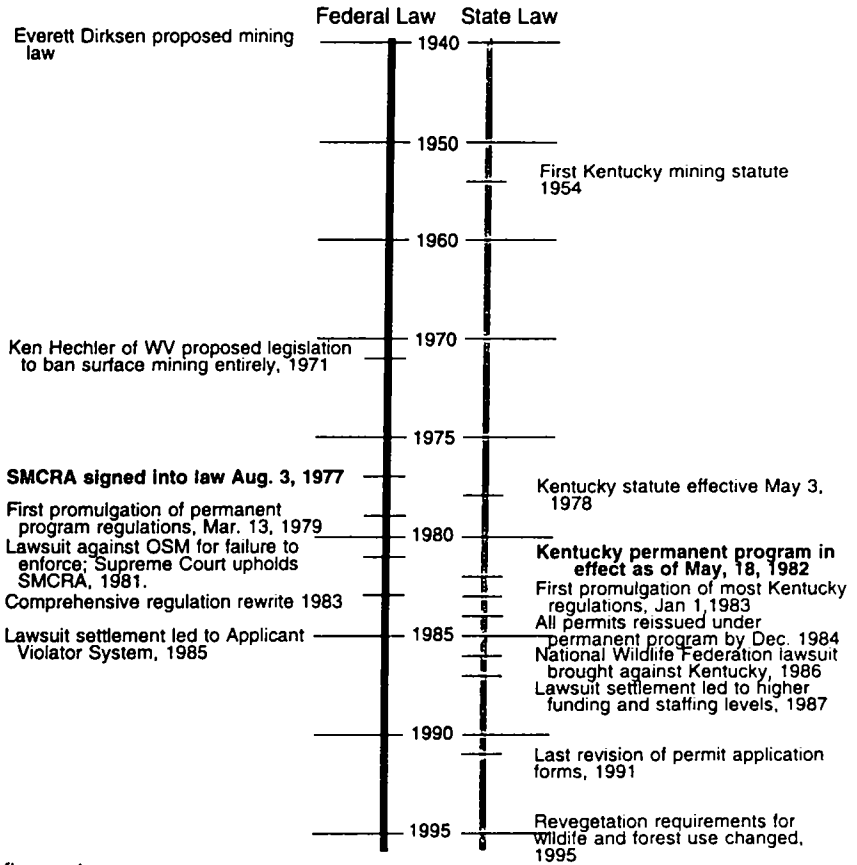


figure 1
Mining Law Timeline

The Act provides the legislative authority and intent upon which the regulations are based. Regulations are subject to challenge and judicial review with regard to whether they actually carry out the intent of a law. (Beck 1993) Within federal law alone there is ambiguity and inherent conflict of purpose. (McElfish and Beier 1990) The intent is laid out in Section 102, Statement of Purpose, to "assure that the coal supply essential to the Nation's ...economic and social well-being is provided and strike a balance between protection of the environment and agricultural productivity and the Nation's need for coal as an essential source of energy...." The relative success with which the regulations have been developed is a function of the degree to which ambiguities in the act are a result of the "political impossibility of developing a consensus" or a lack of understanding of the issues involved. (Miller 1993) The law was enacted on a nationwide basis to set a minimum level of public protection and prevent states from competing with one another on the basis of lesser environmental safeguards. (PL 95-87 §101(g); Scicchitano,and others 1993) The statute does allow states to establish 'primacy' and respond appropriately to local conditions in the design of their regulations. The often tenuous relationship between state and federal agencies has contributed to confusion in implementation of the mining law (Conrad 1993; Miller 1993), at times making compliance difficult for operators who may get conflicting direction from regulators. Similarly, enforcement has been delayed or avoided in some cases. While discretion is built into the law and regulations[5], the exercise of that

[5] For example: 405 KAR 16:020 Section 2. The approved backfilling and grading plan may specify time and distance criteria less restrictive than those set forth in this regulation when the permittee has demonstrated..., and *the cabinet has determined* that use of such criteria will not likely cause adverse environmental impacts. (emphasis added)

discretion is accompanied by uncertainty, political pressure, and litigation. (Morris 1993, Scheberle 1993)

The Intent of SMCRA in Regard to Postmining Land Use

In the Statement of Findings, SMCRA addresses the negative impact pre-law and irresponsible mining has had on the natural environment, public safety, and public welfare through

> disturbances of surface areas that burden and adversely affect commerce and the public welfare by destroying or diminishing the utility of land for commercial, industrial, residential, recreational, agricultural, and forestry purposes, by causing erosion and landslides, by contributing to floods, by polluting the water, by destroying fish and wildlife habitats, by impairing natural beauty, by damaging the property of citizens, by creating hazards dangerous to life and property, by degrading the quality of life in local communities, and by counteracting governmental programs and efforts to conserve soil, water, and other natural resources.(PL 95-87 Title I:§101(c))

The Act thus recognizes a broad standard of utility for land, and also requires that a permit application describe

> the use which is proposed to be made of the land following reclamation, including a discussion of the utility and capacity of the reclaimed land to support a variety of alternative uses, and the relationship of such use to existing land use policies and plans... (PL 95-87 §508(a)(3))

While naively assuming that land use policies and plans exist, this passage illustrates an intent for proposed uses to be examined in a context of surrounding land uses, needs, and markets. The environmental protection performance standards stipulate that land must be returned "to a condition capable of supporting the uses which it was capable of supporting prior to any mining, or higher or better uses of which there is reasonable likelihood....[not] deemed to be impractical or unreasonable, inconsistent with applicable land use policies and plans, [nor involve] unreasonable delay in implementation.. "(PL 95-87 §515(b)(2)) The ambiguity of many undefined terms, e.g., "reasonable likelihood" or "impractical," by default leaves interpretation of the standards to the regulations by which the law is implemented, and to the regulators in the field who are ultimately responsible.

The phrase "higher or better use," however, is from real estate law, referring to the most intensive land use allowed or available, generally one which offers a higher return on investment. Therefore, while public safety is a main focus of the law, there is also a clear commitment to economic productivity, not only by facilitating coal extraction, but also through protecting and restoring the productive capacity and usefulness of the land resource. Confirming this as central to the law is the variance on returning the land to approximate original contour (AOC), allowed for steep slope and mountaintop removal mining. The House Interior and Insular Affairs Committee, which initially considered SMCRA in 1977, determined productive postmining land use to be an adequate reason for an exception to the performance standards of the law.

> The bill is built upon the Committee's finding that in the vast majority of cases, certain reclamation goals must be achieved if the term "reclamation" is to have any real meaning. Nevertheless, the committee has approved exceptions to these requirements to achieve flexibility and avoid arbitrary constraints. For example, the elimination of highwalls, return of the land to approximate original contour, and establishment of viable vegetative cover are among the standards critical to the elimination of the worst effects of coal surface mining and yet these standards are

> either subject to exception, framed in variable terms or both. Rather than weakening the effectiveness of these standards, such treatment is viewed by the Committee as justified and desirable. Workable Federal Requirements must be appropriate to the mining setting and such standards should not preclude practices which are beneficial from a planning viewpoint. (1977 U.S. Code Cong. and Adm. News, p 621-622, quoted in correspondence,permit file #098-0136)

The variance on returning the land to Approximate Original Contour (AOC) applies to steep slope and mountaintop removal mining, in "cases where an industrial, commercial, agricultural, residential or public facility (including recreational facilities) use is proposed...where after consultation with the appropriate land use planning agencies, if any, the proposed [use] is deemed to constitute an equal or better economic or public use of the affected land, as compared with premining use."(PL 95-87 §515(c)(3)) Congress had the vision that with proper planning, reclamation could create economic development possibilities, but recognized that some assurance of support for such development was necessary.

> Surface mining also presents possible land planning benefits as such mining involves the opportunity to reshape the land surface to a form and condition more suitable to man's use. In such instances, the overburden and spoil become a resource to achieve desired configurations rather than a waste material to be disposed of or handled by the most economic means. The performance standards recognize that return to approximate pre-mining conditions may not always be the most desirable goal of reclamation and thus appropriate exceptions to the general requirements are provided. As the realization of such alternative post-mining land uses as industrial, commercial or residential development will often depend on the commitments or assurances that necessary services will be available, evidence of such availability prior to mining is a necessary part of the permit approval process. (H.R. Rep. No.95-218, 95th Cong., 1st Sess., 94 (April 22, 1977), quoted in correspondence, permit file #098-0136)

There is a process written into the performance standards by which "equal or better" use is to be approved. Specific plans for the land use are to be presented, and the permittee must demonstrate its compatibility with adjacent uses, a need or market for the use, and the financial capability to complete the proposed project. Also required is the assurance of "investment in necessary public facilities," and a schedule "integrat[ing] the mining operation and reclamation with the postmining land use."(PL 95-87 §515(c)(3)(B)(iii),(vi)) The exact criteria by which to demonstrate compatibility, financial capability, or market demand are unspecified, however, and are left to the discretion of the regulators. Given the lack of understanding of land use planning, regulators have tended to steer away from built uses. In any case these criteria are generally absent in evaluating potential land use (McElfish and Beier 1990), and the default measure of a reclamation plan is public health and safety coupled with erosion control.

The Influence of the Bonding Process

Reclamation choices have been shaped in large part through the implementation of the bonding mechanism written into law with SMCRA. A percentage of the bond is released upon completion of each of three phases of reclamation but at all times the balance must be sufficient to cover any work remaining. Phase III completion entails the continued success of site vegetation through the liability period in which no appreciable fertilization, mulching or replanting is necessary. The land must also be "capable of supporting the approved postmining land use." In practice, this requires meeting performance standards for the various designated uses, e.g., pasture, prime farmland,

fish and wildlife, or commercial. In Kentucky, Technical Reclamation Memoranda (TRMs) supplement the regulations, guiding coal operators through the standards. Until recently, in steep sloped Eastern Kentucky the major post mining land use designated was pastureland or hayland. Most of the premining land use was either unmanaged forest, primarily third growth, or undeveloped, i.e., derelict land from pre-law mining. There is no prime farmland in the region which must be restored, and any "alternative" postmining land use has been considered a "higher use." Coal operators therefore have chosen to implement the land use with standards they could attain most cost effectively and reliably, usually pasture. Reclamation research on the Illinois basin region suggests that

> reclamation has been guided by two objectives: minimizing economic cost of reclamation, and mitigating physical and aesthetic effects of mining. The result has been widespread planting of simple, homogeneous grassland communities that seem to have limited agricultural and conservation value and may be ecologically unstable. (Brothers 1990)

Kentucky, recognizing that such so-called pasture is not truly a higher use, recently has made regulatory changes.(KDFWR, and others 1995)

> In the mountainous eastern region, the recent trend is toward either fish and wildlife land use or unmanaged forest. Recent regulation changes lowering the stocking rate of trees and shrubs for these uses have prompted permittees to abandon the previously predominant hayland/pastureland postmining land use. Approximately 50% of the areas being mined are returning to forestland or fish and wildlife land uses. Approximately 30% are still being utilized for hayland/pastureland uses, mainly in the areas of more level terrain on the fringes of the eastern coal field. (Smith)

While regulators influenced postmining land use by trading a lower stem count for greater species diversity, making reclaimed areas healthier and more attractive to wildlife while reducing the cost to operators, they have yet to address performance standards and bond release requirements for residential, commercial or industrial use in terms of making reclamation to these uses more attractive or feasible for mining companies.

The Regulations -- Federal and State

Many sections of the federal regulations refer to land use, particularly those outlining the permit application and the environmental performance standards. 30 CFR Section 508 details all the information required on the surface mining permit application. This information, in theory, allows the regulatory authority to determine the probable environmental impacts of a particular operation and the adequacy of the reclamation plan. The performance standards (30 CFR Chapter VII, K, §816) state technical requirements for erosion control, backfilling and grading, disposal of excess spoil, revegetation, and postmining land use. These basic standards primarily consist of engineering specifications deemed sufficient to prevent the major negative impacts of mining. There are additional standards for special categories of mining, including steep slope and mountaintop removal mining in which variances may be granted in lieu of returning the land to AOC. The information required in the permit application is closely related to the performance standards, especially since the application requires submittal of the reclamation plan. This plan, which includes postmining land use, must be able to produce results which fulfill the performance standards.

Performance standards for postmining land use capability require that, before final bond release, "affected areas shall be restored in a timely manner," to conditions capable of supporting prior uses or approved alternative uses. "Higher or better alternative uses may be approved if :

(1) There is a reasonable likelihood that the land use will be achieved

(2) The use will not be

impractical or unreasonable

(3) The landowner or land management agency having jurisdiction over the lands has been consulted, and the proposed alternative postmining land use is consistent with applicable land use policies and plans;

(4) The proposed use will not present an actual or probable hazard to public health or safety or threat of water pollution or diminution of water availability

(5) The proposed use will not involve unreasonable delays in implementation

(6) the proposed use will not cause or contribute to violation of federal, state, or local law."(405 KAR 16:210)

These and other land use requirements are vaguely stated, and do not give regulators much guidance. One intent behind items 1, 2, and 5 is to ensure that reclamation does, in fact, take place, acknowledging that the sooner this happens, the less negative environmental impact will occur. Items 2, 3 and 4 are to protect the landowner and the community. Item 4 is the only one which explicitly deals with the postmining land use capability, the other standards relate to financial feasibility and legality. However, they do not define how to evaluate practical or reasonable, and the meaning in the field has come about by trial and error, negotiation, compromise, and political pressure.

Performance standards for revegetation (405 KAR 16:200) include coverage standards for grasses and legumes, numbers of plants (stem counts) for trees and shrubs, and diversity measures. Pasture and cropland uses must meet specific productivity, or yield, standards based on local conditions. Performance standards for commercial or industrial postmining uses are not clear or specific, however, and this has implications for bond release. As an illustration, ten years after passage of SMCRA, this 1987 memo was sent from the Kentucky Natural Resources and Environmental Protection Cabinet to the Lexington OSM office:

Dear Mr. Tipton,

Attached is a letter from J.R. Harris granting OSMRE approval of an interim program experimental practice permit for commercial development as the primary post mining land use.

As the approved permit contains no information relating to criteria for establishing the commercial development, it is unclear when the permit becomes eligible for a complete bond release.

Since approval of the complete release will require involvement of the OSMRE, I would appreciate your review of the permit and receipt of your understanding of the work to be completed by the permittee to receive the release. (Permit #098-0067)

Potential retention of bond monies, if regulators determine that a postmining land use is not, in fact, established, is a powerful disincentive for coal companies to attempt alternative land uses. Timing is a key element in this determination. The permit application requires a "discussion of how the proposed postmining land use(s) will be achieved within a reasonable time frame." (MPA-03, 21.12(c)) The concern with achieving the land use derives partly from the goal of minimizing erosion by reclaiming as quickly as possible, and partly in response to irresponsible operators. Many had defaulted on their reclamation plans, or, in order to avoid backfilling and returning to AOC, designated commercial uses for benches without ever building anything there (Rothman) Therefore, in practice, the regulatory requirements of "reasonable likelihood" for achievement and no "unreasonable delays"(405 KAR 16:210 §4(1),(5)) have been taken to mean a tangible commitment to development. At the permit application stage this might be proof of a bank's commitment to financing, in addition to the specific plans for development required by the regulations. For final bond release the regulatory

agency wants to see the initial stages of construction. Unfortunately, the realities of the construction and development process require the flexibility to respond to market fluctuations. The likelihood of the timing of reclamation coinciding with an advantageous development window is not high. Coal companies are resistant to the possibility of 1) delaying final bond release and 2) having to hold onto the land, and the liability which that entails, until development becomes practical. As one coal executive noted, they [coal operators] do not want to be in the land business, and are certainly not land use planners. (Geiger) These factors inhibit designation of the more intensive postmining land uses in original permit applications.

Designation of postmining land use can be changed through a major permit revision process, which requires public notice. This is the way most industrial and commercial land use designations have been handled. Operators will weigh the time and cost for the revision against any possible cost savings involved in changing the use. If it involves less reclamation cost and quicker bond release, the company goes through with the revision. As an example, landowners are "quick to take advantage of any level areas adjacent to roads for residential use or commercial/industrial if offices or shop buildings are left in place by the coal company. Final bond releases are relatively easy to obtain if the land is being totally utilized by home construction or occupation of existing buildings by a new business."(Smith) One problem with this approach is that it precludes any long range planning for development, and makes it difficult to coordinate reclamation and development of any particular site with regional trends so as to make infrastructure investment more efficient. A paradox is created, as well. Surface owners of land do have the right to make use of their land, and one often finds that they have initiated projects before bond release, which can be problematic for operators. Even when the surface owner is a subsidiary of the coal company, since proof of the use is required to get approval and bond release, one finds construction beginning on alternative postmining land uses before they are actually approved. This acts to subvert the intent of the public notice provision of the law. In one case in Eastern Kentucky the land holding corporation related to the mining company deeded the surface to the city, which sold it to a private company to build a prison. The construction of the prison created a highwall and expanded a hollow fill, all of it before final bond release and without supervision from the regulatory authority. After the facility was in operation, the mining company filed a major revision requesting approval of a land use change from forest to industrial/commercial. The "walk sheet" inspection for proposed major revision #2, stated, "Company is requesting to change Post Mining Land Use to Industrial Commercial so that a Phase III bond release can be obtained. Otter Creek Correctional Facility has been constructed on the permit."(permit #836-0120)

The Permit Application

Surface coal mining and reclamation regulations are implemented through the permit application and approval process, as well as through enforcement. (figure 2) Kentucky permit application requirements evolved quite a bit through the period between the pre-SMCRA state program and primacy, although the land use regulations remained vague. Submittals improved as engineering firms and regulators grew more familiar with the scope of work required to get permit approval. The areas in the law and regulations where ambiguities existed became more obvious, and in many instances conflicts arose which had to be settled in court. A default, working, interpretation of the regulations developed, as certain disputes were resolved verbally with no written record in the file of the rationale behind final permit conditions, perpetuating misunderstanding of the intent and purpose of the law in regard to land use issues.

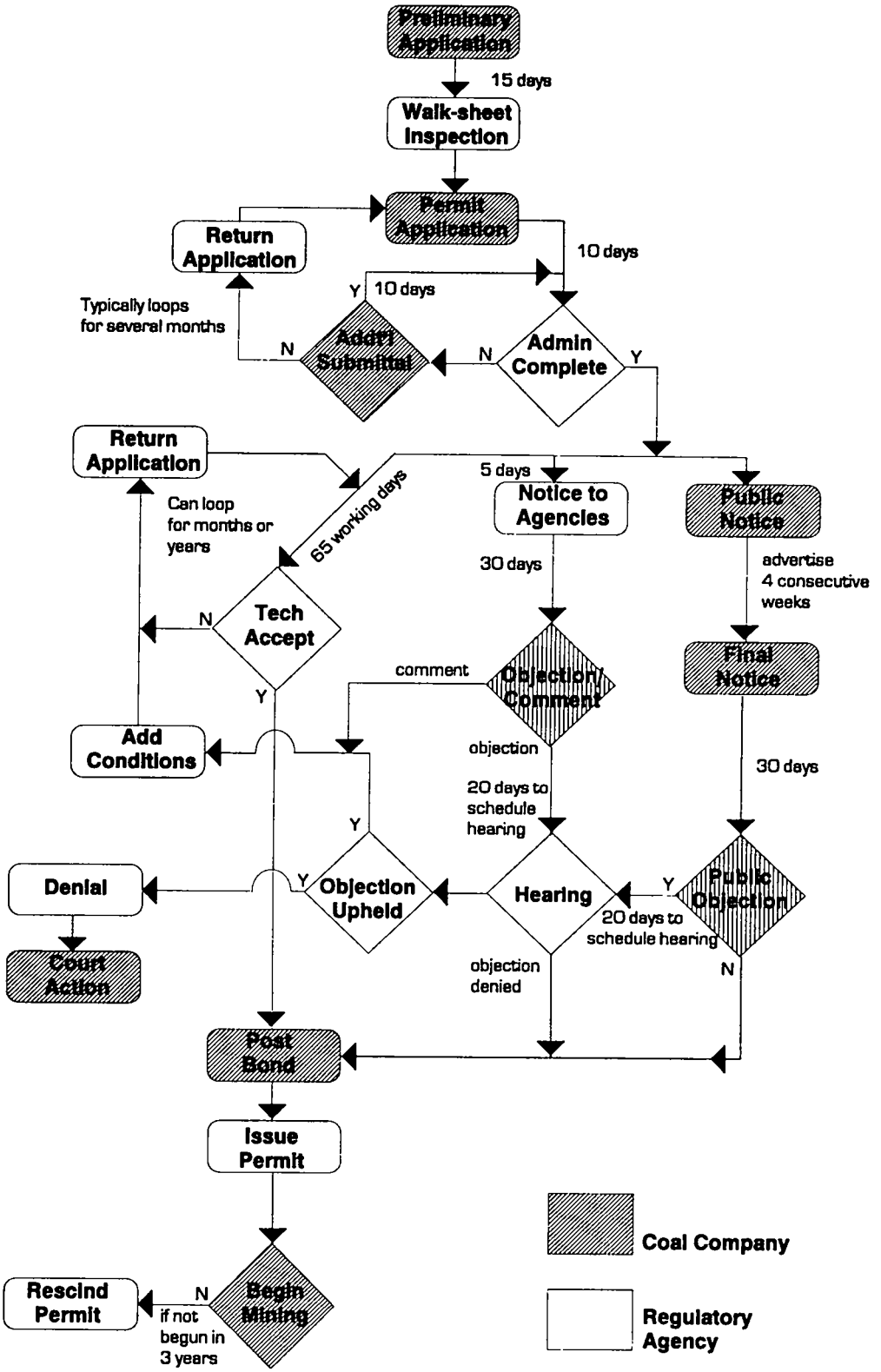


figure 2
Generalized Coal Mining Permit Process

The quality of information submitted with permit applications still varies greatly, depending on the experience and expertise of the applicant and the reviewer. (McElfish and Beier 1990) Based on the information in the application, permits for mining and reclamation are issued with various conditions which the mining operators must follow.(figure 2) In many sections of the application, submittals consist of restating the performance standards. In other words, omitting site specific details, coal operators are in effect complying by stating they will comply. For certain precise technical specifications such as slope, or cross-drain intervals, this provides sufficient and measurable grounds for evaluation and enforcement. But because the land use criteria are vague to begin with, restating them does not give regulators an adequate means of assessing performance.

The major deficiencies in permit applications, associated with land use issues, are poor quality maps, inconsistent responses to the land use questions, an agronomic bias to land use information, and confusion in regard to the difference between reclamation and land use. Item 21.12 of the application requires a discussion of the "feasibility, i.e. suitability, capability, cost effectiveness of the proposed postmining land use(s)...[and] how the proposed postmining land use(s) will be achieved within a reasonable time frame" if the postmining land use is to be different than the existing or pre-mining use. The permit actually requires less explanation than called for in the regulations, dropping discussion of the "utility" of the reclaimed land or capacity to support a "variety of alternative uses." There appears to be no attempt to discern the best use of the land or to place it in context within the local region. A typical submittal to 21.12 follows:

Attachment 21.12 (A,B,C & D)

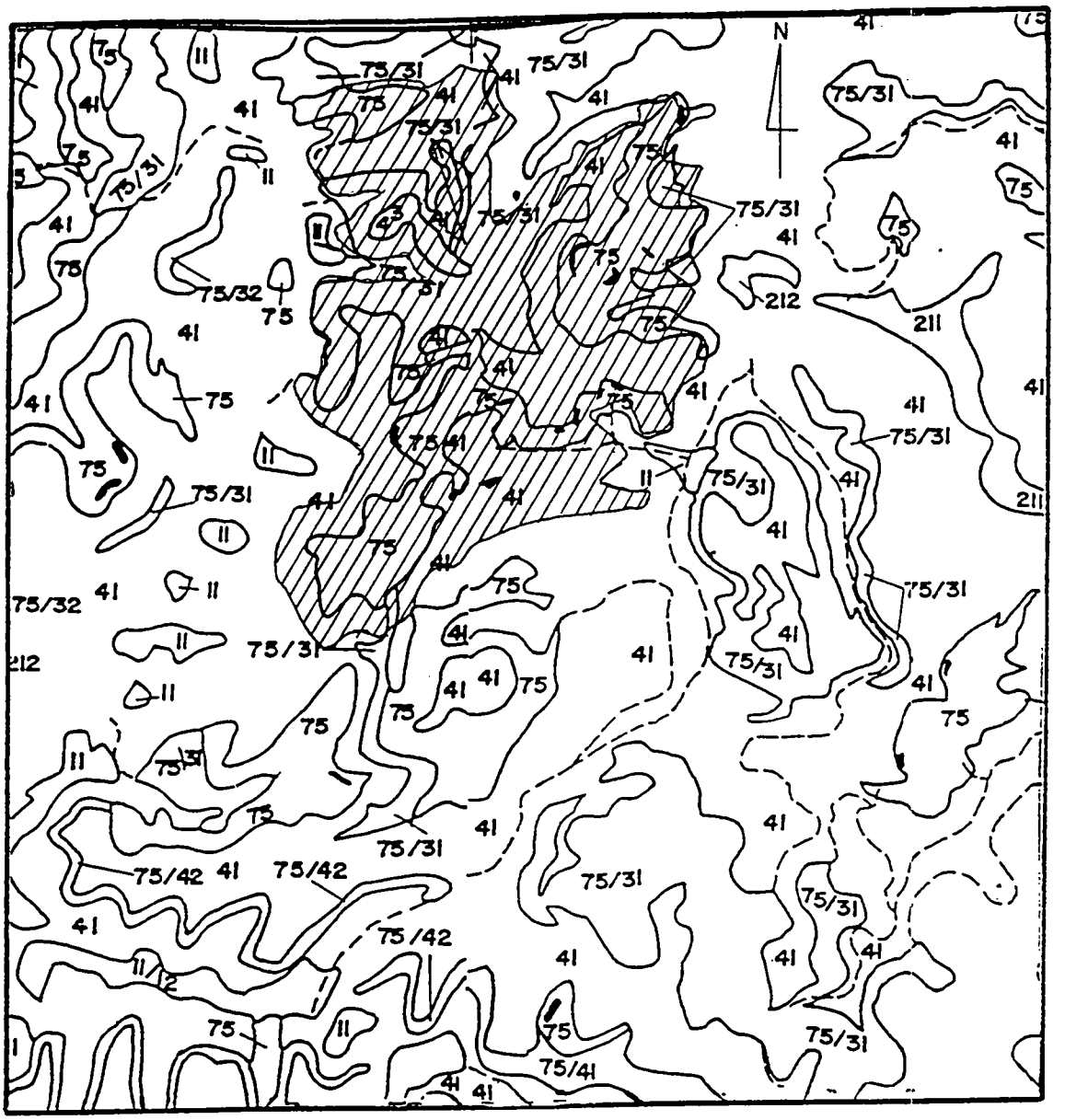
A) The proposed post-mining land use is compatible with adjacent land uses in this part of County. A land use change for fish and wildlife habitat with permanent roads has been demonstrated to be feasible in this part of Eastern Kentucky. This land use has been demonstrated to be expediately [sic] achieved. This is due to planting of quick cover crops such as annual rye and winter wheat.

B) Upon completion of the mining activity, the revegetation plan will immediately be put into place. The post-mining land use will be obtained when the vegetation plan for the area has been successful.

C) The post-mining land use will be achieved as quickly as natural conditions will allow. This will be done by the immediate execution of the revegetation plan. Hand-seeding will be employed if areas of poor vegetation are found. (permit #867-0355)

This submittal does not specify why the land use is compatible with surrounding uses or how it will be feasible. The entry also confuses achievement of revegetation with achievement of the land use.

Engineers, agronomists, and biologists are some of the specialists involved in the permitting process, and their professional bias is evident. Land use analysis is derived primarily from soil survey data and vegetation associations, e.g., upland forest. Granted, much mining occurs in remote areas where the human influence is small, but roads and infrastructure are important facets of land use and must be addressed. In addition, there is little consideration of future land use potential or need. The map in figure 3, which does not show highway 15 at all, as well as omitting parts of the legend, e.g. land use 11 (residential), is typical of how whole categories of land use are ignored.



RIVER COAL CO., INC.
EXISTING LAND USE MAP

LEGEND

31 HERBACEOUS RANGE

41 DECIDUOUS FOREST

43 MIXED FOREST

75 EXTRACTION: ACTIVE OR INACTIVE MINES, QUARRIES OR PITS

APPLICATION NO. 497-0073: AMEND. 2

PERMIT AREA

SCALE: 1" = 2000'

ATTACHMENT NO. 20.7.B

figure 3
Typical Land Use Map (reduced)

Other examples from this and a more recent permit file illustrate that land use and planning are misunderstood and land use issues are inadequately analyzed in permit applications. The first file involves a large site whose permits have continued through the interim, transition and permanent program years (permits #097-0073 and #497-0073). The initial job was a mountaintop removal, and paperwork in 1981 listed both premining and postmining land use as forest. Much of the site had been disturbed in the previous 20 years; the transition application more accurately listed pre-mining land use as "undeveloped." The transition was begun in July 1982, and a permit issued in September of 1984. Much of the original acreage was fully mined by this point, though bond was not released, and Item 31.3 of the application stated

> The proposed post mining land use for this area will be mostly Commercial. A small part of the area which is not directly related to the commercial use will be developed as pasture land. The commercial use intended will be for air traffic. Already, a 3,200 ft. runway has been constructed and used privately. When the mining process allows another strip in excess of 6,000 ft. is to be constructed which will be able to accomadate [sic] larger aircraft. The 3,200 ft. runway already in use is paved and a terminal building is at present (10-13-83) under construction.

No major revision for land use change occurred -- the change was handled *during and after construction* through the transition program permit application.

Amendment 1 to the transition permit proposed to finish a point removal and leave a surface configuration to match the adjacent area on which the airport had been built, stating "The commercial land use option is proposed to allow expansion of the small airport that is presently operating on the interim portion of the permit" (Attachment 20.9.A). Yet the application fails to mention the airport as an existing land use in Attachment 20.7.A, nor shows it on the Environmental Resources Map or Existing Land Use Map.

Entries to Section 20.12 show other weaknesses in analysis of land use issues.

> Attachment 20.12(a):Feasibility of Post-Mine Land Use
> The post-mining land use of hayland/pasture is a feasible alternative to the original usage of forest. The pre-mine forest land occurring on undisturbed and previously mined areas was basically unmanaged. The diversification of habitats through differing land uses are also desirable to wildlife. An "edge" effect will be created, thereby improving wildlife distribution....
> The cost effectiveness of planting pasture as opposed to forest makes it a practical land use alternative. The cost is already reduced by not having to plant developed seedlings into an already established herbaceous cover....
> The commercial land use is proposed to allow expansions of the ...Airport operating on the interim portion of the permit area. This airport is a public facility that benefits the local economy in income and needed air transportation.
> In summary, the post-mine usage of hayland/pasture and commercial instead of forest is a viable alternative. The selected land use will provide erosion control, diversify habitats, be a less expensive alternative to forest, and serve the good of the community.

This entry describes encouraging wildlife. Not only is this not compatible with the adjacent land use of airport (one does not want deer wandering the runways), but also, habitat distribution must be considered within a larger region than investigated in this application. (Nieman and Merkin 1995) Regardless, the proposed interim use actually was for pasture, not

wildlife, and there is no mention of cattle nor a market demand for hay. As for cost effectiveness, this describes the cost effectiveness of the reclamation, not the land use, a very common mistake. Lastly, the value to the community of an expanded airport is only hinted at through the acknowledgment of the value of the existing airport, yet this should be a very strong argument for allowing the land use change.

This permit indicates that the timing of development needs to be addressed in a more flexible manner. Permit file records indicated that in this case, the issue of timing, or implementation, of the land use was debated at length, but finally settled informally. The permittee in essence suggested a phased land use, first of pasture, then commercial use when the airport was actually expanded. Item 20.8 lists the acreage twice, once for each proposed land use, pasture and commercial. A review letter stated, "List the acreage to be used as pasture and acreage for commercial. You cannot propose optional land use change." The permittee replied, "The hayland/ pasture use is requested to be used contemporaneously with the commercial usage. As stated in the application, the rate of expansion of the airport is unknown; therefore, a land use is specified for the interim. Since this is an approved method in the original permit, the same uses can apply to the amendment." The permit authority responded, "An optional landuse can not be proposed," however the application entries stand, but on the permit face condition 10 states, "Approval is granted for the alternate postmining land use of hayland/pastureland as described in the permit application." There existed sufficient proof that the airport would expand, and the land use was productive, feasible and would be established, but it was not presented properly. Neither the permittee nor regulators understood how to document or prove support for this land use, and much time was wasted before the issue was finally let drop in the permit approval process.

In a related case of regulatory dislike for a contingent land use, a much more recent job has a strong potential for development of commercial or residential uses. It is close to a developed area, has utility infrastructure available, has a major highway nearby, and is a large enough site to warrant development. (permit #867-0355) Because the timing both of the actual mining and certain improvements in the area is subject to change, the development time frame is uncertain. In this case, the permittee initially attempted to declare an industrial postmining land use, but the only evidence of this is in correspondence related to the permit review. A deficiency letter stated, "... Item 21.10: What does <u>and Industrial</u> mean? Please remove this." The reply notes, "Industrial has been deleted as a description of the post-mining land use." A fish and wildlife postmining land use has been declared instead, though a careful look at the reclamation plan makes it clear that the site is being handled in such a way as to suit the more intensive use as well. The backfill on the ridge area is being placed only twenty feet deep, as opposed to a potential depth of over a hundred feet. Therefore foundation engineering on this site will be fairly straightforward, conventional in both design and cost. This site will be none the less useful for a fish and wildlife use, but prepared for an eventual residential or commercial use. This is forward thinking; it is unfortunate that discussion of the land use potential and designation of future, more intensive land use, must be skirted to remain within the interpretation of the law.

Conclusion

Given the process of development of the law and regulations, and the nature of the types of problems SMCRA was intended to solve, the problems still remaining are not surprising. The technical answers to issues such as acid mine drainage, erosion control, and stable and safe sediment ponds and fills, have come from engineers who are specialists in their particular fields. The personnel who develop permits and those who review them also are engineers, often come from the mining industry, and are habituated to looking at things a

certain way. They are not land use planners, landscape architects, economic geographers, or from some other discipline which considers land use issues within a broad framework. As one permit reviewer noted, if the technical problems weren't taken care of, the land use designation wouldn't matter because the land would not be useful for anything. But solving the technical problems, while necessary, is little assurance of a wise use of the land.

An important step is to recognize that land use occurs in a regional context. The utility of a particular use is related to the need for that use now and in the future, and whether other land is available to fulfill that need. Universally accepted planning principles recognize that land use needs change over time, as does the availability of the factors which support particular land uses. Time constraints for reclamation should be applied, based on their original intent of protecting the public and environment from negative impacts of mining, to quick achievement of an acceptable interim use, such as pasture or wildlife, which is consistent with future, more intensive development. The permit application should answer questions critical to more intensive land uses such as the relative location and quality of roads and other transportation networks, availability and cost of utilities and telecommunications, nearness to population centers, and capacity of the workforce, as well as general costs of construction based on engineering and location factors. There is a broad spectrum of factors which must be examined and used to justify particular land uses. A clear justification for expecting a site to have high potential for future residential or commercial development must be accepted as fulfilling SMCRA requirements for feasibility, capability, compatibility, etc., and allow designation of a contingent use. In addition, consideration of phased land use more closely follows the regulatory requirement to discuss "the utility and capacity of the reclaimed land to support a variety of alternative uses." There is nothing in the law or regulations to prohibit a phased or contingent land use. This has derived from practice, and this interpretation can and should be changed.

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THE PROSPECT OF GULLY GRAVURE ON RECLAIMED HILLSLOPES

by

Terrence J. Toy

and

W.R. Osterkamp

Abstract. Hillslope development and evolution are the products of process-sequences. One such sequence is gully gravure, a mode of development requiring coarse-textured material overlying fine-textured material. Rill or gully processes erode through the coarse material into the fine material. The coarse material falls into and accumulates in the channel entrapping fine material. The permeability and porosity of the channel-fill is reduced, and runoff is deflected toward the periphery of the deposit. Thus, the loci of erosion shifts laterally along the hillsides. The sequence is completed by channel-filling with additional coarse material from the channel sides. A new surface of coarse material is created at a lower elevation as a result of lateral planation. Field studies near Tucson, Arizona and Denver, Colorado, indicate that gully gravure may occur on reclaimed hillslopes. Previous research suggests that gully gravure results in landscape stability, maintenance of adjusted hillslope angles, and low rates of denudation and sediment delivery, which are fundamental objectives of reclamation.

Additional Key Words: Hillslope Erosion, Hillslope Evolution, Rock Fragment Surface Cover

Introduction

Both natural and reclaimed hillslopes are subject to various geomorphic processes, such as geochemical and pedochemical weathering, mass-movement, erosion, and deposition. The assemblage of processes in a particular environmental setting controls hillslope development and evolution. Each process has been examined in detail by earth scientists in order to understand its mechanics and rates of operation. Infrequently, however, do we consider sequences of processes that may be collectively responsible for hillslope characteristics. Through sequencing, one process prepares the surface for the next process and usually contributes certain attributes to the final form. It is the purpose of this report to describe the process-sequence known as " gully gravure" and suggest, based upon field examples, that this process-sequence may be occurring on reclaimed hillslopes.

The Gully Gravure Process-Sequence

Gully gravure is the term used by Bryan (1940) to describe the process-sequence depicted in the idealized diagram of Figure 1. In profile, the initial stage (A) consists of a nearly planar surface of coarse-textured, erosion-resistant geologic material overlying a fine-textured, less-resistant material. During the second stage (B), the surface is incised by rill or gully processes. In stage (C), coarse material falls by simple gravity processes and accumulates in the channels because flow energies are insufficient to transport it downslope. Weathering processes influence the rate of material

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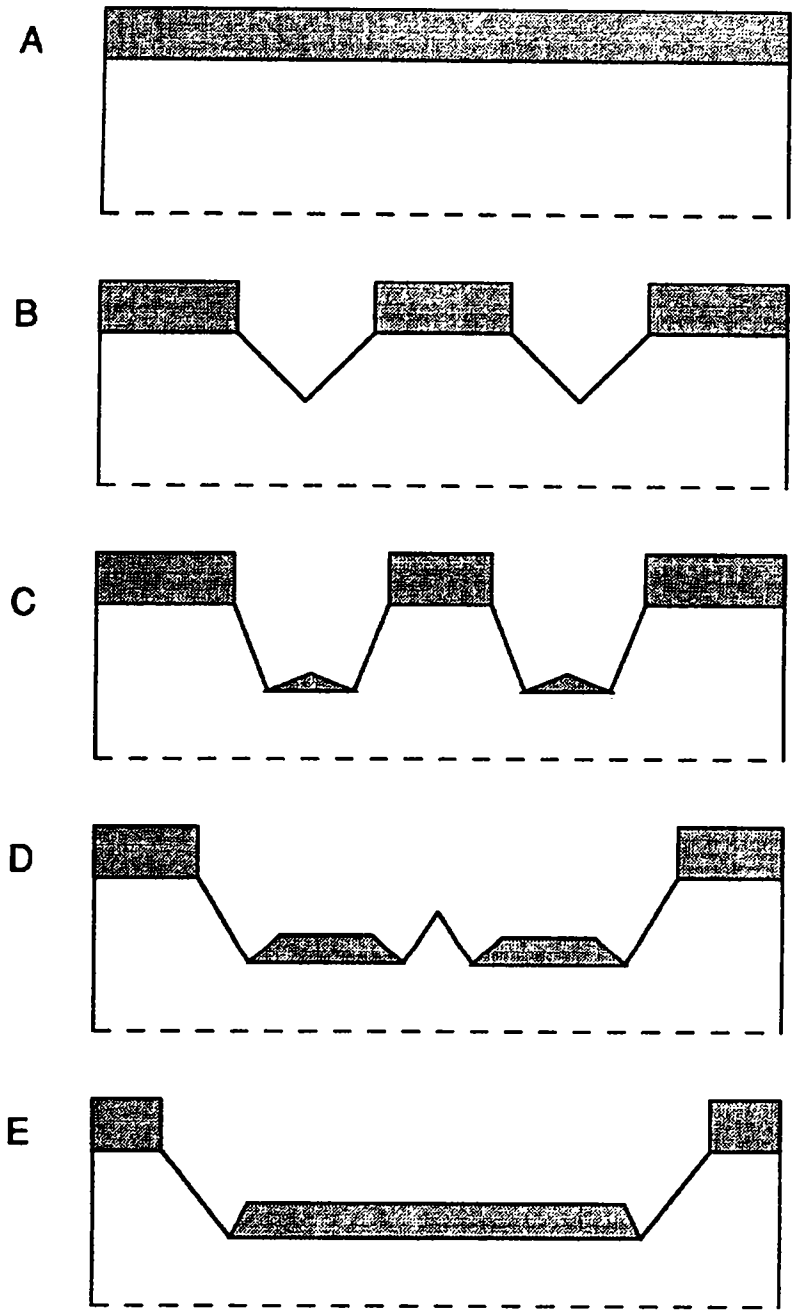

Figure 1: The sequence of gully gravure processes; (A) initial stage of coarse material overlying finer material; (B) incisement of surface by rills and gullys; (C) accumulation of coarse material from above in the incisement; (D) lateral shifting of rills and gullys and continued accumulation of coarse material; (E) lateral planation and formation of a new surface veneered by coarse material.

released from the overlying cap and hence the rate of accumulation in the channels. The coarse channel-fill entraps interstitial fine-textured materials produced by weathering, or wind and water erosion. The permeability and porosity of the channel-fill are reduced, and runoff is deflected toward the periphery of the deposit. As suggested in stage (D), the loci of erosion shift laterally along the hillslope, exploiting the weakness of the fine material. The sequence is completed by the filling of the newly enlarged channel with additional coarse material from the sides of the channel. A new surface of coarse material is created at a lower elevation as a result of lateral planation on the hillside, as shown in stage (E).

A few characteristics of hillslopes experiencing gully gravure are especially noteworthy. First, there seems to be a limit to the depth of incision that can occur before sufficient coarse material accumulates to deflect the flow and erosion toward the periphery of the channel-fill. Second, a topographic inversion takes place between stages (B) and (C), wherein the former channel becomes elevated due to the accumulation of coarse material. Third, in map view, the surface may appear as alternating stripes of coarse and fine materials. Last, surfaces composed of coarse rock fragments, as found in many places, could be products of gully gravure.

Gully gravure has not been widely documented because many hillslopes do not possess the requisite geologic configuration and hillslope processes commonly operate at slow, nearly-imperceptible, rates. Nevertheless, in addition to Bryan (1940), Mills (1981), Twidale and Campbell (1986), possibly Whitney and Harrington (1993), and Osterkamp and Toy (1994) found evidence of gully gravure at various scales and in various environmental settings.

Field Evidence

Osterkamp and Toy (1994) examined hillslope development on two road cuts and one borrow pit southeast of Tucson, Arizona. At these sites sand, gravel, and cobbles cap mid-Pleistocene lacustrine beds. The hillslope surfaces are extensively rilled with coarse material accumulating in the channels. Excavation of cross-sections through the rills revealed sharp textural contrasts between the channel-fill and the lacustrine material adjacent to and beneath the channel. Especially interesting was the wide, roughly horizontal contact along the base of the channel fill, shown in Figure 2. This indicates lateral planation rather than further incision by erosion processes. In some places, topographic inversion appeared to be in progress. The field evidence was analogous to stages (C) and (D) of Figure 1.

Hillslope development at certain locations within the "mesa and butte" topography of the Colorado Piedmont south of Denver, Colorado, also suggests the operation of gully gravure. The hillslopes in this area are products of a complex geomorphic history as portrayed diagrammatically in Figure 3. In profile, the geology consists of horizontal strata of conglomerates or rhyolite overlying sandstones, siltstones, and shales, shown in stage (A) of Figure 3. Erosion by fluvial processes is primarily responsible for the characteristic flat-topped mesas and buttes with steep sideslopes. Colluvial deposits at various places on these sideslopes attest to debris flows following formation of the mesas and buttes, as indicated in stage B (fig. 3). These deposits were incised by fluvial erosion and probably debris avalanches during a high-intensity precipitation event of June 16, 1965, indicated by stage C (fig. 3). Gravitational processes caused both coarse- and fine-textured colluvium to fall into the channels. Fluvial processes since that time have evacuated much of the fine material leaving the very coarse material to accumulate in the channel. Some clasts of this channel-fill approach a mean diameter of 3 meters. This coarse debris deflects flow toward the periphery of the channel resulting in nearly-vertical channel banks in various places. The current cross-section topography is reminiscent of stage C in Figure 1. If this interpretation is correct, then stages D and E of Figure 1 may follow.

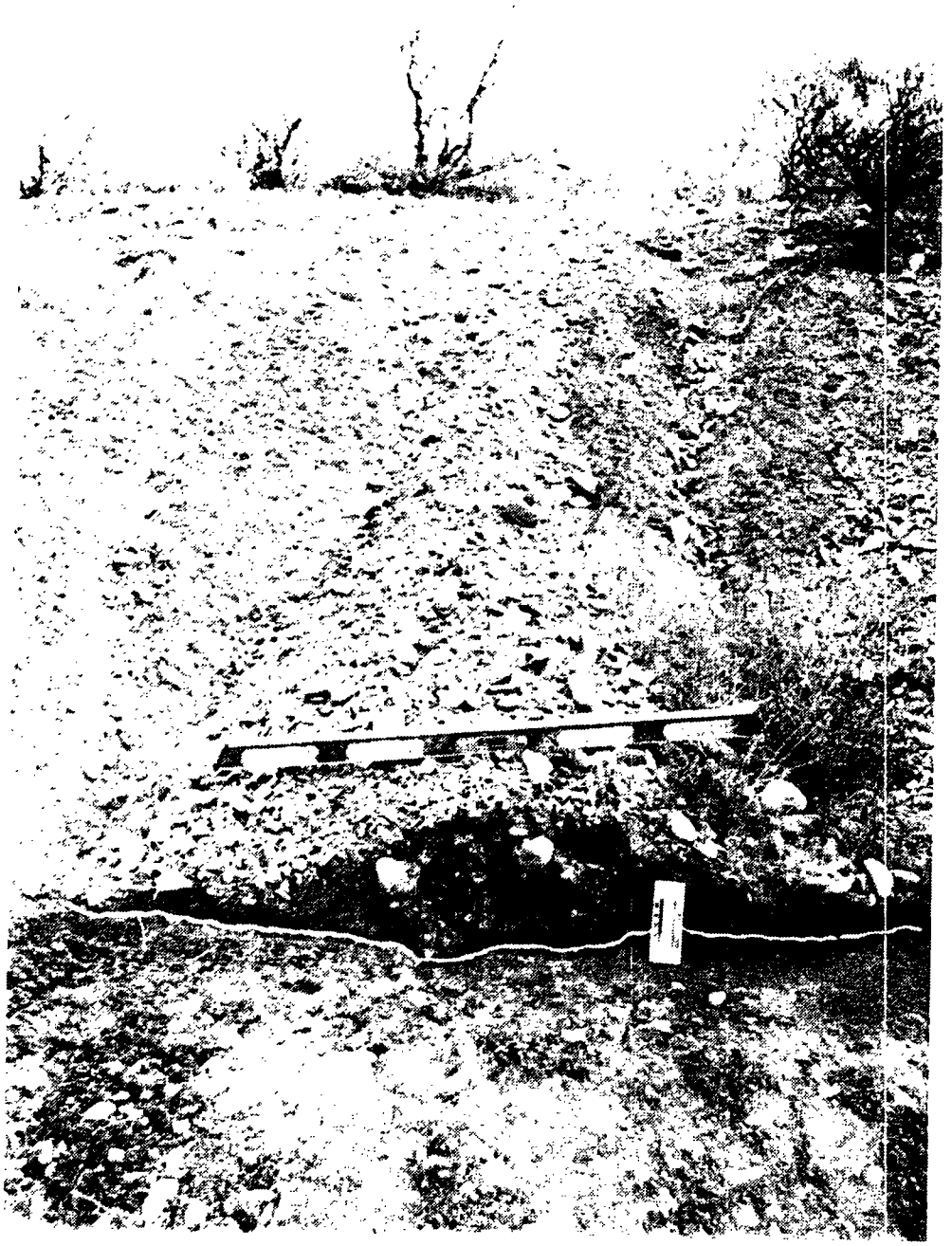

Figure 2: Photograph of rill partially filled with coarse material from above. The white line marks the contact between the coarse material and the underlying fine material. This field site, near Tucson, Arizona, is representative of stage D in Figure 1.

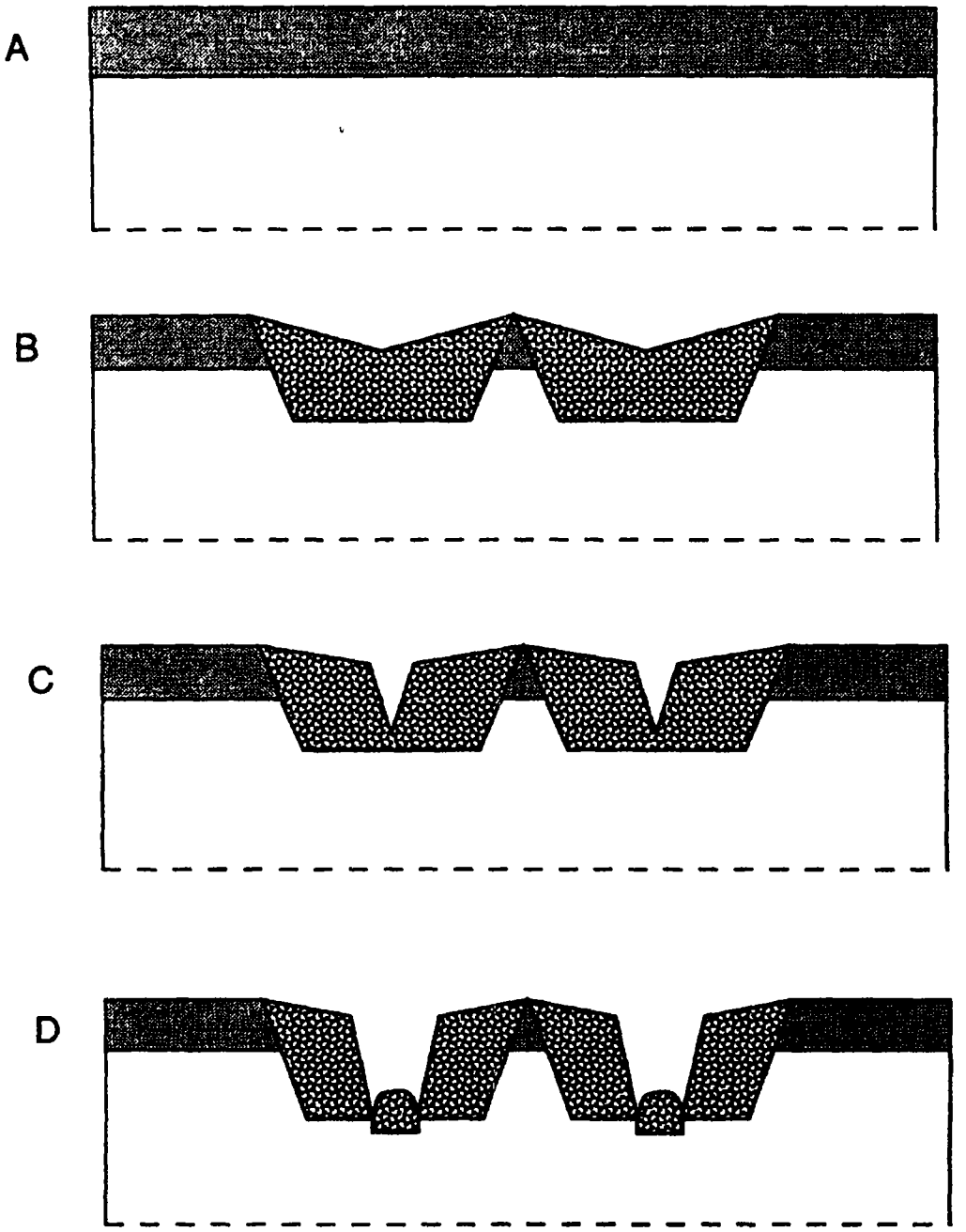

Figure 3: Diagram representing a sequence of hillslope-development processes near Denver, Colorado; (A) horizontal resistant rocks overlying less resistant rocks, forming a mesa-type topography; (B) erosion of sideslopes and filling of gullys with coarse colluvium; (C) fluvial incisement of colluvium; (D) fluvial sorting and additional accumulation of the coarse colluvium, thereby providing the potential for continuing gully-gravure processes.

An implication of this interpretation is that the source of channel-fill in gully gravure need not be a discrete capping layer of coarse materials envisioned by Bryan (1940) and others. A heterogeneous mixture of particle sizes, as occurs in a colluvial deposit or a waste-rock (spoil) disposal site, will suffice. Consequently, gully gravure on reclaimed hillslopes is a distinct possibility.

Conclusion

Osterkamp and Toy (1994) concluded that gully gravure yields general landscape stability, maintenance of adjusted hillslope angles, and low rates of denudation and sediment delivery. These conditions are fundamental objectives of hillslope reclamation. Such surfaces provide stable platforms for revegetation and support eventual land uses.

There is reason to suspect that gully gravure may be occurring on some reclaimed hillslopes. This would be fortuitous because the accumulation of coarse channel-fill limits the depth of incision by fluvial erosion and the development of a coarse surface veneer may reduce soil loss by as much as 99% (Osterkamp and Toy, 1994). Although it would not be prudent to rely upon gully gravure to produce stable hillslopes from waste rock disposal sites because specific geologic conditions and lengthy time periods are required, this process-sequence provides one mechanism that controls the extent of fluvial erosion and drives hillslope development toward stable configurations in many cases.

To be sure, gully gravure is only one mode of hillslope development and evolution. It is, however, a most interesting prospect within the context of disturbed-land reclamation and one deserving of careful scrutiny and verification. Toward this end, we solicit your assistance. If there is evidence of gully gravure on the reclaimed hillslopes with which you are familiar, please bring this to our attention.

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METHODOLOGY FOR BUILDING SOIL BASED VEGETATION PRODUCTIVITY EQUATIONS: A STATISTICAL APPROACH[1]

by:

Jon Bryan Burley[2]

Abstract. Reclamation specialists have been interested in developing predictive equations to assess reclamation efforts in reconstructing soils to support vegetation growth. One predictive effort is associated with a statistical approach examining somewhat large data sets containing plant growth yields and soil variables. While the results from such procedures have been reported for the last seven years, a description of the methodology has not been described since 1987. This paper describes this statistical vegetation productivity model building process. To complete the basic analytic steps in the process, a statistical computing software package is required to conduct principal component analysis (PCA) and multiple regression analysis. The field data required to conduct the analysis are extensive. All crops and woody plants of interest (source of dependent variables) should be grown on all soil profiles (source of independent variables) for a period of approximately 10 years. The time period should include dry years, wet years, and average moisture years and should ideally include reclaimed and undisturbed soils. For any individual investigator, this type of data set would be expensive and time consuming to generate; however, the former United States Soil Conservation Service (SCS) has conducted similar work for a fair number of counties in the United States of America and can provide a substantial portion of the undisturbed soil database for investigators interested in developing a vegetation productivity equation for their region. Different units of measure across crop types are not necessarily an issue in vegetation productivity equation modeling, because each vegetation type is standardized to a mean of zero and a variance of 1. Then the various crop types and woody plants are examined with PCA. This statistical treatment allows the investigator to determine the number of dimensions necessary to explain the variance across all vegetation types. Ideally, if all of the crops of interest covary together, they can be combined into one dimension generating one dependent variable; otherwise an investigator may have to develop an equation for each significant dimension indicated in PCA. Soil factors suitable for regression analysis are calculated by employing a soil profile weighting formula. Before conducting regression analysis, a regression screening procedure may be employed to search for the most promising main effect, squared terms, and two variable interaction terms. The Maximum-R-squared improvement technique has been determined to be the best stepwise selection procedure to search for the best equation. Once a regression equation is selected, it can be further analyzed with bootstrap, subsampling, and jackknife statistical procedures. Finally, developed equations should be evaluated with results from reclaimed soils. These procedures for the basis of the methodology.

Additional Key Words: landscape planning, biometric statistics, soil science, prime farmland reclamation, agroecology

Introduction

This paper describes the fundamental procedures specific to the vegetation productivity model building process. To complete the basic analytic steps in the process, a statistical computing software package is required to conduct principal component analysis (PCA) and multiple regression analysis. In my efforts, I used the Statistical Analysis System (SAS) software for the microcomputer (1985); however, for

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investigators wishing to apply the methodology similar statistical software packages may suffice.

Table 1. Dependent variables and units of measurement as recorded and published by the U.S. Soil Conservation Service (Jacobson 1982).

Abbreviation	Vegetation	Measured Average Yield
Evergreen Trees		
JV	*Juniperus virginiana*	feet/20 years
PG	*Picea glauca densata*	feet/20 years
PP	*Picea pungens*	feet/20 years
PS	*Pinus ponderosa scopulorum*	feet/20 years
CO	*Celtis occidentalis*	feet/20 years
Deciduous Trees		
FP	*Fraxinus pennsylvanica*	feet/20 years
PD	*Populus deltoides*	feet/20 years
ST	*Salix alba tristis*	feet/20 years
UP	*Ulmus pumila*	feet/20 years
Deciduous Shrubs		
CA	*Caragana arborescens*	feet/20 years
CR	*Cornus sericea*	feet/20 years
PA	*Prunus americana*	feet/20 years
PV	*Prunus virginiana*	feet/20 years
SV	*Syringa vulgaris*	feet/20 years
Agronomic Crops		
SW	Spring Wheat	bushels/acre
BA	Barley	bushels/acre
OA	Oats	bushels/acre
SF	Sunflowers	pounds/acre
SB	Sugarbeets	tons/acre
SN	Soybeans	bushels/acre
GE	Grasses/Legumes	tons/acre

1 meter = 3.281 feet; 1 foot = 0.3048 meter
1 hectoliter = 2.837 U.S. bushels;
1 U.S. bushel = 0.363 hectoliter
1 hectare = 2.471 acres; 1 acre = 0.405 hectare
1 kilogram = 2.2046 pounds avoirdupois;
1 pound = 0.4536 kilogram

I (Burley 1988) reviewed the historical development leading to the rise of predictive reclamation modeling as a tool to assist in pre/post-mining landscape planning and design. Significant contributions leading to the development concerning predictive reclamation equations include works by Neill (1979), Pierce *et al.* (1983), Lohse *et al.* (1985), Walsh (1985), Vories (1985), Doll and Wollenhaupt (1985) and Plotkin (1986). In addition to these investigations, Potter (1986) describes two methods to assess the vegetation productivity capacity of the landscape. In the first method an empirical "shot-gun" approach, labeled by Potter as "inductive," is used where a wide array of variables is examined by sampling the landscape and making statistical comparisons/ inferences. The second method is a heuristic approach, labeled by Potter as "deductive," where transects are sampled and the investigator develops generalities about the physical and chemical parameters governing vegetation potential of the landscape. Complementing Potter's descriptions, Reith (1986) provides an explanation of the fundamentals concerning reclamation models across a broad spectrum of reclamation applications ranging from the prediction of rill formation through multiple regression techniques to multi-equation stochastic ecosystem modeling of a grassland. These papers, published by the mid-1980s, represent the general knowledge base prior to the actual development of statistical vegetation productivity models.

Burley and Thomsen (1987) describe a discrete methodology to produce a quantitative reclamation productivity equation. Unfortunately, this approach is labeled by Potter (1986) as the "shot-gun" approach. I would like believe that scientists have been working for generations posing hypotheses and examining variables to determine their importance. Eventually, a somewhat small set of potential predictor variables can be examined in greater refined statistical detail. While some individuals may wish to consider this a shot-gun approach, I would characterize this approach as an indicator of the maturity of soil science and reclamation activities allowing multiple variable studies to be conducted. This advanced multi-variable complexity is evident in other disciplines such a econometrics, wildlife habitat modeling, water quality prediction models, and visual quality modeling.

In my reclamation modeling efforts, the basis for this methodology originated with multivariate statistical concepts presented by Kendall (1939), requiring computationally complex matrix algebra (see Johnson and Wichern 1988). With the advent of the computer to perform matrix algebra operations for dimensions greater than three, multivariate statistical techniques made reclamation productivity development possible. By computing eigenvalues and eigenvectors for all possible dependent variables such as crops and woody plants, an investigator could determine the extent of multi-variable covariance and develop an equation to represent a linear combination of variables to generate a single dependent variable. In other words,

if the first eigenvalue was relatively large and the coefficient loadings of the eigenvector for the first eigenvalue were relatively similar, a simple equation derived from the eigenvector would suggest a linear combination of dependent variables that can be combined into one value per soil type. With one dependent variable value per soil type, it is possible to perform multiple regression analysis using one dependent variable. Gersmehl and Brown (1990) employed Kendall's method to examine multiple crop productivity values across geographic regions in the United States of America. Their work suggests that in the Midwest and Northern Great Plains, multiple crop productivity values often covary across soil types. Their work substantiates the concept that the soil preferences of various agronomic crops are indeed similar. Table 1 lists the vegetation types in Clay County, Minnesota that have been employed by Burley and colleagues to generate dependent variables, generally covarying as a group.

These plant types are employed to predict vegetation productivity. However, the term "vegetation productivity" is a relatively weakly developed construct. In many respects vegetation productivity has been operationally expressed as vegetation yield, such as bushels per acre of harvested seed or feet of new apical terminal shoot growth per year and represents a certain anthropocentric perspective concerning plant growth. A plant physiologist may suggest that an abundance of seeds per acre does not necessarily mean that a vegetation type is internally healthy and an ecologist may suggest that unsustainable lush plant growth is not necessarily a sound ecological condition. Consequently I recognize that there exists the potential to develop new operational constructs for vegetation productivity. Nevertheless, in my vegetation productivity work, I have made that assumption that existing measures of vegetation yield and new plant growth are reasonable indicators of productivity and that my interests lie in the relationships between existing productivity measures and soil parameters. I also assume that these variables can be studied with multiple regression analysis.

In the multiple regression analysis portion of the model building methodology, a single independent variable value for each soil parameter was generated by applying a weighting formula (Equation 1 in Figure 1) suggested by Doll and Wollenhaupt (1985), where the soil parameters in the first foot of a soil profile contribute 40% of a plant's vegetation production, the second foot contributes 30%, the third foot contributes 20%,. the. fourth. foot.. contributes 10%, and the

[Eq 1]

$$V=[(\sum_{i=1}^{12} v_i)*0.4]+[(\sum_{i=13}^{24} v_i)*0.3]+[(\sum_{i=25}^{36} v_i)*0.2]+[(\sum_{i=37}^{48} v_i)*0.1]$$

Where:

V = Weighted Soil Variable Value

v_i = Value for Soil Variable in One Inch Layer of Soil Profile

i = Soil Layer in Profile

Figure 1. Weighting equation based upon soil depth.

remaining layers do not contribute to vegetation growth. With this formula, any soil parameter for a specific soil profile can be measured on a foot by foot basis (even inch by inch) and the investigator can generate a single value for each soil profile, such as a single weighted pH value or a single weighted bulk density value (see Burley and Thomsen 1987). Table 2 lists the typical soil variables employed by Burley and colleagues to generate independent variables. A document written by the Soil Survey Division Staff (1993) describe current methods to measure these variables.

It is also important to recall that most land-use disturbances do not typically affect some plant growth variables, such as climate. Instead, disturbances associated with surface mining activities usually affect the soil. Thus for effective reclamation, a reclamation success predictor such as an equation should focus upon the environmental feature that has been disturbed, the soil. Some investigators and reviewers of vegetation productivity papers have confused "real time crop-yield indexes" with reclamation productivity equations. While real time crop-yield equations can compute the predicted level of vegetation production for a particular year under specific field conditions experienced over the growing season, a reclamation productivity equation predicts the average expected yield across many years of cultivation. This average yield is produced by employing crop yield values that were measured over many years including drought years, wet years, warm growing seasons, and cold growing seasons. This averaging effect thereby negates the yearly variances upon crop yields produced by climate, allowing an investigator to study more closely the influences of soils upon vegetation growth over many growing seasons.

Burley et al. (1989) applied the multiple regression analysis statistical approach to produce a

Table 2. Main effect independent variables and units of measurement from the U.S. Soil Conservation Service (Jacobson 1982 and U.S. Department of Agriculture 1951).

Abbreviation	Factor	Unit of Measurement
FR	% Rock Fragments	Proportion by weight of particles > 7.62 cm
CL	% Clay	Proportion by weight
BD	Bulk Density	Moist Bulk Density g/cm cubed
HC	Hydraulic Conductivity	Inches/hour (1 inch = 2.54 cm)
PH	Soil Reaction	pH
EC	Electrical Conductivity	Mmhos/cm
OM	% Organic Matter	Proportion by weight
AW	Available Water Holding Capacity	Inches/inch, cm/cm
TP	Topographic Position	Scale 0 to 5 Where: 0=Low (Standing Water) 2.5=Mid-slope 5=High (Ridge Lines)
SL	% Slope	(Rise/Run)*100

productivity equation for seven agricultural crops: spring wheat, barley, oats, soybeans, sunflowers, sugarbeets, and grasses/legumes. The database for this investigation was the Clay County soil survey (Jacobson 1982). The result was a reclamation productivity equation with a coefficient of multiple determination R^2 = 0.740. In other words, the regressors explain 74% of the sum of squares variation in the regression model. This equation did not consider woody plants and thus is not an all inclusive vegetation productivity model. Since reclamation often includes woody vegetation for the development of housing or commercial/industrial sites, wildlife habitat, agricultural shelterbelts, and forestry post-mining land-use applications, the development of a productivity model which includes woody plants would be more universally applicable in reclamation planning and design, including the development of prime farmland where woody plants composed of shelterbelts and windrows can be intricate components of an agricultural landscape. An equation developed by Burley (1991) using Burley and Thomsen's (1987) methodology, is presented as the best universal reclamation equation, because it was suitable to a lage number of vegetation types. R^2 for this equation is 0.795, explaining approximately 80% in the sum of squares variation for vegetation from the regression model. Other equations reported include a Clay County, Minnesota sugar beet (*Beta vulgaris* L.) equation (Burley 1990), two equations for Polk County, Florida (Burley and Bauer 1993), a two county equation in the Red River Valley of the North (Burley 1995a), and an equation for Oliver County, North Dakota (Burley *et al.* 1996). Burley (1992) presented a series of issues associated with these equations that may merit further investigation.

In contrast to the approach developed by Burley and Thomsen (1987), an alternative productivity index has been applied by several investigators. This approach is termed the "sufficiency approach" and follows more closely the work of Neill (1979) and Pierce *et al.* (1983). Huddleston (1984) and Henderson *et al.* (1990) review the formative development of this approach. With this approach, single independent variable models are developed to predict soil vegetation productivity. The single variable models have some degree of statistical reliance and may be normalized as illustrated by Wollenhaupt (1985). However, each variable is then combined into a full order interaction term where each independent variable is multiplied together as a group and may be corrected with the geometrical mean (Gale 1987). Several investigators have reported experiments with these full interaction term sufficiency equations (Barnhisel and Hower 1994, Burger *et al.* 1994, Barnhisel et al. 1992, Hammer 1992, and Gale *et al.* 1991). My criticism of this approach is that the equations are heuristically derived and the functions are not statistically validated and are therefore less rigorous. It is not surprising to me that efforts to corroborate equations based upon this methodology have met with mixed results. The research conducted with this heuristic approach has several limitations. First, the variables presented in the

equations may potentially be highly over specified as investigators have not demonstrated the contribution of each variable within the equation and have not followed searching procedures associated with regression modeling. Investigators have not accomplished sufficient equation sifting to eliminate linear models, squared terms, second-order interaction terms, or any other equation configuration. In my opinion, investigators have prematurely jumped to full interaction terms and geometric means. Second, the soils employed in the models are often restricted in external validity applications, meaning that significant results may be limited to a small set of soils studied, sometimes only two or three soil types. The data sets are not broad and thus are not applicable to many soil types. Third the vegetation types studied often consist of one dependent variable type, meaning the reported results are actually only applicable to the crop studied, such as corn or Eastern white pine. These three issues are my current reservations about this heuristic approach.

The sufficiency approach and the multiple regression modeling approach are two current vegetation productivity approaches. Regardless of the shortcomings for either approach they are models that may merit potential use in landscape planning. Numerous states require quantitative reclamation assessment procedures (primarily for coal surface mining reclamation on prime farmlands), suggesting that soil productivity equations are potentially compatible with these quantitative assessment demands and could make a contribution in evaluating the post-disturbance soil environment. Burley and Thomsen (1990) have described the application of a soil productivity equation for reclaiming surface mines. The application illustrates how these equations may be used to interpret landscape reconstruction configurations and how to evaluate the effectiveness of various reclamation treatments.

Discussion

The Dependent Variables

The field data required to conduct the analysis are extensive. All crops and woody plants of interest should be grown on all soil profiles of interest for a period of approximately 10 years to gain a perspective of vegetation performance on soils across climate variability. The time period should include dry years, wet years, and average moisture years. For any individual investigator, this type of data set would be expensive and time consuming to generate; however, the United States Soil Conservation Service (SCS) has conducted similar work for a fair number of counties in the United States of America and can provide a substantial portion of the database for investigators interested in developing a vegetation productivity equation for their region. Since the SCS operates under a county administrative structure, the data sets are naturally organized by county. These data sets are derived from soils on non-mined land. Ideally, a investigator should include results from reclaimed soils also. Nevertheless, I am somewhat disappointed in the conviction expressed privately by some colleagues that a statistical equation derived from soils not disturbed by mining is not valid for reclaimed soils, especially when many of these soils formed within the last 12,000 years, deposited by glacial and related surficial activity. Consequently, the soils that Burley and colleagues have used are relatively new and originated from relatively freshly disturbed materials. In addition, many of these soils are highly disturbed due to recurring fluvial processes, wind erosion, deep tilling, and soil amendments. Thus, these soils are not necessarily undisturbed material weathering in one location for millions of years, such as the soils that may be found in the tropics. Compared to millions of years, 12,000 years is a rather short time span, meaning the glacially deposited soils may have much in common with reclaimed surface mine soils. Concurrently, other than variations in physical parameters such as bulk density and chemical parameters such as nitrogen levels, no investigator has presented any information to suggest that reclaimed soils are intrinsically any different than pre-mine soils. No investigator has suggested that unmined versus mined soils should be a qualitative independent regression variable. If a reclaimed surface mine soil originated from substrate within the bounds of the soils studied to develop a soil productivity equation, and if the mine soil is within the physical and chemical bounds of soils studied to develop the soil productivity equation, then in theory, the soil productivity equation based upon non-reclaimed soils should be applicable to the reclaimed surface mine soil. The work of Burley et al. (1996) suggests that a such an equation may be able to predict productivity on reclaimed soils. Unless an investigator can find the physical or chemical variable that indicates reclaimed soils are substantially different, I will maintain there is no fundamental difference. In fact, within the horticulture and construction industry, soils are rebuilt everyday from highly disturbed areas to support turf growth, wood plants, and vegetable gardens. If these reconstructed soils were really any different from what we know and apply from non-mined soils, then theoretically, landscape construction

activities to build soils for lawns and gardens should not be successful. However, careful soil profile development on construction sites results in successful soil conditions for plant growth. Unless the reclaimed soils generate unsuspected toxic conditions such as high selenium values, surface mine soils are really no different than the collective properties of non-mined soils. For example, in the study areas I have examined, I have found dense clays, acidic clays, alkaline clays, clays on slopes, clays in wetlands, permeable clays, deep clays, and shallow clays, replicating almost any non-toxic soil condition found on reclaimed clay soils. This broad variability of the data set allows reclaimed soils with properties that fit within the parameter bounds of this data set to estimate vegetation performance. I would suggest that those studies that have been unsuccessful at predicting vegetation productivity have either been working with heuristic equations or have not generated a data set with substantial breadth across many years with many soil types and many vegetation types to develop statistically predictive results.

In my studies, crop harvest data and woody plant growth rates are the typical dependent variables employed in a vegetation productivity equation study. Crop harvest data may be in bushels per acre, tons per acre, pounds per acre or any other quantitative harvest value typical for the crop of interest. The woody plant growth rates have been presented by the SCS in feet per years of growth, although other forest measurements or horticultural vegetation growth rates or plant volumes could potentially be employed. In addition, the crop logical types do not have to be consistent. For example, a woody crop species and a mixed crop such as a grasses and legume mix can be employed in the study. The analysis will indicate the statistical relationship between the various vegetation types. This is a difficult concept for some investigators to intellectually grasp because apples and oranges plus many other types of numerical information can actually be compared and combined. Different units of measure across crop types are not necessarily an issue in vegetation productivity equation modeling, because each vegetation type is standardized to a mean of zero and a variance of 1. Then the various crop types and woody plants are examined with PCA. This statistical treatment allows the investigator to determine the number of dimensions necessary to explain the variance across all vegetation types. Ideally, all of the crops of interest covary together and can be combined into one dimension; otherwise an investigator may have to develop an equation for each significant dimension indicated in PCA. In other words, PCA is a data reduction tool that may allow an investigator to determine whether corn, soybeans, wheat and *Fraxinus pennsylvanica* can be combined together or must be analyzed separately. This is an important issue. If all vegetation types do not covary in productivity, then the investigator must develop a large number of individually tailored vegetation productivity equations and the reclamation specialist may reclaim a landscape suitable for one crop but not suitable for another, thereby excluding future production options for the farmer. If the crops do covary, a universal vegetation productivity equation may be possible for the study site.

PCA results typically begin with presentation of an eigenvalue for each dimension in the data set. The largest number of dimensions is equal to the number of variables present in the data set. For example if three crop types are presented for PCA, then the largest number of dimensions is three. This approach is extremely useful for a large number of variables, because the mathematics employed in the technique can examine a data set in a multidimensional space greater than three dimensions. Until the development of the computer, PCA was often limited to three dimensions as the matrix algebra required to compute the eigenvalues was difficult to compute by hand. The important feature of the eigenvalue is that each dimension is orthogonal to every other dimension, meaning that the information associated with each eigenvalue is independent of every other dimension. PCA assumes that the latent roots of the data set are definite and real and the data set is composed of multivariate-normal variables.

The largest eigenvalue is presented as the first eigenvalue in PCA. With standardized variables, the eigenvalue can be no larger than the sum of the variables under study. For example if five crop types are being studied, the largest eigenvalue can be no greater than 5.0. In addition, the sum of all the eigenvalues can be no greater than 5.0. This means that if the largest eigenvalue is 4.8, the sum of the remaining eigenvalues must be equal to 0.2. The proportion of the sum for any combination of eigenvalues indicates the amount of variance explained by those eigenvalues. Suppose two eigenvalues from standardized variables sum to 3.0 and there are four variables in the analysis. These two eigenvalues represent 75 percent of the variance in the data set.

Eigenvalues greater than 1.0 originating from standardized variables are considered to represent significant dimensions. The significant dimensions are then inspected by examining the eigenvector

coefficients. Each variable in the analysis contains an eigenvector coefficient associated with each eigenvalue. The eigenvector coefficient indicates the strength of association the variable has with the eigenvalue. In vegetation productivity analysis studies, the investigator is interested in which variables are associated with which dimensions. Burley *et al.* (1989) discovered that crop variables in their study covaried together and that the eigenvector coefficients were relatively equitable across the first eigenvalue. Thus Burley *et al.* (1989) were able to develop a linear equation to compute the generation of one combined plant growth productivity value per observation case. The eigenvector coefficients indicate the weighting of each crop variable for a specific dimension and range in value from 1.0 to -1.0. The crop value multiplied by the weighting coefficient and then summed with the results from the other crops forms a linear combination equation to build a combined vegetation productivity value for each observation case. Consequently, an investigator may be able to build a data set containing one dependent variable per observation case from a set of many dependent variables.

The Independent Variables

The independent variables are comprised of soil characteristics from each soil profile of interest. A major issue associated with soil characteristics is the variability of any one soil parameter within any soil profile. Where should the investigator measure soil reaction? Doll and Wollenhaupt (1985) suggest that any soil parameter should be measured through the first four feet of the profile and weighted according to the equation identified in Equation 1. While some researchers may question the accuracy of the weighting formula, this formula represents the current understanding and state-of-the-art of profile contribution in vegetation growth, and therefore this is the weighting equation employed in my work.

Analysis

Once the soil profile weighting formula has been employed and the dependent variables have been constructed, the data set is ready for regression screening. In SAS, regression screening can be accomplished with the RSREG procedure. This procedure examines main effect, squared terms, and two variable interaction terms for association with the dependent variable. The most promising variables can then be entered into a regression stepwise procedure to evaluate various combinations of regressors.

The Maximum-Rsquared improvement technique has been determined to be the best stepwise selection procedure (SAS 1982). The investigator must then examine the results of the stepwise procedure to assess which equation is the best from those presented. Typically the investigator will examine the multiple coefficient of determination, preferring larger values which explain a larger percentage of the variance in the data set. Values near 0.4 are considered weak and values near 0.9 are considered strong. In addition, the investigator will examine the p-value for the overall regression and the p-value for each regressor. I recommend that one examine p-values that originate from Type II Sum of Squares, a technique which computes the p-value for the regressor assuming that all other regressors are already in the equation. This is a conservative approach to determining p-values and is consistent with not overestimating the p-value's significance. P-values less than or equal to 0.05 are considered statistically significant. P-values less than or equal to 0.01 are considered highly significant. Therefore, the investigator is searching for an equation which has at least significant p-values and explains as much of the data set's variance as possible.

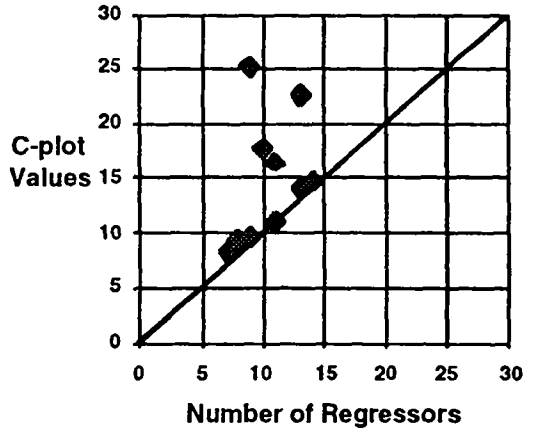

Figure 2. Plot of regressors and C-plot values for equations presented by Burley (1995b).

However, the investigator must be concerned about over specifying and multicollinearity issues associated with regression modeling. One approach in evaluating various equations is to prepare a C-plot of the equations, where the number of regressors are plotted against the C-plot value (Younger 1979). The equations above the 45 degree line are considered equations which are not over specified. Equations that approach the 45 degree line without crossing into the lower portion of the quadrant are preferred over

equations distant from the 45 degree line (Figure 2). Those equations which are further from the origin and along the 45 degree line are preferred over equations close to the origin. This technique can remove much of the collinearity problems inherent in regression modeling. One can also conduct collinearity diagnostics to determine the collinearity associated with any regression equation (SAS Institute 1982). The diagnostic procedure produces a condition index number indicating the degree of collinearity associated with dimensions of a regression equation. A condition index of less than 10 is considered to indicate relatively little or no collinearity (Rawlings 1988). A value greater than 10 indicates weak collinearity. A condition index number between 30 to 100 indicates moderate to strong collinearity. Values above 100 indicate serious collinearity problems. With each condition index, weighting coefficients indicate the level of association for regressor variables. Regressors containing coefficients approaching 1.0 are strongly associated with that particular dimension. More than one regressor with substantial values for a single condition index identifies the potential collinear variables.

[Eq. 2]

Pseudo-Value = k * (parameter estimate calculated from the whole sample) - (k-1)*(parameter estimate calculated from the group with the j sample removed)

Where: k = total number of observation cases

Figure 3. Nonparametric pseudo value jackknife equation.

[Eq. 3]

Standard Error =

$$(\text{Variance of the Pseudo-values}/(k-1))^{0.5}$$

Where: k = total number of observation cases

Figure 4. Nonparametric standard error estimate equation for jackknife procedure.

Analysis of the Analysis

There are several methods to evaluate the estimates of parameters and validity of a particular regression equation. One method is the jackknife approach (Efron 1982). With this technique a nonparametric statistic is produced for each regression parameter by applying Equation 2 in Figure 3. The variance for this sampling procedure is employed to calculate the standard error associated with the estimates of each parameter (Equation 3 in Figure 4, Mathsoft 1988).

In contrast to the jackknife approach, the bootstrap technique samples the data set with replacement to build a larger observational set to compute parameter estimates (Mathsoft 1988 and Efron 1982). For bootstrap analysis, the standard deviations of the coefficient estimates are the standard error estimates for the coefficients. In other words, an estimated distribution of the coefficients can be calculated. Wide distributions indicate unstable coefficients and narrow distributions indicate relatively stable coefficients.

Another procedure which can be employed to study the applicability of a regression equation is the subsampling procedure, where a portion of the data set is removed from the equation development process and then employed later to compare the results from the equation with the subsample.

In addition to the collinearity diagnostics, jackknife techniques, bootstrap techniques, and subsampling procedures, one can inspect the plots of the residuals associated with a specific regression equation. The plots can identify regression assumption violations concerning constant variance and histogram plots can identify violations concerning the normality of the residuals.

Finally, selected vegetation productivity equations developed with the procedures described by Burley and Thomsen (1987) should be assessed with data sets from reclaimed soils. Considering the number of reclamation research centers in North America, one might expect that suitable data sets exist for conducting a reclaimed soil investigation. However, these potential data sets often do not contain observations according to depth, nor do they contain all of the desired soil variables needed by the developed equation. While equations derived from non-mined soils are certainly possible to construct, the methodology described in this paper is directly applicable to unreclaimed data sets also, providing the data set is comprised of all vegetation types across all disturbed soil types. However, it appears that currently, no reclamation center or reclamation investigator has such a large and extensive data set.

Concluding Remarks

This technique represents the basic approach I have folllowed in developing vegetation productivity equations and presents a few more assessment procedures not initially described by Burley and Thomsen (1987). It is my belief that investigators across the globe and especially in North America will employ the methodology that I have described in this paper and build regionally specific equations to study the reconstruction of soils to support plant growth..

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AN INVESTIGATION OF BROWN IRON ORE SPOILS IN TENNESSEE

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Abstract. Many acres, concentrated primarily in the eastern United States, have been disturbed by iron ore mining in the past 250 years. Much of the disturbed soils research to date has been conducted on coal minesoils which is not usually applicable to iron ore spoils. The objective of this study was to compare properties of 140-170 year old brown iron ore spoils (minesoils) in Tennessee to those in adjacent undisturbed soils (native). Two minesoils and two native soils were described and sampled according to National Cooperative Soil Survey Standards. Variances of selected chemical properties were compared between sites using a grid sampling design and Hartley's test for homogeneity of variances. Differences in origin of overburden material and the degree of weathering prior to disturbance accounted for morphological and chemical differences between the minesoils. Minesoil 1 had developed a cambic horizon and minesoil 2 had not developed a diagnostic subsurface horizon. Bridging voids were observed in both minesoils but not in the native soils. Trace elements were highest in the surface horizons of both minesoils and native soils and the buried surface horizon in minesoil 1. Variations for total carbon, total aluminum, barium, manganese, phosphorus, titanium, and zirconium between 40-60cm were not consistently significantly different between minesoils and native pedons at a 0.05 alpha level.

Additional Key Words: Iron Ore Spoil, Minesoil, Minesoil Morphology, Bridging Voids, Minesoil Genesis

Introduction

The current *Soil Taxonomy* system used for soil mapping usually does not adequately identify properties in minesoils (Thurman and Sencindiver 1986; Schafer et al. 1980; Schafer 1979). Characterization studies of minesoil properties are critical to develop appropriate criteria to better separate, classify, and manage minesoils. As a result, better decisions can be made for their profitable use under field crops, pasture, or wood-land.

The differences in characteristics of a disturbed soil from its premined condition depend on several factors including the type of equipment used for mining and reconstruction, the chemical composition of material exposed to/near the surface, the pre-weathered condition of the material prior to disturbance, site factors (i.e. slope, irregular topography, etc.), and the time since disturbance (Potter et al. 1988; Thurman and Sencindiver 1986; Haering et al. 1993; Bell et al. 1994). Properties altered by surface mining generally include soil horizonation, soil structure (affecting macroporosity and permeability), bulk density, coarse fragment content, uniformity in color and chemical properties of parent material, and hydraulic conductivity (Haering et al. 1993; McSweeney and Jansen 1984; Ammons and Sencindiver 1990; Bell et al. 1994; Potter et al. 1988; Indorante and Jansen 1984). The rock type dominating the overburden and the degree of in situ weathering of that material before disturbance by mining strongly determines immediate properties such as pH, extractable bases, and weathering rates of minesoil material (Haering et al. 1993).

Much of the disturbed soils research to date has been conducted on coal minesoils which is not usually applicable to iron ore spoils. Iron ore spoils are not often associated with acidity problems, unlike coal minesoils which typically contain pyrite and sulfur compounds (Smith et al. 1971). Many acres, concentrated primarily in the eastern United States, have been disturbed by iron ore mining in the past 250 years. The Western Highland Rim physiographic region of Tennessee was a

large producer of iron ore during the 1800's (Burchard 1934; Safford 1856). Most of the Western Highland Rim's uppermost geology is of Mississippian age(Hershey and Maher 1963). Thin quaternary-aged loess, Upper Cretaceous gravel, sand, and clay non-uniformly overlie these geologic units (Burchard 1934; Springer and Elder 1980)(Fig. 1).

LOESS (0-1 M.Y.B.P.)
CRETCACEOUS (65-140 M.Y.B.P.)
MISSISSIPPIAN (330-365 M.Y.B.P.) St. Louis Limestone
Warsaw Limestone
Fort Payne Formation
Chattanooga Shale

Figure 1. General geologic cross section of Tennessee's Western Highland Rim physiographic region.

The iron ore mined here is believed to have formed during the Tertiary period. Glauconitic sand beds, of Upper Cretaceous age, are postulated to have been the source of iron (Burchard 1934). Theoretically, iron moved readily through the sand layers, precipitated as iron oxides, and accumulated as ore at the interface of cherty portions of underlying limestone (Safford 1856; Killebrew 1874). The ore was deposited in irregular pockets, veins, or banks of variable size and was commonly located on ridge crests in zones of clay and cherty masses. This depositional pattern may reflect areas of the limestone that had formed sink holes or areas of preferential water flow. Depth to the ore deposits varied between 30 and 100ft parallel to the surface. This was shallow enough for the ore to be mined in shallow open mine pits. Soils disturbed during iron mining operations at that time were not reclaimed and offer an opportunity to study soil development from a known date of disturbance. The objective of this study was to compare properties of 140-170 year old iron ore minesoils to adjacent undisturbed soils.

Materials and Methods

Site selection and History

The study sites were located in Montgomery Bell State Park, in Dickson County Tennessee, near the former Laurel Furnace (Fig 2). The Laurel furnace was a steam cold-blast charcoal type furnace, operating between 1815 and the late 1850's (Smith et al. 1988). The spoils left from the mining operation at the Laurel furnace provided an ideal study site because they have not been influenced by intense anthropogenic activities or natural catastrophes since the mining ceased, and spoil piles were easily identified adjacent to excavated pits.

Field Methods

Study sites were located within 800 meters northwest of the Laurel Furnace remnants. Two minesoil profiles and two soil profiles on adjacent undisturbed soils were described and sampled according to National Cooperative Soil Survey Standards (Soil Survey Staff 1993). Soil samples were taken from the entire thickness of each horizon. A grid design was also sampled for each profile to allow statistical analysis for selected chemical properties. The profile grid was 1 meter deep and 1 meter wide and was divided into 20 cm by 20 cm square sections (Fig. 3). Samples were taken and homogenized from the entire squares that were in columns 2 and 4 and row 3. The variance of selected chemical properties for each sampled column and row was compared between all profiles to determine if they were significantly different between the native pedons and spoils. Hartley's test for comparison of homogeneity of two independent variances was used (Snedecor 1956; Ott 1993). The test compares a calculated F value (F_c) to a tabular F value (F_t). F_c is determined by dividing the larger of the two variances by the smaller as shown by the following equation

$$F_c = \frac{s^2max}{s^2min}$$

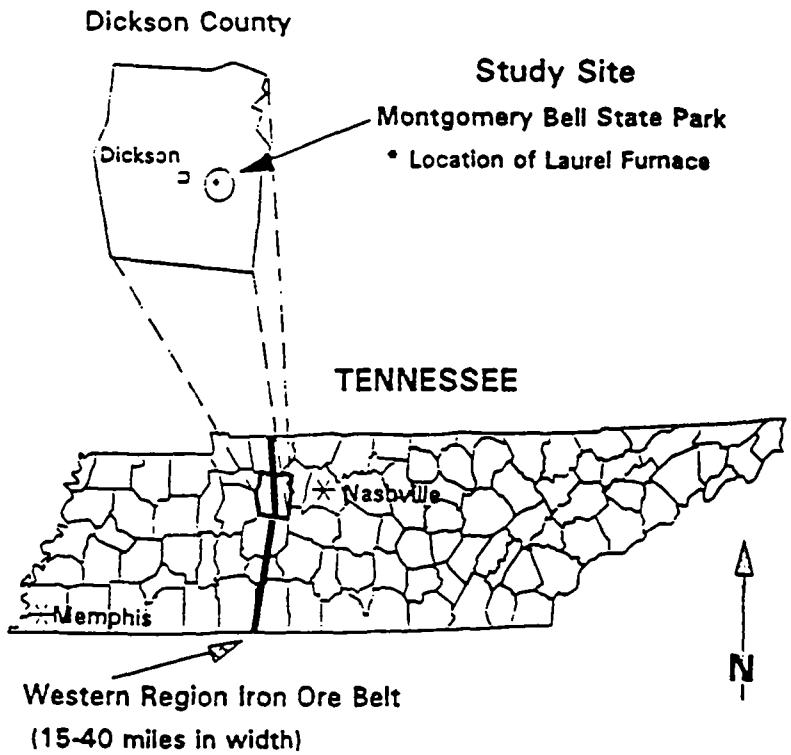

Figure 2. General location map of study area.

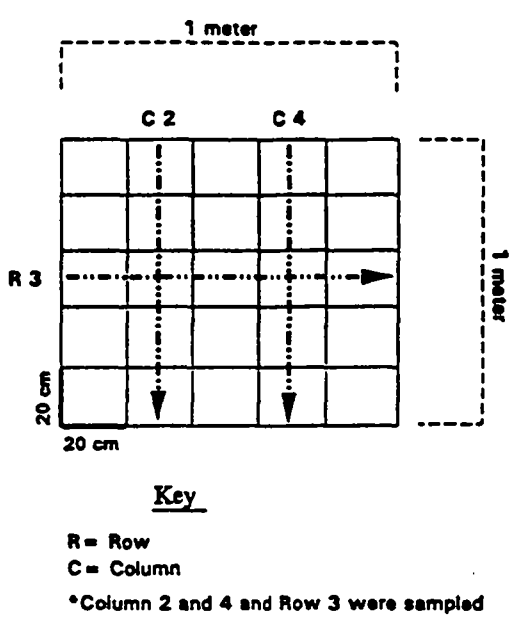

Figure 3. Grid outlay used for vertical and horizontal sampling for statistical analysis.

The null hypothesis states that $\sigma^2_1 = \sigma^2_2$ if F_c is less than or equal to F_t. If F_c is greater (>) than F_t, then the compared variances are significantly different at that chosen alpha level. Note that the degrees of freedom (n=4) were the same for both compared variances and that a 95% confidence level was used for this study.

The undisturbed soils were referred to as native pedon 1 and 2, and the disturbed soils were referred to as spoil 1 and 2. Native pedon 1 developed on the shoulder of a convex side slope, and native pedon 2 developed on a nearly level portion of a ridgetop. Spoil 1 developed in overburden material on a ridgetop, and spoil 2 developed in overburden material in a drainageway.

Laboratory Methods

Horizon samples were air dried, sieved through a 2mm diameter limiting sieve, and homogenized thoroughly (Soil Survey Staff 1984). A sand sieving and sedimentation-pipette method was used to determine particle size distribution of clay, silt, and sand size fractions (Kilmer and Alexander 1949; Gee and Bauder 1986). Fine clay fractions were determined using a combination of the pipette method and centrifugation. Samples with > 0.5% organic carbon were pretreated with hydrogen peroxide (30%) to reduce potential interference of organic matter with sedimentation rates. Concentration of iron oxides, cation exchange capacity (CEC), exchangeable bases, total concentration of 24 elements, and pH were the selected chemical analyses. All procedures included two replications of each sample.

Iron oxides were extracted using a citrate-dithionite (cd) method described by Olsen and Ellis (1982). Exchangeable cations were extracted by saturation with ammonium acetate at pH7 combined with a NaCl vacuum extraction procedure (Jackson 1958; and Baker and Suhr 1982; Hammer and Lewis 1987). The extracts for both procedures were analyzed for Fe and bases using a Perkin-Elmer 5000 Atomic Absorption Spectrophotometer. A direct displacement method and titration with 0.01 M HCl was used to determine CEC (Rhoades 1982) which was measured with a Lab Conco distillation unit. A modification of Nadkarni's (1984) microwave oven acid digestion technique which uses HF and an aqua regia solution (mixture of HCl and HNO_3) to digest inorganic matrices was used to solubilize the soil samples for elemental analysis (Gallagher 1993; Ammons et al. 1995). After the neutralization of the unreacted HF with boric acid, extracts were analyzed with a Thermo Jarrel Ash Model 61 Inductively Coupled Argon Plasma Spectrometer (ICAP-AES) for 24 elements (Al, As, Ba, Ca, Cd, Co, Cr, Cu , Fe, K, Mg, Mn, Mo, Na, Ni, P, Pb, S, Se, Si, Sr, Ti, Zn, and Zr). A combination pH electrode and a 1:1 soil sample to water ratio was used to determine pH (McLean 1982).

For purposes of simplification, results graphed by depth were plotted by midpoint depth for each horizon so that the plot point value represents the value for the entire horizon.

Results and Discussion

Native pedon 1 formed in colluvial material over a residual paleosol, and native pedon 2 developed in loess over a residual paleosol. The horizonation sequence in both native pedons included argillic horizons (Table 1). Percent base saturation, pH, color, and clay content in both paleosols were similar, indicating that the paleosol in both native pedons formed in the same residual material (Tables 1 and 2). Fine clay to total clay ratios, free iron, and cation exchange capacity all reached maximums in the argillic horizons of both profiles (Fig 4).

Spoil 1 developed in backfill material which appeared to have been highly pre-weathered prior to deposition in a spoil. Properties in spoil 1 (2 - 140 cm) such as base saturation, CEC, pH, and matrix color greatly resembled those for the paleosols in native pedons 1 and 2 (Tables 1,2,3 and 4). This suggests that the buried paleosols in the native pedons are the dominant source of overburden material in which spoil 1 formed (Fig 5). Assuming this, the red color due to high iron content in spoil 1 reflect the development of the spoil material (paleosol

Table 1. Morphology of native pedons 1 and 2.

Horizon	Lower Depth	Color	Texture	Structure	Boundary	Cons.	Mn Concr.†	Mottles
Native1	cm	moist				moist		
Oe	3-0							
A	16	10YR 4/4	SiL	2 mgr	cw	vfr	--------	--------
EB	35	7.5YR 5/6	SiL	1 mgr	cw	vfr	--------	--------
Bt1	60	5YR 5/6	gr. SiCL	1 msbk	cw	fr	c,f	c,d 7.5YR 5/6
Bt2	78	7.5YR 5/6	C	2 msbk	cw	fi	--------	f,p 10R 4/8
BC	94	10YR 6/6	vgr. C	1 fsbk	cw	vfr	--------	c,p 5YR 5/8
2Bt	116+	5YR 4/6	gr. C	2 msbk	--------	fi	--------	c,p 10YR 6/6
Native 2								
Oe	3-0	--------	--------	--------	--------	--------	--------	--------
A	21	10YR 4/4	SiL	2 mgr	cs	vfr	--------	--------
BE	32	7.5YR 4/6	SiL	1 msbk	cs	fr	--------	--------
Bt	58	7.5YR 4/4	SiCL	2 msbk	cw	fr	--------	--------
2Ab	95	10YR 5/4	vgr. SiL	1 fgr	cs	vfr	--------	--------
3Bt	108+	5YR 4/6	v.cob. C	2 msbk	--------	fi	--------	f,p 10YR 6/6

† Manganese concretions

Table 2. Chemical properties including exchangeable cations, base saturation, and pH, and total clay of native pedons 1 and 2.

Horizon	Lower Depth	NH_4OAc Extractable Cations				Total Bases	NH_4OAc CEC	Base Saturation	pH 1:1 H_2O	Total Clay
		Ca	Na	K	Mg					
	cm	$cmol_c\ kg^{-1}$						%		%
Native 1										
A	16	0.92	0.04	0.22	0.42	1.60	8.10	19.8	4.7	19.2
EB	35	0.47	0.04	0.13	0.66	1.30	4.40	29.5	4.8	23.9
Bt1	60	1.39	0.04	0.12	1.53	3.08	9.19	33.5	4.7	38.5
Bt2	78	1.84	0.05	0.16	1.37	3.42	12.11	28.2	4.7	54.7
BC	94	1.85	0.05	0.14	1.09	3.13	8.83	35.4	4.6	45.5
2Bt	116+	2.13	0.05	0.20	1.51	3.89	13.23	29.4	4.6	67.0
Native 2										
A	21	0.55	0.08	0.20	0.45	1.28	6.30	20.3	4.5	16.0
BE	32	0.87	0.06	0.21	0.82	1.96	7.54	26.0	4.5	25.1
Bt	58	2.70	0.10	0.19	1.19	4.19	11.67	35.9	4.7	33.4
2Ab	95	0.88	0.10	0.18	0.46	1.61	7.21	22.3	4.7	24.0
3Bt	108+	1.91	0.05	0.25	1.04	3.25	11.24	28.9	4.7	54.1

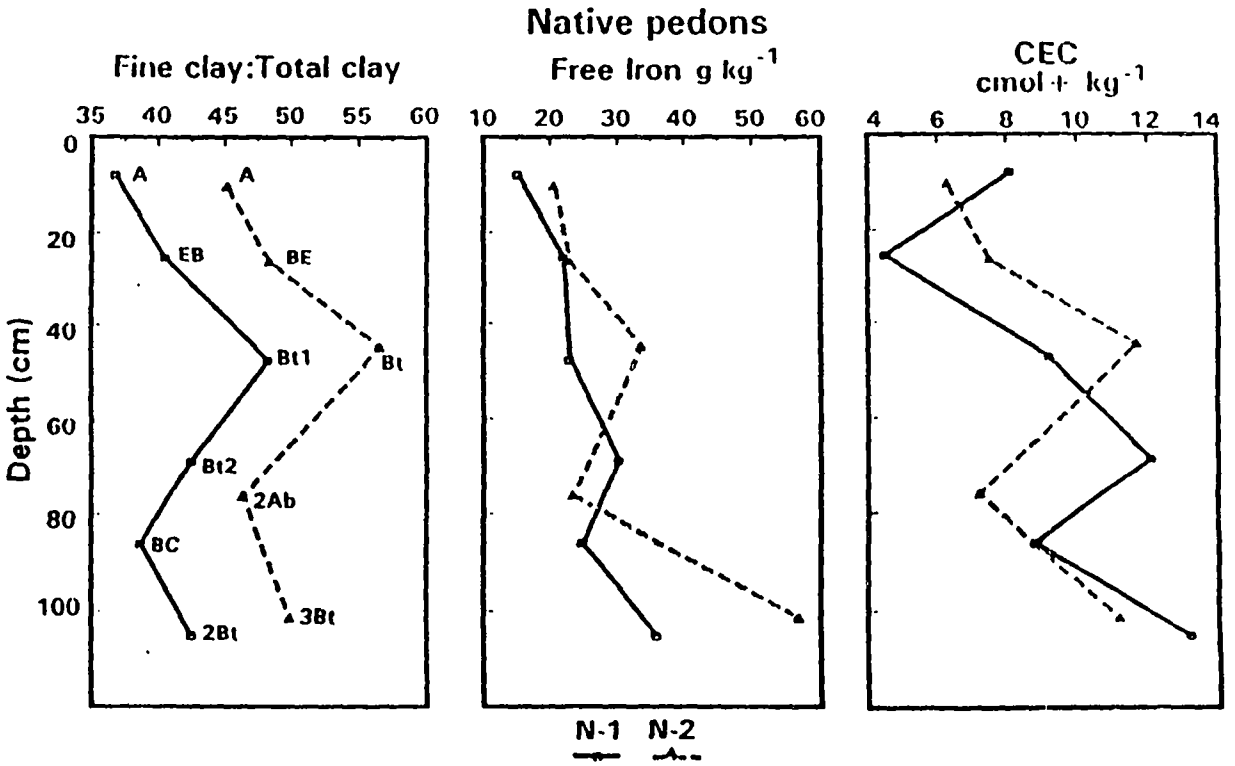

Figure 4. Fine clay to total clay ratio, free iron, and cation exchange capacity distribution by depth in native pedons 1 and 2.

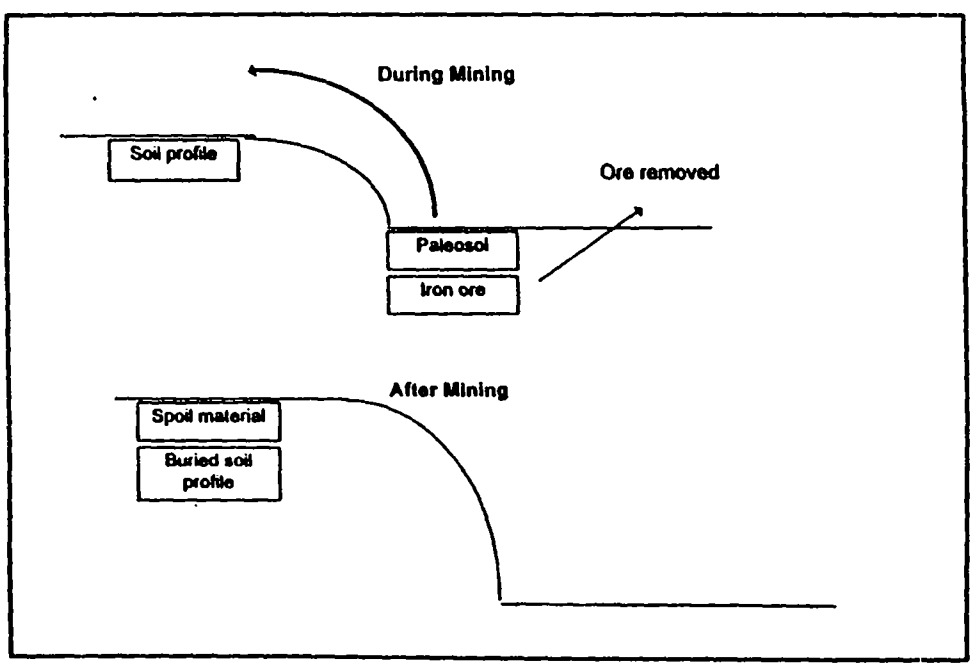

Figure 5. Hypothesized origin of spoil material for spoil 1.

Table 3. Morphology of spoils 1 and 2.

Horizon	Lower Depth	Color	Texture	Structure	Bridging Voids	Boundary	Cons.	Mn Concr.†	Mottles
Spoil 1	cm	moist					moist		
Oe	3-0	----------	----------	----------	----------	----------	----------	----------	----------
A	2	7.5YR 3/1	CL	2 mgr	----------	as	vfr	----------	----------
Bw1	30	5YR 4/6	SiC	2 mgr*	few	cs	l	c,f	c,d 7.5YR 4/4
Bw2	70	2.5YR 4/8	vgr. SiC	2 mgr*	common	gs	l	----------	m,d 7.5YR 4/4
C	140	2.5YR 3/6	SiCL	2 mgr*	common	gw	l	----------	c,d 7.5YR 4/6
2Ab	160+	7.5YR 4/4	SiCL	2 mgr*	----------	----------	fr	----------	----------
Spoil 2									
Oe	2-0	----------	----------	----------	----------	----------	----------	----------	----------
A	6	10YR 3/4	gr. CL	2 mgr	----------	cw	vfr	----------	----------
C1	41	7.5YR 5/6	ex.cob. C	0 m	few	cs	fr	c,f	m,p 2.5YR 4/8
C2	93	7.5YR 4/4**	vgr. C	0 m	few	cw	fr	c,f	c,d 7.5YR 5/8
C3(L)»	108	2.5YR 4/8 10YR 4/6	vgr. C	0 m	----------	cs	fr	c,f	c,d 7.5YR 3/2 c,p 7.5YR 6/8
C3(R)»	108+	7.5YR 5/6	ex.gr. C	0 m	----------	----------	fr	----------	f,p 2.5YR 4/6

† Manganese concretions
*Material did not fit morphology structures in Soil Survey. To avoid confusion of field descriptions, the Taxonomy structure the material resembled most was selected.
**Mottled matrix color
»Material contrasted in horizon between left and right portion of the profile

Table 4. Chemical properties including exchangeable cations, base saturation, and pH, and total clay of spoils 1 and 2.

Horizon	Lower Depth	NH_4OAc Extractable Cations				Total Bases	NH_4OAc CEC	Base Saturation	pH 1:1 H_2O	Total Clay
		Ca	Na	K	Mg					
	cm	$cmol_c\ kg^{-1}$						%		%
Spoil 1										
A	2	19.71	0.27	0.46	4.19	24.64	32.54	75.7	5.1	29.5
Bw1	30	0.43	0.04	0.22	0.67	1.36	12.48	10.9	4.3	45.3
Bw2	70	1.63	0.22	0.22	1.95	4.02	11.92	33.7	4.7	42.8
C	140	1.67	0.06	0.24	1.44	3.42	12.50	27.4	4.7	35.6
2Ab	160+	0.97	0.08	0.19	1.04	2.29	10.83	21.1	4.9	30.7
Spoil 2										
A	6	23.32	0.12	0.64	5.61	29.69	30.87	96.2	6.2	36.3
C1	41	5.73	0.07	0.53	3.30	9.63	14.22	67.7	5.7	53.5
C2	93	1.32	0.05	0.24	0.98	2.58	9.62	26.8	4.6	45.3
C3(R)	108	2.68	0.05	0.14	1.71	4.58	12.05	38.0	4.7	59.2
C3(L)	108+	1.64	0.04	0.19	1.35	3.22	10.54	30.6	4.5	49.8

material) prior to deposition in a spoil. Differences in clay content in spoil 1 compared to the paleosol material indicates that mixing with loess material had occurred during placement of the overburden material. Spoil 2 developed in backfill material that was less pre-weathered and originated from limestone material originally underlying the iron ore before mining occurred. It was abundant with cherty coarse fragments. Coarse fragments in the upper portion of spoil 2 appeared less weathered than fragments at lower depths, reversing the weathering sequence expected from pedogenic processes. This suggests that during the mining process the material was removed and backfilled from higher depths to lower depths, so that material originally at a lower depth (less weathered) was deposited at the upper portion of the spoil.

Bridging voids (packing voids), averaging 1 cm in diameter, were observed between rock fragments and soil material in the subsoil of both spoils. The voids were not present in either native pedon. Both spoils had a higher occurrence of color mottling not associated with anaerobic conditions than the native soils (Table 3). Presumably, these mottles resulted from contrasting materials mixed together during the spoil construction. Moderate medium granular structure was observed in the surface horizon of both spoils. Differences in thickness of the surface horizon between spoil 1 and 2 were a result of differences in landscape position. Spoil 1 was positioned at the shoulder of a hollow more susceptible to surface erosion than the more stable position of spoil 2 (in a drainageway).

Subsoil structure observed in spoil 1 was better described by the "man-made" structure termed "fritted" proposed by McSweeney and Jansen (1984) than the morphology structures outlined in Soil Taxonomy. Massive structure was observed throughout the subsoil of spoil 2. Subsoil horizonation differed between the spoils. A cambic horizon had developed in spoil 1 noted by change in color and fine clay movement (Table 4 and Fig. 6). Diagnostic subsurface horizons did

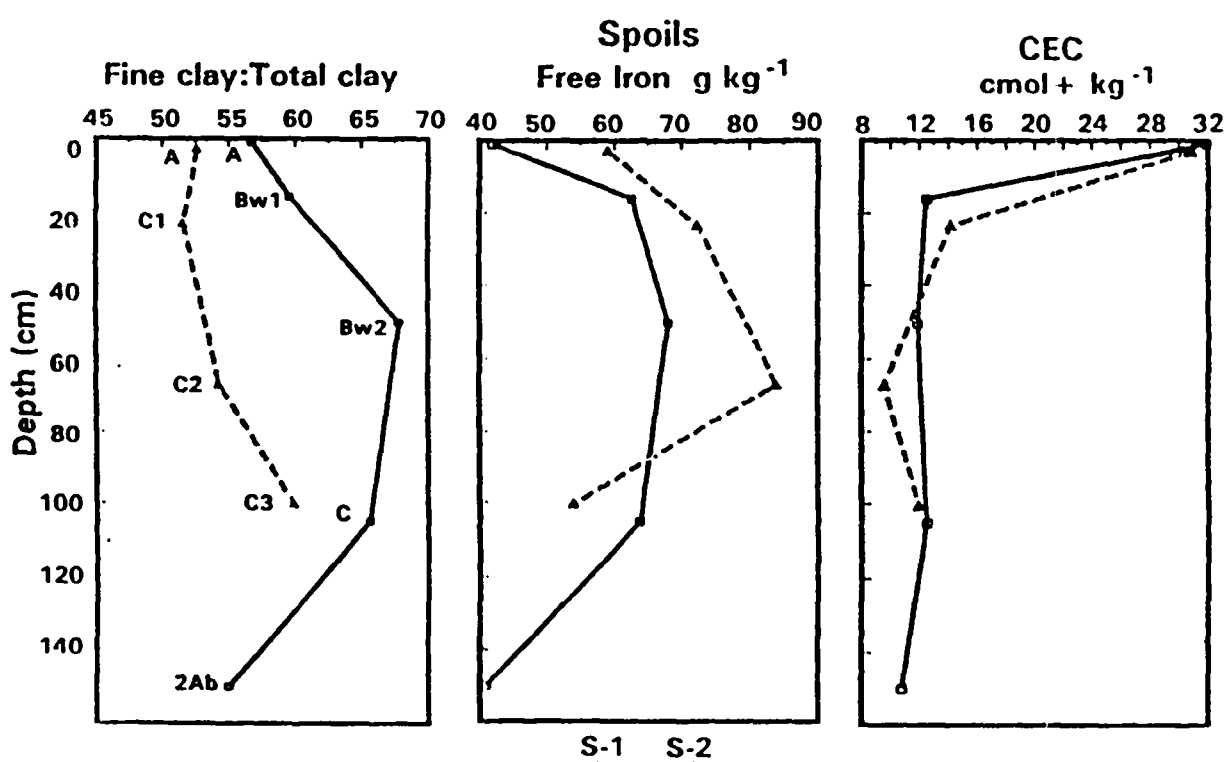

Figure 6. Fine clay to total clay ratio, free iron, and cation exchange capacity distribution by depth in spoils 1 and 2.

not develop in spoil 2. Differences in soil structure and preweathered condition of parent material most likely accounts for the difference in subsoil development between them.

Total iron and free iron concentrations in both spoil profile's were approximately double that of the native pedons (Figs. 4 and 6; Table 5a and b). An "enrichment" of iron oxides was expected in the spoils due to their parent material source having been associated with the origin and mining of the iron ore. Total barium (Ba) was highest in the surface horizons of both native pedons and spoils except for the buried horizon in spoil 1 (Table 5a and b). Total barium almost doubled in the buried A horizon (from the overlying horizon) in spoil 1 but not in the buried A horizon of native pedon 2. Total manganese (Mn) concentrations were significantly highest (α 0.05) in the surface horizons of all four pedons and generally decreased with depth. In spoil 1, total manganese significantly increased in the buried A horizon from overlying horizons. Trace element concentrations in the buried A horizon in native pedon 2 did not tend to increase such as those in the buried A horizon of spoil 1. Lower clay content, lower total carbon content, differences in parent material, and higher coarse fragment contents in the buried horizon in native pedon 2 presumably reduced its ability to retain trace elements compared to the buried horizon in spoil 1. Total copper (Cu) was significantly higher in the surface horizon compared to subsurface horizons in native pedon 1 and both spoils. Distribution of sulfur by depth differed between the native pedons and the spoils. Total sulfur (S) was significantly highest in the surface horizons of the spoils.

Hartley's test indicated if variations of a selected element in row 3 (40 - 60cm) were significantly different between any two sites. Total carbon, total aluminum, barium, manganese, phosphorus, titanium, and zirconium were the elements compared between sites (Table 6). Variation of manganese and zirconium were not significantly different (α 0.05) between any two sites. Variances of total carbon, total aluminum, barium, phosphorus, and titanium were not consistently significantly different between the native pedons and the spoils.

Conclusions

1. The buried paleosols in the native pedons are the primary origin of overburden material in which spoil 1 formed. Mixing of this material with loess had occurred. Less preweathered cherty limestone material originally below the iron ore was the source of overburden material for spoil 2.

2. The differences in origin of overburden material and the degree of weathering prior to placement in a spoil accounted for morphologic differences such as subsoil development, color, carbon content, rock fragment content, and iron oxide concentration between spoil 1 and 2.

3. A cambic horizon developed in spoil 1. A diagnostic subsurface horizon did not develop in spoil 2. The "fritted" structure with a loose (noncoherent) consistence observed in spoil 1, not present in spoil 2, accelerated the rate of fine clay and free iron movement to the degree of forming a weakly developed cambic horizon within 140-170 years. The massive structure throughout spoil 2 did not allow as much water through the profile, limiting profile development.

4. Bridging voids, averaging about 1 cm in diameter, were observed in both soils (minesoils) but not in the native soils.

5. Moderate medium granular structure formed in the surface horizons of both spoils as well as the native pedons.

6. The association of the spoil material with the iron ore during the mining process enriched the spoil materials with iron oxides.

7. In general, trace elements were highest in the surface horizons of both minesoils and native soils and

Table 5a and b. Total elemental concentrations determined using an acid digestion technique for native pedons 1 and 2 and spoils 1 and 2.

Horizon	Lower Depth	Al	Fe	K	Ba	Ca	Co	Cr	Cu
	cm	g kg^{-1}			mg kg^{-1}				
Native 1									
A	16	27.2	17.7	9.3	420	4,510	50	30	70
EB	35	33.8	22.3	9.6	370	1,250	10	40	<2
Bt1	60	51.2	36.5	9.4	320	2,030	<3	70	<2
Bt2	78	61.4	38.9	10.4	280	1,190	20	70	10
BC	94	44.5	29.9	9.9	220	1,520	10	50	10
2Bt	116+	71.6	43.1	12.0	240	1,150	<3	80	<2
LSD*$_{05}$		2.6	2.2	1.5	90	750	30	10	5
Native 2									
A	21	29.6	23.3	11.2	500	1,240	30	30	20
BE	32	42.2	25.3	13.4	460	930	20	30	20
Bt	58	49.7	37.5	11.8	390	1,200	20	40	20
2Ab	95	32.8	29.4	8.9	290	890	20	30	10
3Bt	108+	71.1	59.1	11.9	260	1,000	10	90	5
LSD*$_{05}$		1.1	1.6	0.6	170	360	10	10	20

*LSD= Least Significant Difference calculated using PC SAS (SAS Institute Inc., 1988). If the difference between horizon values is greater than the LSD, the values are significantly different at an alpha level of 0.05.

Horizon	Lower Depth	Mg	Mn	Na	Ni	P	S	Sr	Ti	Zn	Zr
	cm	mg kg^{-1}									
Native 1											
A	16	1,270	1,560	5,560	30	250	870	80	1,580	80	80
EB	35	1,480	600	1,910	30	190	120	30	2,340	40	100
Bt1	60	2,700	140	1,060	60	220	270	50	1,920	210	80
Bt2	78	3,080	160	620	50	230	270	60	2,020	200	80
BC	94	2,810	110	360	60	80	650	60	1,520	130	60
2Bt	116+	3,930	80	260	60	300	470	50	2,050	220	80
LSD$_{05}$		170	50	1,400	20	80	290	30	200	70	10
Native 2											
A	21	1,310	1,230	3,200	20	70	120	60	2,190	50	100
BE	32	1,940	620	2,840	20	50	100	50	2,600	160	110
Bt	58	2,680	510	2,040	20	<5	100	50	2,770	60	110
2Ab	95	1,670	480	1,440	20	<5	40	40	2,130	60	90
3Bt	108+	3,340	90	270	40	90	100	70	2,480	120	80
LSD$_{05}$		200	20	870	20	30	100	20	90	160	5

Table 5a and b. Total elemental concentrations determined using an acid digestion technique for native pedons 1 and 2 and spoils 1 and 2.

Horizon	Lower Depth	Al	Fe	K	Ba	Ca	Co	Cr	Cu
	cm	g kg^{-1}			mg kg^{-1}				
Spoil 1									
A	2	30.9	41.2	4.9	280	5,790	30	40	40
Bw1	30	66.3	66.4	8.6	250	750	10	60	5
Bw2	70	62.1	68.5	4.9	280	1,040	10	50	10
C	140	58.4	75.6	6.8	260	780	20	40	10
2Ab	160+	51.1	44.7	12.0	450	870	20	30	10
$LSD_{.05}$*		16.5	12.4	0.6	170	520	20	10	30
Spoil 2									
A	6	39.4	61.1	8.6	280	8,670	<3	60	30
C1	41	73.0	76.9	13.1	230	2,100	<3	100	<2
C2	93	72.0	94.0	9.0	280	810	20	60	<2
C3(L)	108+	92.6	65.9	11.6	360	1,200	10	120	10
C3(R)	108+	73.1	53.0	14.8	280	970	20	110	10
$LSD_{.05}$*		2.0	1.6	0.7	10	140	5	10	10

*LSD= Least Significant Difference calculated using PC SAS (SAS Institute Inc., 1988). If the difference between horizon values is greater than the LSD, the values are significantly different at an alpha level of 0.05.

Horizon	Lower Depth	Mg	Mn	Na	Ni	P	S	Sr	Ti	Zn	Zr
	cm	mg kg^{-1}									
Spoil 1											
A	2	1,580	820	1,980	20	520	770	70	1,330	140	50
Bw1	30	2,400	380	990	40	110	150	40	2,660	150	90
Bw2	70	2,280	310	870	40	130	110	30	2,820	120	90
C	140	2,180	490	750	40	60	80	40	2,820	90	80
2Ab	160+	2,330	780	2,380	30	30	10	50	2,720	50	100
$LSD_{.05}$		230	120	1,090	20	140	70	30	170	60	10
Spoil 2											
A	6	2,930	1,510	3,890	40	700	1,130	60	980	240	40
C1	41	4,290	790	400	60	580	30	70	2,610	290	70
C2	93	2,180	630	1,040	30	180	110	60	2,690	160	80
C3(L)	108+	3,580	180	360	70	750	180	150	2,650	260	100
C3(R)	108+	4,290	690	320	80	530	50	70	1,900	380	80
$LSD_{.05}$		70	100	90	20	100	90	5	970	70	10

Table 6. List of calculated F values, using Hartley's test of homogeneity of variances, between all sites.

Between Sites	% total carbon	Al	Ba	Mn	P	Ti	Zr
	F_c						
N1/N2	3.0	16.8*	1.8	2.8	3.2	8.8*	2.0
N1/S1	9.0*	27.1*	17.9*	3.7	1.2	1.6	4.4
N1/S2	1.6	1.4	7.5*	5.7	3.3	1.7	1.0
N2/S1	2.9	1.6	9.9*	1.3	3.7	5.5	2.2
N2/S2	4.8	19.0*	4.2	2.0	10.6*	5.3	2.0
S1/S2	14.0*	30.0*	2.4	1.5	2.9	1.0	4.4

*Calculated F value exceeded tabular F value at a 95% confidence level ($F_{\alpha=0.05\ df\ 4,4} = 6.4$) indicating a significant difference between the variations for the two sites for that elemental concentration.

the buried surface horizon in minesoil 1. Of the trace elements, total barium and manganese best identified the buried A horizons.

8. Variations for seven selected chemical analyses at a depth of 40-60cm were not consistently significantly different between spoils and native pedons at a 0.05 alpha level.

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CROP AND SOIL RESPONSES TO SEWAGE SLUDGE APPLIED TO RECLAIMED PRIME FARMLAND[1]

BY

Qiang Zhai and Richard I. Barnhisel[2]

Abstract. Improvements in reclamation of surface mined prime farmland may be abtained by adding sewage sludge to topsoil and subsoil. This prime farmland reclamation study was done in western Kentucky. The experiment was conducted to investigate effects of the sludge amendment to topsoil and subsoil on soil and crop responses. The experiment showed, in most cases at highest application rates, that the sludge addition significantly increased the soil organic matter, total N content, and available P levels. However, water holding capacity, CEC, and exchangeable cations were not significantly affected. Higher microbial populations and activites were also obtained. The wheat biomass, tiller number, tissue N, grain N, grain yield, and N removal in grain were well correlated with application rates of sewage sludge. Corn also responded positively to additions of sewage sludge. The corn ear-leaf N concentration, grain yield, and grain N removal increased with application rates of sewage sludge. Experiments indicated that topsoil and subsoil sewage sludge addition was beneficial practices in terms of increasing crop yield and improving some soil properties.

Additional Key Words: organic matter, reclamation, sewage sludge, soil properties, wheat and corn yields.

Introduction

The original soil ecosystem, structure, horizons, microbial community, and fertility may be drastically affected by surface mining (Barnhisel et al., 1987). Return of the productivity of mined prime farmland to conditions equal to or greater than those of the original soil prior to mining is a requirement of Public Law 95-87, the Surface Mining Control and Reclamation Act of 1977 (SMCRA).

In the past 10 years, several projects have been reported on the use of inorganic fertilizer to meet the requirements of recovery of soil properties and crop yield for reclaimed prime farmlands, but few have included organic amendments. Organic materials, such as sewage sludge, peat, and wood residue have been extremely successful only for spoil reclamation compared to inorganic fertilizers (Fresquez and Lindemann, 1982). The high organic matter of waste amendments largely ameliorates the physical conditions of mine spoil. Soil properties that benefit from sludge incorporation includes water holding capacity, bulk density, water-stable aggregates and hydraulic conductivity. (Sopper, 1992).

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Chemical properties of mine spoils affected by organic amendments include raising acidic spoil pH, and increasing CEC, N and P nutrients, but effect on K, Ca, and Mg levels varies depending on the characteristics of amendments (Sopper, 1992). Microbial population and activities are often stimulated by organic amendments (Sopper, 1992). However, few studies if any have quantitatively measured the effect of organic waste application on physical, chemical, and biological properties as well as on crop yield of disturbed prime farmland.

The objectives of this study were to determine the effects of sewage sludge applied to either topsoil and/or subsoil on soil physical, chemical, and biological properties, as well as wheat and corn responses seeded to a reclaimed prime farmland soil following surface mining. The hypotheses were that adding sewage sludge in topsoil or subsoil may significantly improve soil physical, chemical, and biological properties and increase crop yields.

Materials and Methods

Field procedures

The mine is located in Henderson County, western Kentucky. The replacement of spoil and subsoil was completed in 1989 and seeded to cover crops including wheat and tall fescue. The reconstruction of topsoil was done in the summer of 1992 following sewage sludge additions. The original soil disturbed by surface mining activities was the Grenada (fine-silty, mixed, thermic Glossic Fragiudalfs) silt loam soils. The topsoil and subsoil were separately stockpiled and were mixed during the soil removing and replacing processes.

Two sewage sludge experiments were initiated, one in which sewage sludge was applied only to the subsoil (Zhai, 1995), and the other in which sewage sludge was applied to both subsoil and topsoil. Only the results from the latter experiment will be reported here. The sewage sludge was from the Henderson Water and Sewage Utilities. Characteristics of the sludge are shown in Table 1. The experiment design was completely randomized block with three replications.

Table 1. Physical and chemcal properties of sewage sludge.

Water content	(%)	42.9
Total N	(g kg^{-1})	26.3
NH_4-N	(mg kg^{-1})	536
NO_3-N	(mg kg^{-1})	330
Cd	(mg kg^{-1})	0.16
Cr	(mg kg^{-1})	0.64
Cu	(mg kg^{-1})	2.23
Pb	(mg kg^{-1})	0.16
Ni	(mg kg^{-1})	0.80
Zn	(mg kg^{-1})	0.16

Sewage sludge at rate 22.4 Mg ha^{-1} was uniformly applied to the subsoil prior to the topsoil replacement. Then 0, 11.2, and 22.4 Mg ha^{-1} sewage sludge were applied to the topsoil. All sewage sludge was applied as described above and incorporated to a 15 cm depth in both the topsoil and subsoil with a chisel plow.

The topsoil (0-15 cm) was sampled after sludge application on September 6, 1992. Every soil sample from each plot was taken by a hand sampler and was made up of by ten cores taken throughout the study period.

On October 16, 1992, the entire area received a broadcast application of N fertilizer (NH_4NO_3) at a rate of 70 kg N ha^{-1} and planted to wheat (*Triticum aestivum*, L.) at seeding rates of 136 kg ha^{-1}. On May 11, 1993, this experiment site was divided into two parts. One part was retained for wheat yield measurements. For the other part, the wheat was killed with a mixture of atrazine (4WDL) and "Roundup" at a rate of 3.0 and 1.8 L ha^{-1}, respectively.

Corn (Zea mays L.) was planted at a seeding rate of 59,000 kernels ha^{-1} using a Kinze no-till planter.

Topsoil (0-15 cm) was sampled again on September 1993 for biological analyses. The soil sampling procedure was the same as described above.

Wheat tiller numbers were counted from 20 randomly selected plants from each plot and their above-ground plant tissue was sampled on May 5 for biomass determination and chemical analyses. Wheat grain was harvested on July 3, 1993 with a plot combine. Yields were measured based on the standard 12.5% moisture. A wheat grain sample (200 g) was collected from each plot at harvest for chemical analyses.

Corn ear-leaves were taken during late silking growth stage on July 29, 1993, from six randomly selected corn plants in each plot. Corn grain was harvested on October 2, 1993 with the same plot combine. Yields were measured and based on the standard 15.5% moisture content. A corn grain sample (200 g) was collected at harvest from each plot for chemical analyses.

Soil analyses

Soil bulk density was determined by Blake and Hartge, except that the core size was 4.5 cm (1986). Water-holding capacity was determined using the method developed by Klute (1986). The soil samples for chemical analyses were air-dried, and ground to pass a 2-mm sieve. Soil pH was determined with a pH meter on a 1:1 deionized water to soil mixture (McLean, 1982). Exchangeable Ca, Mg, K, and Na and CEC were determined with the neutral M NH_4OAc method (Thomas, 1982). Available P was determined using the Bray-I (Bray and Kurtz, 1945). Organic matter (Nelson and Sommers, 1982) and total nitrogen (Bremner and Mulvaney, 1982) were performed by the University of Kentucky Soil Testing Laboratory, Division of Regulatory Services.

Soil microbiological analyses were performed using moist soils. Total heterotrophic and facultative anaerobic organisms were enumerated by the Most Probable Number method (Alexander, 1982). The numbers of organisms were calculated based on oven-dry soil. Microbial biomass (C) was determined using a chloroform fumigation procedure (Parkinson and Paul, 1982). Soil respiration rate (CO_2 evolution rate) in the laboratory was measured by gas chromatography on a Varian 3700 gas chromatograph equipped with Porapak Q column operated at 80°C (Rice and Smith, 1982).

Plant analyses

Wheat tissue and corn ear-leaf samples were dried and ground to pass a 0.425 mm sieve. The samples were then analyzed for N, P, K, Ca, and Mg which were all determined based on Jones and Case (1990).

Statistical analyses

Statistical analyses were performed using the Statistical Analysis System (SAS Institute, 1985). General linear Models (GLM) procedure was used to obtain the analyses of variance based on the experiment design.

Results and Discussion

Soil responses

The effects of sewage sludge application on soil physical, chemical, and biological properties are shown in Tables 2. The significant increase in soil organic matter, total N, and Bray I-P, is attributed to the sludge application. Soil water holding capacity, pH, CEC, and exchangeable Ca, Mg, K, and Na were not significantly altered. Sewage sludge application to subsoil had similar effects on soil physical and chemical properties as observed in the sludge applied to

Table 2. Physical and chemical properties of topsoil (0-15 cm) after sewage sludge application on August 25, 1992.

Sludge (Mg/ha)	WHC† (%)	PH	OM (%)	TN (-- mg/kg	Bray I-P -)	CEC (--------	Ca cmol(+)/kg	Mg ---------)	K	Na
0	25.2	5.0	0.82	1110	5.5	9.4	3.26	1.34	0.17	0.07
11.2	25.8	5.0	1.15	1470	6.8	9.8	4.04	1.35	0.19	0.07
22.4	25.2	4.9	1.11	1490	9.2	10.1	4.03	1.35	0.17	0.07
$LSD_{(0.1)}$	NS‡	NS	0.23	267	1.7	NS	NS	NS	NS	NS

† WHC, water holding capacity; OM, organic matter content; TN total nitrogen content.
‡ Not significantly different.

Table 3. Total heterotrophic and facultative anaerobic organisms, microbial biomass, and respiration rates after sewage sludge application to topsoil (0-15 cm) on 7 Sept., 1993.

Sludge Rate (Mg/ha)	Total Heterotrophs (10^7/g)	Facultative Anaerobs (10^7/g)	Microbial Biomass C (mg/kg)	Respiration Rate (mg CO_2/kg h)
0	0.31	0.12	224	30.0
11.2	4.49	0.56	267	39.7
22.4	10.9	1.19	331	55.3
$LSD_{0.1}$	4.25	0.31	58	9.6

topsoil. Significantly higher levels of organic matter, total N, and Bray I-P and pH were observed compared to the control, but other measured properties were not significantly affected.

These results were consistent with those that have been reported for spoil reclamation with sewage sludge. Sewage sludge has increased organic matter content, total N, and Bray I-P in reclaimed spoils according to Sopper (1992).

Total heterotrophic and facultative anaerobic organisms, microbial biomass C, and respiration rates of topsoil affected by sewage sludge application are shown in Table 3. Total heterotrophic and facultative anaerobic organisms increased. The significant increases were observed at the highest sludge application rate. Microbial respiration and biomass C also increased in response to increased sludge addition.

Sewage sludge addition stimulated microbial populations. This agrees with previous research that sewage sludge increases the microbial population of disturbed soils (Sopper, 1992). The enhancement of microbial activity in amended soil is primarily due to two factors. Sewage sludge has high microbial activity with the high bacterial numbers and large amounts of fungal hyphae as well as a nutrient-rich composition (Sopper, 1992).

Microbial activity usually increases as sludge application rate increased as observed in this study. However, this is not always consistent, particularly for sewage sludge. Inhibition of microbial activity was observed by others when heavy loading rates were used (Sopper, 1992). The

inhibition is usually caused by a high content of heavy metals and/or salt. Such an inhibition was not observed here, probably due to the lower contents of heavy metals and salt (Table 1), as well as the lower application rates used here as compared to the sludge studies conducted by other researchers.

Wheat response

Wheat response to sewage sludge were in Table 4. A significant increase in grain yield was observed as a result of sewage sludge addition. A significant difference was not observed between application rates of 11.2 and 22.4 Mg ha^{-1}. The application rates had significant linear and quadratic effects on yields. Grain N removal followed the same basic trend that was observed for grain yield. The amount of N removed significantly increased as the sludge application rate increased.

Plant biomass and tiller numbers are usually good indicators of wheat growth after improvement of soil nutrient status and other physical and chemical properties by organic amendment (D.A. Van Sanford, personal communication). Application of sewage sludge significantly increased wheat above-ground biomass and tiller numbers, but there was no significant difference between the 11.2 and 22.4 Mg ha^{-1} application rates. Significant linear and quadratic responses to application rates were obtained for both biomass and tillers.

Wheat tissue (flag leaf) or grain N concentrations can be used as indicators of N nutrition (Grove et al., 1984). In this study, greater wheat tissue and grain N concentrations were observed in sewage sludge treatments than in the control. Significant linear and quadratic responses of tissue and grain N concentrations to application rates were observed. Tissue and grain N concentrations were significantly correlated with each other (r=0.87***). Tissue and grain N concentration have been frequently used to evaluate crop N nutrition.

Forage crops, as well as others such as silage corn, have significantly responded to sewage sludge in undisturbed prime farmlands (Smith and Peterson, 1982). Winter wheat response to sewage sludge may not be the same as the forage crops because wheat growth is often more sensitive to available soil N. Wheat grain yield is often low due to either low levels of available soil N or too much N. Grove et al. (1984) reported that the lush growth and lodging of winter wheat was due to excessive available soil N. The results of tissue and grain N concentration, grain yield, N removal in the grain have been related to available soil N in studies of wheat response to N fertilizer (Grove et al., 1984). The response of tissue and grain N concentration, grain yield, grain N removal, and tiller numbers in this study indicates that the increase in each of those parameters is well correlated with the addition of sewage sludge.

The sludge addition did not significantly affect wheat tissue P, K, Ca, and Mg. The sewage sludge contribution to crop growth is probably due to supplying available N from the sludge mineralization. This is consistent with results reported by Smith and Peterson (1982) who demonstrated that crop response to sewage sludge is mainly due to available N supplied by the sludge, although sewage sludge may also provide other nutrients.

Corn response

Grain yields and grain N removal in 1993 are shown in Table 5. Grain yields were significantly increased as the results of sewage sludge addition. There was a linear and quadratic rsponse of yield to application rates.

Table 4. Wheat about-ground biomass, tillers, tissue composition, grain yield, and N removal as affected by sewage sludge applied to topsoil and subsoil.

Sludge†	Biomass	Tiller	Tissue Composition					Grain	Grain	N
			N	P	K	Ca	Mg	N	Yield	Removal
(Mg/ha)	(g/tiller)	(#/plant)	(g/kg)					(g/kg)	(Mg/ha)	(kg/ha)
0	2.37	4.30	8.43	1.26	15.5	2.11	0.93	15.1	3.31	49.7
11.2	2.84	5.90	10.2	1.18	17.5	2.01	1.05	17.4	4.24	73.0
22.4	3.38	6.40	11.1	1.25	15.7	1.97	1.09	19.6	4.65	89.9
$LSD_{(0.1)}$	0.64	0.82	0.55	NS‡	NS	NS	NS	16.8	0.52	13.3
C.V.	12.8	8.54	4.49	4.6	22.6	17.6	11.0	5.55	8.16	10.8
Linear	**	***	***					***	***	***
Quad.	**	**	***					***	**	***

† Rates of sewage sludge applied to topsoil were 0, 11.2, and 22.4 Mg ha^{-1}, respectively, and 22.4 Mg ha^{-1} was applied to the subsoil for the entire Experiment II. Linear, linear effect; Quad., quadratic effect.

‡ NS, not significantly different; *, significant at $p<0.1$; ** significant at $p<0.05$; ***significant at $p<0.01$.

Table 5. Ear-leaf composition, grain yield, and N removal in 1993 as affected by organic wast application to topsoil and subsoil.

Trt	Ear-leaf					Grain	N
	N	P	K	Ca	Mg	Yield	Removal
	(g/kg)					(Mg/ha)	(kg/ha)
0	24.7	2.27	19.2	4.02	2.02	4.03	49.5
11.2	25.7	2.33	23.9	4.36	1.87	5.25	73.1
22.4	26.8	2.47	22.4	4.37	2.24	6.22	85.6
$LSD_{0.1}$	0.7	NS†	NS	NS	NS	0.42	6.1
C.V.	2.0	9.7	9.3	6.7	15.7	2.3	4.3
Linear	*		*	*		***	***
Quad.						***	***

† linear effect; Quad., quadratic effect. NS, not significantly different; *, significant at $p<0.1$; ***significant at $p<0.01$.

Corn yields were measure again in 1994. Significantly higher yields were abserved in sludge rates of 5 Mg ha^{-1} (72.9 Mg ha^{-1}) and 10 Mg ha^{-1} (92.9 Mg ha^{-1}) than control plots (52.2 Mg ha^{-1})

Grain N removal followed the trend observed for grain yields. The sewage sludge addition significantly increased grain N removal.

The application of sewage sludge increased ear-leaf N concentrations. Significant sludge effects occurred between application rates. A significant linear relationship was found between ear-leaf N and application rates. The level of ear-leaf N seemed normal. It was between the critical values of 28.2 g kg^{-1} reported by Steele et al. (1982) and the 23 g kg^{-1} reported by Dirks and Bolton (1980).

Ear-leaf P concentration was not significantly affected by the application of sewage sludge, although there was a slight increase as application rates increased. This was probably due to the fact that sewage sludge also supplied available P. Ear-leaf K was significantly increased in both trials, but Ca and Mg was not significantly affected by sewage sludge amendments in either study.

The addition of sewage sludge to soil had apparently increased of ear-leaf N concentration, grain yield, and grain N removal. The N from sludge mineralization likely contributed to these responses. The response of corn to sewage sludge has been reported in

similar experiments on unmined lands. A single application of sewage sludge at rate 7.5 to 30 Mg ha^{-1} following sorghum-sudan significantly increased the corn dry-matter yield at all sludge application rates (Bouldin et al., 1984). Data for yield and N uptake indicate that mineralization of sludge N was supplying a significant amount of N for corn uptake. The amount of N recovered by crops reached 6 to 19 percent of the total N applied, and these values were generally lower than those found for conventional N fertilizer (Bouldin et al., 1984).

The experiment indicated that topsoil and subsoil sewage sludge amendments improved some soil physical, chemical, and biological properties, and crop N nutrition and yields.

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Acid Mine Drainage and Minesite Salinity in Australia. D.J. Williams.

Abstract: Acid mine drainage and minesite salinity are, arguably, the most serious threats posed to the environment by open-cut mining and mineral processing. Acid mine drainage arises when sulphur, usually in pyrite form, contained in the ore and/or waste materials, is exposed to oxidation, the products of which are subsequently leached out by water as acid. As the pH of the system drops, bacteria begin to catalyse the chemical reactions, and any metals present go into solution. Open-cut mining and mineral processing activities also lead to the storage of water on the surface, where evaporation and leaching raise its salinity. At many mines in Australia in dry climates, soil covers over potentially acid generating waste rock dumps are being actively trialed. The aim is to maintain the soil cover in a saturated state by covering it with a sacrificial mulch of loose-dumped inert material. The mulch also serves to take up rainfall, preventing excessive runoff which may cause erosion of the soil cover. In the coalfields of Queensland, Australia, measured pH and salinity levels have been found to vary widely between minesites, within any given minesite, and with changing climatic conditions.
Additional Key Words: dissolved metals, open-cut mines, pyrite oxidation, soil cover.

Organic Matter and Nutrient Accumulation in Reclaimed Kaolin Mine Soils of Georgia.

R. B. Haddock, L. A. Morris, R. L. Hendrick, and E. A. Ogden

Abstract: Deficiencies in soil nutrients, particularly nitrogen, commonly result from replacement of natural soil profiles by overburden low in organic matter and essential elements. Reclamation success depends largely upon creating nutrient reserves, establishing adequate nutrient cycles, and developing a functional soil profile. Loblolly pine (*Pinus taeda* L.) stand productivity and soil characteristics of reclaimed kaolin mines in Georgia were evaluated along a chronosequence. Estimated site index (age 25) ranged from 11.3 m on poor sites to 24.1 m on productive sites. Projected volume at age 25 ranged from 119 m^3/ha on poor sites to 280 m^3/ha on productive sites. These values indicate that growth on many mined lands is comparable to growth on adjacent non-mined lands. Soil profile development below the surface soil was nonexistent on all sites and differences in soil texture and color did not adequately explain observed differences in site productivity. Foliar deficiencies in phosphorus, potassium and several micronutrients suggest that availability of nutrients and potential mineralization explain the observed differences in site productivity.

Plant Materials and Amendments for Controlling Wind and Water Erosion on a Fly Ash Disposal Area: TVA Colbert Fossil Plant, Alabama. Jimmy J. Maddox, David Behel, John M. Soileau, and Jimmie Kelsoe.

Abstract: Fly ash disposal sites adjacent to fossil fueled generating plants are subject to wind and water erosion which increases the operation and maintenance costs. Gullies and unstable areas in the disposal sites require expensive leveling and filling practices. Test evaluated both warm- and cool-season cover crops established by either sod or seed. Amendments to the ash consisted of composted poultry litter (CPL), soil, soil+CPL, fertilizer and beneficial soil microbes including mycorrhizal fungi. Turf sods (419 Bermuda, Emerald zoysia, and Raleigh St. Augustine) were compared in greenhouse and field studies. Six legumes and 12 grass species were tested in the greenhouse as seeded cover crops using similar amendments and raw poultry litter (PL). Legumes grew better with CPL and soil amendments and grasses grew better on PL and soil amendments possibly due to differences in N requirements and N supply. Cool season crops generally grew faster than warm season species in the greenhouse tests. Amendments should be mixed with the FA to ameliorate the effects of boron and salt toxicity and to increase the water holding capacity. Bermuda sod grew faster than either St. Augustine or Emerald zoysia,but requires more water. A microbial amendment increased dry matter yields of bermuda sod 2 to 3 times after 40 to 60 days over unamended controls. Microbial amendments may be justified on an economic and sustainable basis. A field study is assessing the environmental and cultural requirements to grow a cover crop on an annual basis.

Additional Key Words: restoration, ash ponds.

Reclamation of Acidic, Denuded Copper Basin Land: Revegetation Performance of Phosphate Rock VS Other Nutrient Sources. J. M. Soileau, F. J. Sikora, J. J. Maddox, and J. J. Kelsoe.

Abstract: Open pit smelting of Copper ore about 100 years ago resulted in approximately 9,300 ha of severely eroded, very acidic (pH 4.0 to 5.0) soils at Copper Basin, Tennessee. Along with other essential nutrients, phosphorus (P) amendments are critical for long-term productivity and sustainability of vegetation on this depleted soil. A field study was conducted (1992-1995) to compare revegetation from surface-applied North Carolina phosphate rock (PR) and triple superphosphate (TSP) at 20, 59, and 295 kg P ha^{-1}, and to determine benefits of starter NPK tree tablets. The experimental design consisted of 7.3 x 9.1 m replicated plots, each planted to 20 loblolly pine seedlings and aerially seeded with a mixture of grasses and legumes. Tree survivability was high from all treatments. Through the third year, tree height and diameter increased with increasing P to 59 kg P ha^{-1}, without fertilizer tablets. There were no pine growth differences between PR and TSP. Weeping lovegrass has been the dominant cover crop through 1995, with increased stimulation to tree tablets and surface P. Tall fescue (KY 31), sericea lespedeza, and black locust responded more to PR than to TSP. Surface soil pH increased, and 0.01 M $SrCl_2$ extractable Al decreased, with increasing rate of PR. For future loblolly pine plantings in the Copper Basin, this study suggests there is no benefit to applying both tree tablets and surface P at rates above 59 kg P ha^{-1}. For reclaiming land with high acidity and low P fertility, PR has significant benefits. In reclaiming steep, gullied land, there is great potential for aerial application of PR and/or pelletized liming agents.

Additional Key Words: land restoration, P fertilization.

Improved Methods for Acid Base Accounting Utilizing Geospatial Modeling.

Michael DiMatteo(1),and Mark Gaylord(2), ((1)Pennsylvania Department of Environmental Protection, (2)Dynamic Graphics, Inc.).

Abstract: A comparison between the traditional mass averaging Acid Base Accounting (ABA) procedure and a geospatial ABA method is applied to a mine site in Fayette County, Pennsylvania. The traditional method for determining mass-weighted or volume-weighted acid base accounting (ABA) typically uses Theissen polygons and straight line interpolation to determine an area of influence for each overburden drill hole on a mine site. The traditional ABA method has been used to characterize overburden in terms of net neutralization potential (NNP) and to define the location of toxic strata. Recent developments in geospatial modeling, available as part of the Office of Surface Mining's TIPS system, allows significant improvement in the estimated property distribution, estimated concentration and mass and/or volume calculations. A geospatial approach, which includes better representation of topographic variability and incorporates stratigraphic and geologic structural features in a model, results in improved volumetric calculations, and ultimately, in a more precise site characterization. Improved site characterization provides the basis for integrating material handling and alkaline addition plans into an overall mine plan.

Additional Key Words: geospatial modeling, acid base accounting, alkaline addition, material handling

Age and Site-Related Patterns of Carbon Allocation to Wood, Foliage and Roots on Reclaimed Kaolin Mines in Georgia.

A. Legerski, R. Hendrick, E. Ogden.

Abstract: We quantified merchantable stand volume, leaf area indices (LAI), and root densities in twelve *Pinus taeda* L. forests growing on reclaimed kaolin mines in Georgia. Stands were 11 to 36 yrs old, and formed two productivity classes. Projected wood volume on the poor sites ranged from 104 to 142 m^3/ha and from 164 to 298 on the better sites. LAI was not related to age on either the good or poor sites. LAIs reach their maximal values early in stand development (<12 yrs). Fine roots (0-1 mm) within the upper 1 m were most abundant on the poorer sites. Roots were nearly absent at depths>50 cm in stands aged 14 or less, but nearly equal in density to surface roots in the older stands. Stand age was strongly and negatively related to fine root density on both site types. Densities of larger, more perennial roots (1-2 mm) decreased with stand age on poor sites and increased on good sites. Stand productivity is closely related to LAI, and root densities show that trees allocate more energy into fine, absorbing roots on the poorer sites. Deep fine root densities indicate that trees must explore progressively greater volumes of soil to meet water and nutritional needs in reclaimed soils.

Mycorrhizal Fungi + Trees -- Practical Beneficial Tools for Mineland Reclamation.

C.E. Cordell, D.H. Marx, and B. Jenkins.

Abstract: Successful consistent revegetation of drastically disturbed sites (ie., acid coal spoils and mineral waste dumps) throughout the U.S. and several foreign countries has been achieved by using the biological "tools" -- Mycor Tree™ seedlings and native shrub and grass species. These unique plants are custom-grown in bareroot and container nurseries with selected mycorrhizal fungi. On disturbed sites, specific mycorrhizal fungi such as Pisolithus tinctorius (PT) or VAM provide significant benefits to the plant symbionts through increased water and nutrient absorption, decreased toxic materials absorption, and overall plant stress reduction. During the past 15 years, the Ohio Division of Reclamation--Abandoned Minelands Project (AML) has utilized the combination of the PT fungus and reforestation to significantly improve the effectiveness and reduce the cost of AML projects. Since 1981, over 3.5 million PT-inoculated pine and oak seedlings have been planted on approximately 2,500 acres of unreclaimed AML sites. Tree survival has averaged over 85 percent in the PT-inoculated tree plantings with few failures as compared with less than 50% survival and over 75% failures in previous plantings with the same noninoculated tree species. From 1981 to 1995, the 2,348 acres reclaimed in Ohio have cost approximately $832,000.00. Traditional reclamation would have cost approximately $14 million and represents a 94% cost reduction. The total PT reforestation cost in 1995 was $354.00 per acre and the added cost of the PT-inoculated seedlings is approximately 13% ($45.00/acre) or $.03 per seedling. This is a minute expense when compared to conventional AML reclamation costs ($6,000/acre). Interest in the application of this natural environmentally-friendly technology to mineland reclamation programs throughout the U.S. and abroad is expanding.

COMPARATIVE ASSESSMENT OF GROUNDWATER CHEMISTRY EVOLUTION AT THREE ABANDONED MINES AND ITS PRACTICAL IMPLICATIONS

Jaroslav Šolc[1]

Abstract: The reclamation effort typically deals with consequences of mining activity instead of being planned well before the mining. Detailed assessment of principal hydro- and geochemical processes participating in pore and groundwater chemistry evolution was carried out at three surface mine localities in North Dakota—the Fritz mine, the Indian Head mine, and the Velva mine. The geochemical model MINTEQUA2 and advanced statistical analysis coupled with traditional interpretive techniques were used to determine site-specific environmental characteristics and to compare the differences between study sites. Multivariate statistical analysis indicates that sulfate, magnesium, calcium, the gypsum saturation index, and sodium contribute the most to overall differences in groundwater chemistry between study sites. Soil paste extract pH and EC measurements performed on over 3700 samples document extremely acidic soils at the Fritz mine. The number of samples with pH <5.5 reaches 80%-90% of total samples from discrete depth near the top of the soil profile at the Fritz mine. Soil samples from Indian Head and Velva do not indicate the acidity below the pH of 5.5 limit. The percentage of samples with EC > 3 mS cm^{-1} is between 20% and 40% at the Fritz mine and below 20% for samples from Indian Head and Velva. The results of geochemical modeling indicate an increased tendency for gypsum saturation within the vadose zone, particularly within the lands disturbed by mining activity. This trend is directly associated with increased concentrations of sulfate anions as a result of mineral oxidation. Geochemical modeling, statistical analysis, and soil extract pH and EC measurements proved to be reliable, fast, and relatively cost-effective tools for the assessment of soil acidity, the extent of the oxidation zone, and the potential for negative impact on pore and groundwater chemistry. Interpretive techniques used in this project can be used in premining environmental characterization to allow for better mining and reclamation design and reduce costs associated with improperly designed material deposition.

Additional key words: reclamation design, environmental impact

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Injection of Alkaline Ashes Into Underground Coal Mines for Acid Mine Drainage Abatement. William W. Aljoe.

Abstract: The injection of fly ash, scrubber sludge, fluidized-bed combustion (FBC) ash, and other alkaline waste materials into abandoned underground coal mines for acid mine drainage (AMD) abatement has obvious conceptual appeal. This report describes three ongoing projects -- one each in West Virginia, Maryland, and Ohio -- where field demonstrations of the technique are being pursued in cooperative efforts among State and Federal agencies and/or private companies. The West Virginia site produces AMD that is causing the State to incur very high treatment costs and operational problems, especially in the storage and disposal of metal hydroxide sludges that result from treatment. In an attempt to achieve a more cost-effective long-term remediation scheme, the State is working with local coal companies and power generators on a plan to fill part or all of the mine voids with slurries of fly ash and/or FBC ash. At the Maryland site, the goal is to demonstrate the feasibility of completely filling a very small underground mine with an FBC ash slurry. The information gained here will determine whether large-scale AMD remediation can be achieved if deep mine disposal of ash is incorporated into the design of a new FBC power plant. In Ohio, it is believed that sealing and complete flooding of a relatively small mine will be able to curtail its AMD production. In order to accelerate the flooding process and insure that alkaline conditions will prevail in the mine, a waste slurry of calcium hydroxide from a nearby source will be injected into the mine voids in conjunction with mine sealing.

Additional Key Words: mine hydrology, site characterization

Closure of the Brewer Gold Mine by Pit Backfilling. Anne Lewis-Russ (1), John F. Lupo (1), Jon M. Bronson (2), Joe M. Flynn (2) and B. G. "Bud" Long (3). [(1)Titan Environmental Corporation, Englewood, CO, USA; (2) Titan Environmental Corporation, Tempe, AZ, USA; (3) Brewer Gold Company, Jefferson, SC, USA].

Abstract: Brewer Gold Mine, located in north-central South Carolina, is implementing an innovative reclamation plan that includes backfilling the main Brewer open pit with mine waste. The primary goals of the closure are to reduce acid rock drainage and minimize or eliminate long-term operation and maintenance requirements by restoring the site property to approximate pre-mining topography. The plan calls for consolidation of approximately 200 acres of waste into approximately 20 hectares (50 acres). Much of the material to be backfilled into the pit, including spent heap leach material and waste rock, has acid-generating potential. Therefore, the backfill design integrated geochemical properties of the backfill materials with expected post-closure conditions. A prime consideration was the final position of the water table. Since mining at the site started in the early 1800's, no records exist of the original groundwater levels. Therefore, the design incorporates a large anoxic limestone drain to control the final groundwater level. Additional amendments are to be placed in targeted areas of the backfill to maximize their utilization. A low-permeability cap system that includes a geosynthetic clay liner has been designed to limit infiltration into the backfill.

LEACHATE GENERATION, QUALITATIVE TRENDS AND GROUNDWATER CONTAMINATION POTENTIAL IN DRY DEPOSITS OF FLY ASH

Jaroslav Šolc[1]

Abstract: Globally, coal conversion solid residues (CCSRs) currently pose a potentially serious environmental problem due to the increasingly large volumes to be disposed of and the complexity of chemical and physical behavior. During the past 5 years, several experimental field test cells were designed and constructed under the sponsorship of the U.S. Department of Energy using conventional industrial landfill practices as guidelines. Two 2.5-m-thick cells containing approximately 700-1000 tons each of waste were emplaced. Advanced process residues from Ohio (LIMB combustion process) were used in this study. The experiments focused on 1) engineering behavior, 2) diagenesis of buried advanced process residues, and 3) long-term impact of natural leaching processes on the surrounding environment. An intensive monitoring effort generated a substantial project database consisting of baseline chemical characterizations of initial and buried fly ash; x-ray diffraction (XRD), x-ray fluorescence (XRF), and scanning electron microscopy (SEM) analysis of mineral composition and transformations within the cells; American Society of Testing and Materials (ASTM) leachate chemistry; physical properties of soil and ash drill cores; water chemistry of runoff; and pore waters from soil vacuum pressure lysimeters and groundwater in surrounding monitoring wells. On-site meteorological, borehole permeameter, and moisture density data also support quantification of the extent of percolation and flushing. Although permeabilities of $X.10^{-7}$ ms^{-1} initially slowed downward migration of strongly alkaline pore waters (pH of 10 to 12), with time, sufficient water was collected in the vacuum pressure lysimeters. A time-dependent decrease of Na, K, SO_4, and Cl concentration, with an attendant decline in pH and EC, indicates intensive leaching near the top of the ash profile and downward displacement toward the cell base. Monitoring wells used to assess leachate transport to the saturated zone did not indicate significant change in groundwater chemistry; however, pore-water chemistry in base soils beneath the cells reflects an impact from cell leachate.

Additional key words: contaminant migration, mobility, environment

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Backfill Hydrologic Characteristics in a Tennessee Area Surface Coal Mine

by
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The hydrologic characteristics of backfill in a Tennessee area surface coal mine provide useful information concerning the aqueous environment in a recovering water table within a reclaimed surface coal mine. Chemical analyses of water samples from observation wells emplaced in a reclaimed surface coal mine provide a measure to evaluate the overall effectiveness of post-mining reclamation activities. Graphs of chemical parameters including Total and Dissolved Fe, Total and Dissolved Mn, Alkalinity, Sulfate and Specific Conductance reveal that reclamation efforts can be effective in reducing adverse chemical impacts. Trends for Total and Dissolved Fe in water from several observation wells indicate that following an initial increase in Fe concentrations, there was a general decreasing trend to levels below the standard for concentrations permitted for discharge in natural streams. Graphs of chemical parameters also reveal that many traditional rules of aquatic chemistry are not always followed during water table recovery in a reclaimed surface mine. This is most evident in the case of metal concentrations. Increases or decreases in metal concentrations have no apparent correlation with pH levels. Metal concentrations fluctuate as pH remains relatively constant. Total alkalinity levels also fluctuate as pH remains relatively constant. Graphing water quality over time provides valuable information on observation well water quality which can be useful in regards to maintaining enforced water quality standards and as a tool in understanding the chemistry within a surface mine hydrologic regime.

Erosion Control on Tailings Dams Using Soil Bioengineering. Moreen Miller.

Abstract: Increased awareness of the complexity of ecosystems has generated the need to integrate all facets of mining into the development of sustainable environmental management plans. Environmental management goals that may previously have concentrated on final closure of the mine are now becoming part of the preliminary planning and day-to-day activities of new mines, and are being integrated into planning goals of established operations. These management goals must address all components of a mining operation, including those relating to long-term maintenance of tailings areas. The long-term maintenance of tailings dams presents a challenge to the operations and reclamation divisions of minesites. Reducing, and eventually eliminating long-term maintenance and monitoring of surface slope stability on tailings dams has been identified as a cost-effective measure that can lead to the diversification of natural systems. An application of bioengineering techniques on a tailings dam at the INCO Limited Copper Cliff tailings complex is the focus of this poster. Soil bioengineering is a slope stabilization technique that uses dormant live plant material as the major engineering component. Unrooted live vegetation is installed on the slope, which provides immediate stabilization. Roots and shoots then develop to form a permanent vegetative cover and root reinforcing matrix. Bioengineered slopes strengthen over time as the root matrix develops, and provide a microclimate for the invasion of natural species, thus encouraging biodiversity and successional development. Live fascines were installed in the fall of 1994 on the face of the dam to slow the velocity of surface water runoff and to facilitate subsurface water movement in saturated conditions. The structures performed extremely well during their first growing season, and are currently being monitored to assess long-term performance.

Additional Key Words: soil bioengineering, erosion control, fascine, biodiversity, live construction, slope stabilization

WQEP - A COMPUTER SPREADSHEET PROGRAM TO EVALUATE WATER QUALITY DATA [1]

by

ROBERT G. LIDDLE [2]

A flexible spreadsheet Water Quality Evaluation Program (WQEP) has been developed for mining companies, consultants, and regulators to interpret the results of water quality sampling. In order properly to evaluate hydrologic data, unit conversions and chemical calculations are done, quality control checks are needed, and a complete and up-to-date listing of water quality standards is necessary. This process is time consuming and tends not to be done for every sample. This program speeds the process by allowing the input of up to 115 chemical parameters from one sample. WQEP compares concentrations with EPA primary and secondary drinking water MCLs or MCLG, EPA warmwater and coldwater acute and chronic aquatic life criteria, irrigation criteria, livestock criteria, EPA human health criteria, and several other categories of criteria. The spreadsheet allows the input of State or local water standards of interest. Water quality checks include: anion/cations, TDS_m/TDS_c (where m=measured and c=calculated), EC_m/EC_c, EC_m/ion sums, TDS_c/EC ratio, TDS_m/EC, EC vs. alkalinity, two hardness values, and EC vs. Σ cations. WQEP computes the dissolved transport index of 23 parameters, computes ratios of 26 species for trend analysis, calculates non-carbonate alkalinity to adjust the bicarbonate concentration, and calculates 35 interpretive formulas (pE, SAR, S.I., unionized ammonia, ionized sulfide HS-, pK_x values, etc.). Fingerprinting is conducted by automatic generation of stiff diagrams and ion histograms. Mass loading calculations, mass balance calculations, conversions of concentrations, ionic strength, and the activity coefficient and chemical activity of 33 parameters is calculated. This program allows a speedy and thorough evaluation of water quality data from metal mines, coal mining, and natural surface water systems and has been tested against hand calculations.

Key Words: Water Quality, Computer Programs, Chemistry, Coal mining, Mining

Introduction

The evaluation of water quality data can be just as tedious and time consuming as collecting the data itself. Poor interpretation of data may lead to erroneous decision making. The science of hydrologic data interpretation is a complex and ever changing. New techniques are constantly being developed to evaluate the data.

Some evaluation methods work better for certain types of water characteristics, but not all (See Hem, 1992, and Hounslow, 1995). Other methods may work well but either do not catch on or fall out of favor. Often, the methods of interpretation exist but we do not have the time to apply them to all our analyses. The hydrologist must also possess a diversity of skills to be a good evaluator of data. The task requires one to be an organic chemist, an inorganic chemist, a mathematician, a computer specialist, a statistician, knowledgeable in regulatory standards, experienced with data interpretation and methodology, and possessing good intuition. Unfortunately few of us are this diverse.

A computer program was developed to aid in water data interpretation of single samples. The program is geared toward inorganic constituents with

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some water quantity calculations included. The program is designed to be flexible, easily expanded, and receptive to modification. It does not handle large sets of data; rather, it is designed to look at one or two samples at a time and provide a thorough interpretation. A full suite of chemical analyses is necessary to optimize the program's capability.

Methods

A Water Quality Evaluation Program called WQEP has been developed. The program is based on a water quality evaluation program developed in OSM's Kansas City Field Office in 1989 (Liddle, 1989). The program has been continually modified and debugged and has been used in-house on hundreds of sites. The program results were frequently compared with those obtained by hand calculation or other software programs such as Wateq4F (Ball, 1991) and HC-GRAM (McIntosh, 1986). The program was developed for Lotus 1,2,3 spreadsheets and can be converted to other spreadsheet platforms such as Excel.

Lotus was chosen because the program could be easily changed by inexperienced programmers to tailor the model to different regions and ever changing water quality criteria. Lotus can handle small databases depending on the computer's memory and has data transfer capabilities with many other databases such as dBase, FoxPro, Oracle, ParLens, and SQL Server (Bradley, 1994, p. 1028). From a practical standpoint, since the program is designed to analyze full suite water chemistry, the cost of data collection usually limits the number of samples to be analyzed. This makes most databases of full suite water chemistry manageable.

Tedious hydro-chemical calculations were programmed to allow the user to concentrate on the results not the calculations. Data interpretation methods were obtained from numerous publications. Greenberg (1992), Hem (1985), and Minear (1982) provides good discussions about the interpretation techniques. General water quality references showing examples include Benefield (1982), Stumm (1996), Novotny (1994), Manahan (1991), and Bodek (1988). General chemistry calculations can be found in Sawyer (1967). Statistical methods for water resources can be found in Helsel (1992), Koch (1970), Barnes (1994), and Haan (1977). Soil, rock, and water chemistry interrelations are discussed in Loeppert (1995), Stumm (1995), Bohn (1985), and Garrels (1965). The pH concept is explained by Westcott (1978). Case studies, water quality summaries, and hydrologic investigation reports were used to find additional ideas. Examples include Likens (1995) and USGS (1993). Being an avid reader of USGS Water Resources Investigations and Open-File Reports is beneficial in seeing a variety of interpretation techniques and illustrations from many hydrologists.

Model Components

The program conducts several types of quality assurance checks, comparisons with water quality standards, data manipulations, calculation of other parameters, and graphical presentations. WQEP allows the importation of 115 chemical parameters including dissolved and total species; metals and trace elements; conventional pollutants; and physical parameters. They include the following:

acidity	carbonate	lead	phosphate	sulfate
alkalinity	cesium	lithium	potassium	sulfide
aluminum	chloride	magnesium	radium	TDS
ammonia	chromium	manganese	radon	Temp.
antimony	cobalt	mercury	scandium	thallium
arsenic	conduct.	molybdenum	selenium	titanium
barium	copper	nickel	SS	TSS
beryllium	cyanide	nitrate	silica	uranium
bicarb.	fluoride	nitrite	silver	vanadium
boron	gallium	ORP	sodium	yttrium
cadmium	hardness	DO	strontium	zinc
calcium	iron	pH	sulfate	zirconium

The model immediately conducts calculations to predict the value of other species such as un-ionized ammonia, ionized sulfide HS, sulfide (S2-), bicarbonate, hardness, etc.

WQEP runs more than 10 different quality assurance checks to evaluate the quality of the data. Water quality checks include: anion/cations, TDS_m/TDS_c, EC_m/EC_c, EC_m/ion sums, TDS_c/EC ratio, TDS_m/EC, EC vs. alkalinity, two hardness values, EC vs. Σ cations, and ionic strength comparisons.

The program contains the latest water quality criteria, standards, and threshold values cited in the

literature. Federal Register notices were compiled on each parameter to understand the latest regulatory requirements rather than rely on outdated water quality tables.

The program automatically compares each of the 115 chemical parameters to up to 16 categories of criteria and standards. These include:

- EPA Drinking Water MCL
- EPA Drinking Water MCLG
- EPA Secondary Drinking Water MCL
- EPA Warmwater Aquatic Criteria
  - Chronic
  - Acute
- EPA Coldwater Aquatic Criteria
  - Chronic
  - Acute
- Irrigation Criteria (various sources)
- Livestock Criteria (various sources)
- EPA Human Health Criteria
  - Water and Organisms
  - Organisms Only
- National Ambient Levels (Bodek)
- State Sediment Criteria
- Other State Criteria
- Optional Criteria

The program displays a "WARNING!" note beside each parameter that exceeds some water quality criteria and shows which column the parameter exceeded.

Several aquatic life criteria for metals vary depending on the hardness or alkalinity of the receiving stream. The values are derived from regression equations relating the two parameters. The program allows the user to enter a hardness value and the criteria are automatically calculated. The parameters calculated are as follows:

CHRONIC CONCENTRATIONS SHOULD NOT EXCEED:

$$\mu g/l\ Cd = e^{(0.7852\ [\ Ln\ (\ Hardness\ mg/l\ )\ ]\ -3.490}$$

$$\mu g/l\ Cr(III) = e^{(0.8190\ [\ Ln\ (\ Hardness\ mg/l\ )\ ]\ +1.561}$$

$$\mu g/l\ Cu = e^{(0.8545\ [\ 1\ Ln\ (\ Hardness\ mg/l\ )\ ]\ -1.465}$$

$$\mu g/l\ Pb = e^{(1.266\ [\ 1\ Ln\ (\ Hardness\ mg/l\ )\ ]\ -4.661}$$

$$\mu g/l\ Ni = e^{(0.8460\ [\ Ln\ (\ Hardness\ mg/l\ )\ ]\ +1.1645}$$

$$\mu g/l\ Ag = e^{(1.72\ [\ Ln\ (\ Hardness\ mg/l\ )\ ]\ -6.52}$$

$$\mu g/l\ Zn = e^{(0.8473\ [\ Ln\ (\ Hardness\ mg/l\ )\ ]\ +\ 0.7614}$$

more than once every three years on average.

ACUTE CONCENTRATIONS SHOULD NOT EXCEED:

$$\mu g/l\ Cd = e^{(1.128\ [\ Ln\ (\ Hardness\ mg/l\ )\ ]\ -3.828}$$

$$\mu g/l\ Cr(III) = e^{(0.8190\ [\ Ln\ (\ Hardness\ mg/l\ )\ ]\ +3.688}$$

$$\mu g/l\ Cu = e^{(0.9422\ [\ 1\ Ln\ (\ Hardness\ mg/l\ )\ ]\ -1.464}$$

$$\mu g/l\ Pb = e^{(1.266\ [\ 1\ Ln\ (\ Hardness\ mg/l\ )\ ]\ -1.416)}$$

$$\mu g/l\ Ni = e^{(0.8460\ [\ 1\ Ln\ (\ Hardness\ mg/l\ )\ ]\ +3.3612)}$$

$$\mu g/l\ Zn = e^{(0.8473\ [\ Ln\ (\ Hardness\ mg/l\ )\ ]\ +\ 0.8604)}$$

more than once every three years on average.

The program also calculates bicarbonate alkalinity, carbonate, alkalinity, hydroxide alkalinity, free carbon dioxide, and total carbon dioxide, and noncarbonate alkalinity. The dissolved transport index (DTI) (Horowitz, 1991) is calculated for 23 parameters based on the dissolved and total species present in the water. More than 25 different ionic ratios are calculated such as sulfate/TDS, Cu/Mo, Ca/SO_4, etc. The program allows user defined ratios to be calculated.

Index parameters such as ESP, SAR, Morphoedaphic index (Brocksen, 1992), Langelier saturation index, etc. are also calculated. Conversions into equivalents and moles are conducted. Dissociation constants, dielectric constant, pE from eH, and other tedious calculations are automatically done.

The model provides some rough dual species solubility calculations but is not intended to replace hydro geochemical models such as WATEQ4F (Ball, 1991) or Solmineq.88 (Kharaka, 1988). Chemical activities are calculated using the Debye-Huckel ionic strength method. Ion solubility product constants K_{sp} and activity product constants K_{iap} are printed for 56 species.

One additional feature allows the input of a flow measurement in cfs or gpm. This value is used for the mass balance mixing model calculations. Downstream water chemistry can be predicted based on baseline data and the addition of one or more discharges knowing the flow rates.

Mass loading calculations are also calculated in several ways. The program takes the concentration of the constituent such as iron, adjusts for the flow rate, and calculates loading in pounds per day, pounds per year, tons per day, and tons per year. By entering a sediment density value into the program, the potential sediment accumulation is calculated in cubic feet per month and cubic feet per year. This is beneficial for dam and reservoir siltation estimations.

WQEP automatically produces a histogram of major ions and a STIFF diagram for water quality fingerprinting. These graphics along with the normal graphing features of a spreadsheet program allow a quick and effective illustration of data. The nature of a spreadsheet allows additional water evaluation techniques to be added to the program as they are developed with little effort.

Conclusions

A Water Quality Evaluation Program (WQEP) was developed for special use in situations where full suites of major ions, metals, trace elements, and field parameters are collected for individual samples. The program does many water interpretation calculations that are generally too tedious and time consuming to do for every sample. The program enables quality assurance and detailed evaluations to be quickly conducted so that the hydrologist can concentrate on the other aspects of the data.

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LIMESTONE DRAINS TO INCREASE pH AND REMOVE DISSOLVED METALS FROM AN ACIDIC COAL-MINE DISCHARGE IN PENNSYLVANIA[1]

Charles A. Cravotta III[2] and Mary Kay Trahan[3]

Extended Abstract

Limestone drains are used to increase pH and remove dissolved metals from acidic mine drainage (Hedin et al. 1994). However, the chemistry of mine drainage is variable and geochemical processes within these treatment systems are poorly understood. To resolve uncertainties about some of the factors affecting chemical reactions within limestone drains, three identical drains were constructed in parallel to treat acidic drainage from an abandoned coal mine in east-central Pennsylvania (fig. 1). A static mixer was installed to enable aeration of the inflow to one or all three drains. Samples of water were collected at the inflow to the drains, at points within the drains, and at the outflow from the drains. The samples were analyzed to evaluate the rate of dissolution of limestone and the extent of hydrolysis and precipitation of iron, aluminum, manganese, and other dissolved metals. The inflow rate was varied to determine any effects on the rates of dissolution and precipitation reactions and the transport of reaction products through the drains.

As the water flowed through the drains, pH and concentrations of alkalinity and calcium increased, and concentrations of acidity and of dissolved and suspended iron and aluminum decreased, and concentrations of sulfate, magnesium, manganese, nickel, and zinc did not change (figs. 2-4). These water-quality trends and computed saturation indices (table 1) indicate that throughout the drains, limestone ($CaCO_3$) can dissolve and neither calcium nor sulfate can precipitate as gypsum ($CaSO_4 \cdot 2H_2O$), because the water is undersaturated with respect to these solids. However, as pH increases, first iron, and then aluminum, rapidly undergoes hydrolysis and precipitates as hydroxide compounds within the drains. At slow flow rates, iron and aluminum hydroxides tend to settle out of suspension and do not adhere strongly to (armor) limestone surfaces. Concentrations of dissolved manganese, nickel, zinc, and other trace metals are not limited directly by precipitation or sorption reactions. Only minor amounts of the trace metals are removed from solution by sorption onto iron and aluminum hydroxides at pH < 5 (Coston et al. 1995) because protons compete for sorption sites under the relatively low-pH conditions at which iron- and aluminum-hydroxide solids precipitate. Under "closed system" conditions in which hydrolysis products including protons and carbon dioxide can not escape the drains and are retained as reactants, limestone surfaces rapidly dissolve and hence armoring is avoided, despite oxygenated conditions. Water quality changed most near the inflow (fig. 3), because the rate of limestone dissolution slows with increasing pH and increasing concentrations of calcium and bicarbonate (Plummer et al. 1979). Although the concentration of calcium added to treated effluent was greatest at slower flow rates (fig. 5), or increased residence time, the overall average rate of dissolution of limestone, 4.4 kg/day, was independent of residence time. If the rate of dissolution remains constant, the 40,000 kg of limestone used to fill the drains theoretically will last 25 years.

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Table 1. Measured composition and calculated saturation indices for water samples[1]

[Water-quality constituents in milligrams per liter, except for pH; saturation index and $\log P_{CO_2}$, unitless; -- no data; < less than; > greater than]

Sample	Measured Water-Quality Constituent								Saturation Index[2]				
Dist-ance	pH	SO_4	Alka-linity $CaCO_3$	Ca	Mg	Fe	Mn	Al	Cal-cite	Gyp-sum	Fe-$(OH)_3$	Al-$(OH)_3$	log P_{CO2}
0 ft	3.50	210	0	28	26	1.70	3.0	0.79	--	-1.6	-0.8	-6.1	--
5 ft	4.30	210	0	36	26	.680	2.9	.64	--	-1.5	-.2	-3.8	--
10 ft	4.85	210	3	39	27	.240	3.0	.63	-4.7	-1.5	-.3	-2.2	-1.4
20 ft	5.95	210	24	49	26	.020	2.9	.27	-2.6	-1.4	-.1	-.4	-1.6
40 ft	6.40	200	70	70	26	.007	3.0	.08	-1.5	-1.3	-.1	-.4	-1.6
60 ft	6.70	200	110	81	26	.011	2.9	.05	-1.0	-1.2	.4	-.3	-1.7
80 ft	6.65	200	118	88	26	.007	2.9	.05	-1.0	-1.2	.1	-.3	-1.6

[1]Water-quality data for samples collected on April 25, 1995, from points at increasing distances downflow in drain 2; influent is at 0 ft and effluent is at 80 ft.

[2]Saturation index [SI = log (IAP/Ksp)] calculated with WATEQF (Plummer et al. 1976) and reported measured data (above), other constituents (silica, chloride, sodium, potassium, strontium, and zinc), temperature, and Eh. A value of zero ($-0.5 < SI < +0.5$) indicates the solution is in equilibrium with the solid; values that are negative ($SI < -0.5$) and positive ($SI > 0.5$) indicate the solution potentially can dissolve or precipitate the solid, respectively.

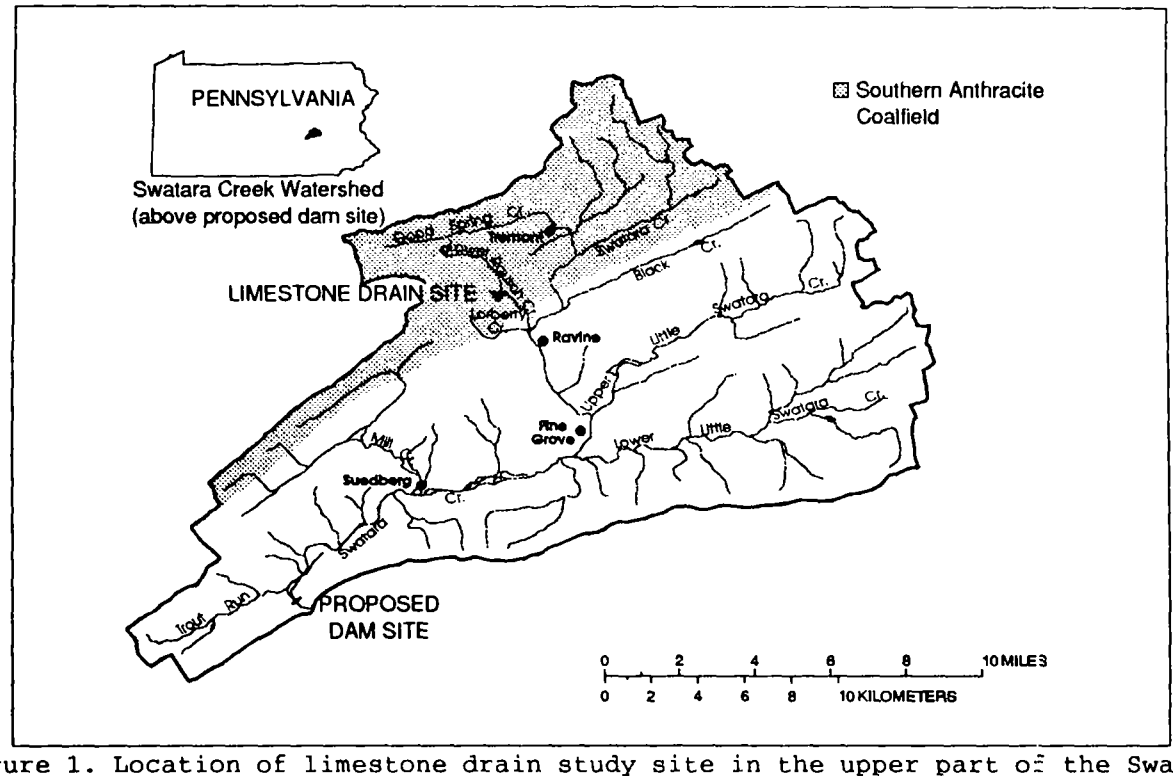


Figure 1. Location of limestone drain study site in the upper part of the Swatara Creek watershed, east-central Pennsylvania.

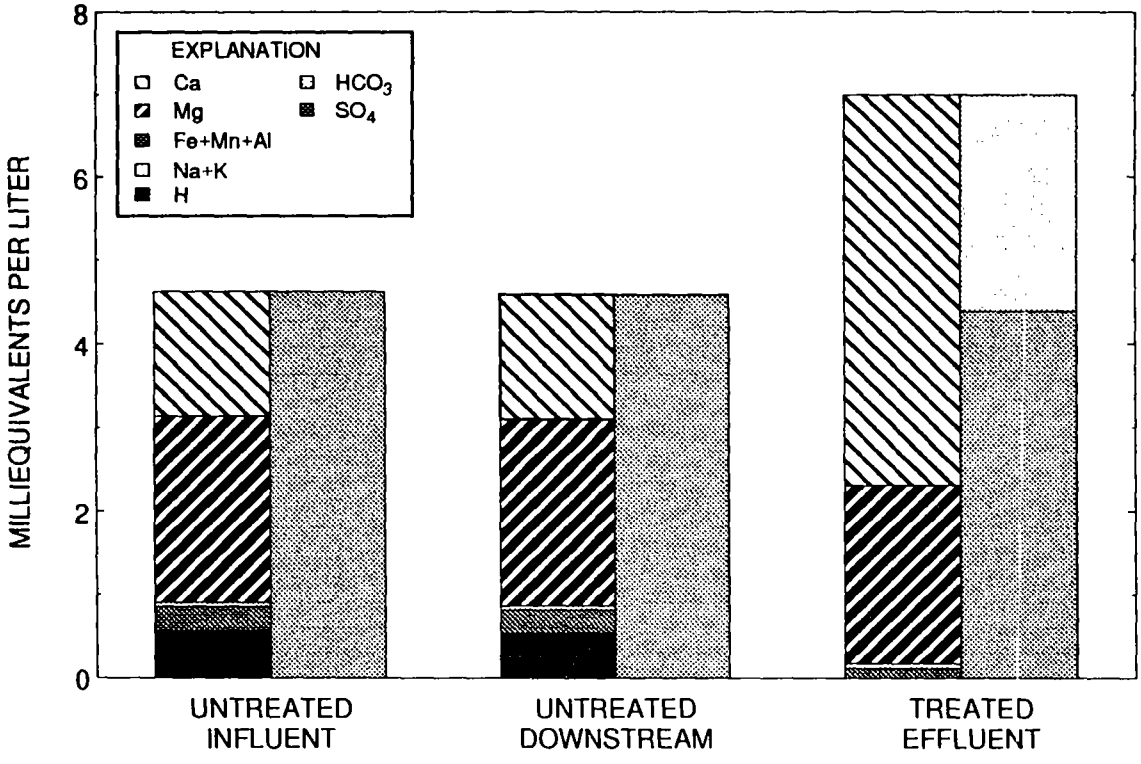
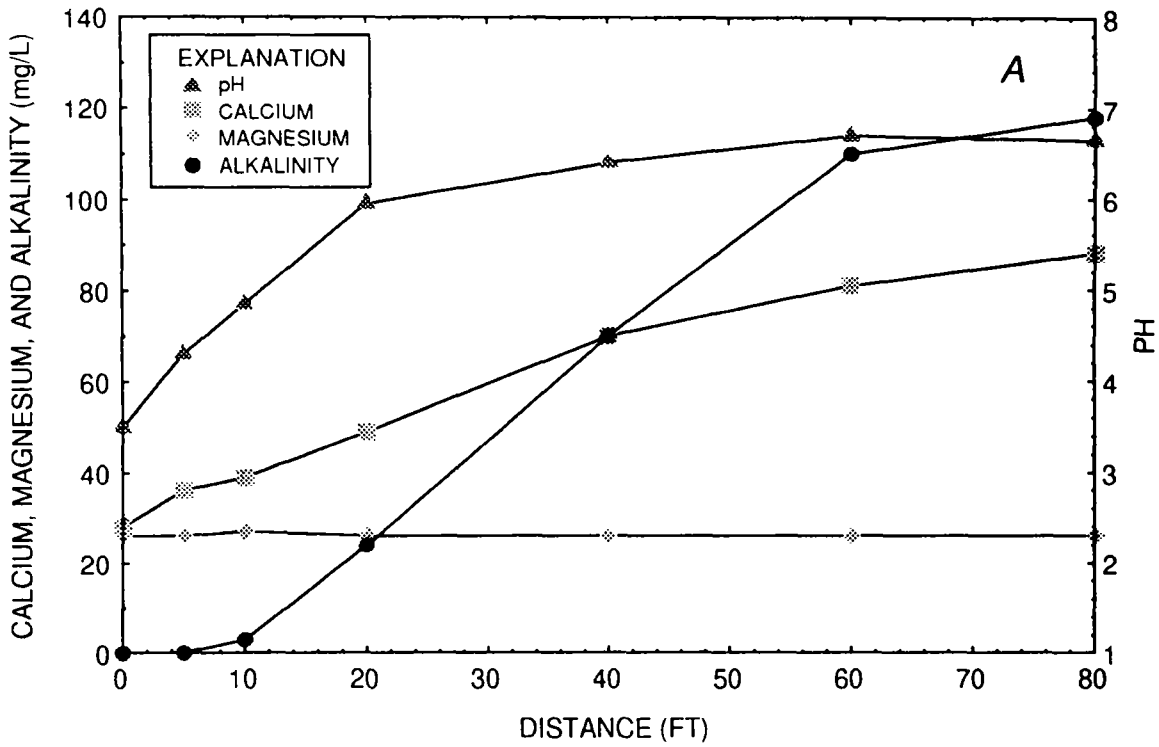


Figure 2. Major-ion composition of discharge before and after treatment by limestone drain 2. Data for samples collected March 23, 1995.



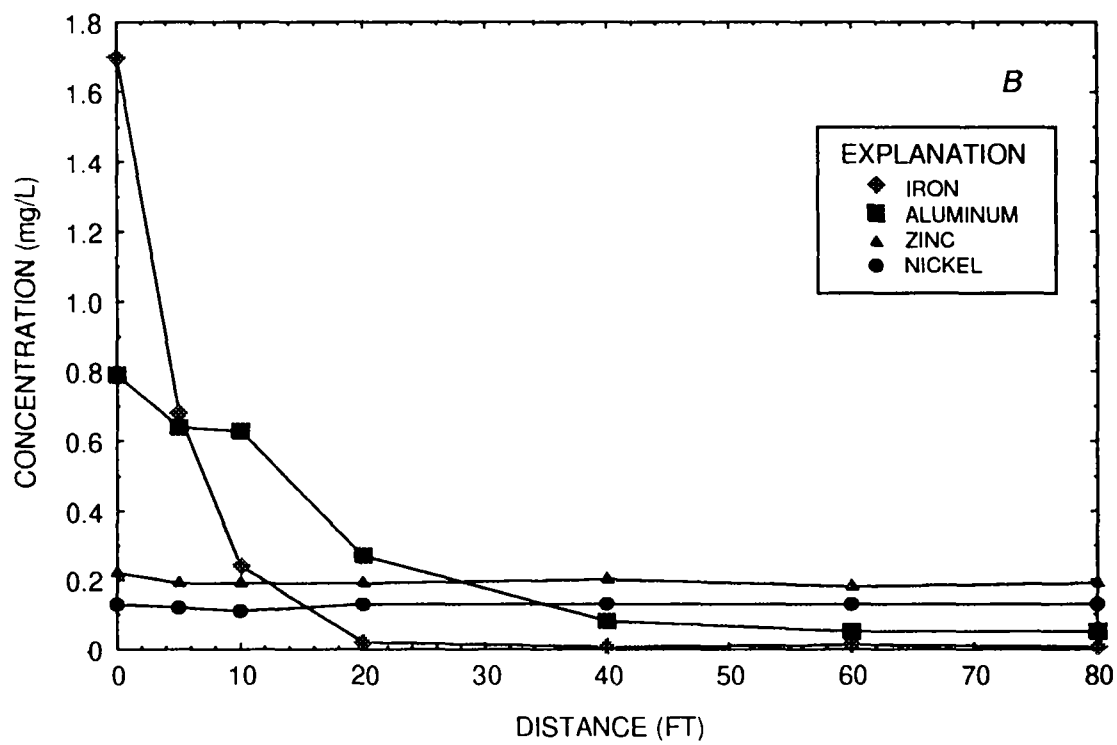


Figure 3. Sequential changes in water quality within a limestone drain: *A*, pH, and concentrations of alkalinity and dissolved calcium and magnesium; *B*, concentrations of dissolved iron, aluminum, zinc, and nickel. Data for samples collected April 25, 1995 from drain 2.

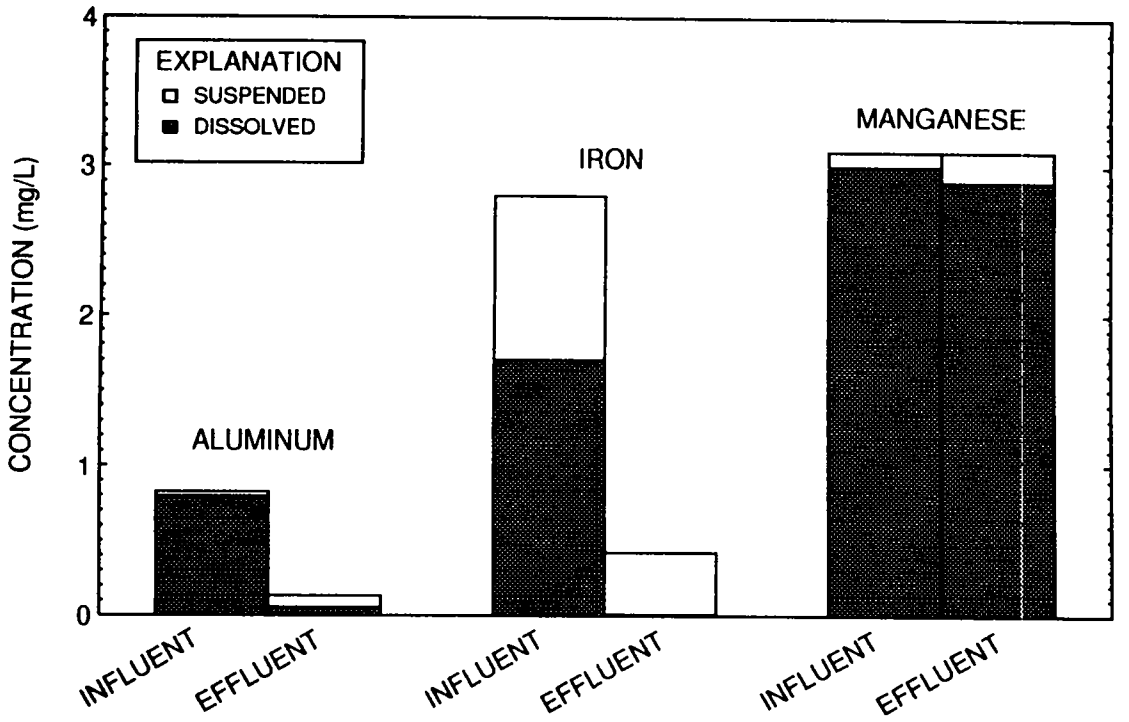


Figure 4. Concentrations of suspended and dissolved manganese, iron, and aluminum in discharge before (influent) and after (effluent) treatment by limestone drain. Data for samples collected April 25, 1995 from drain 2.

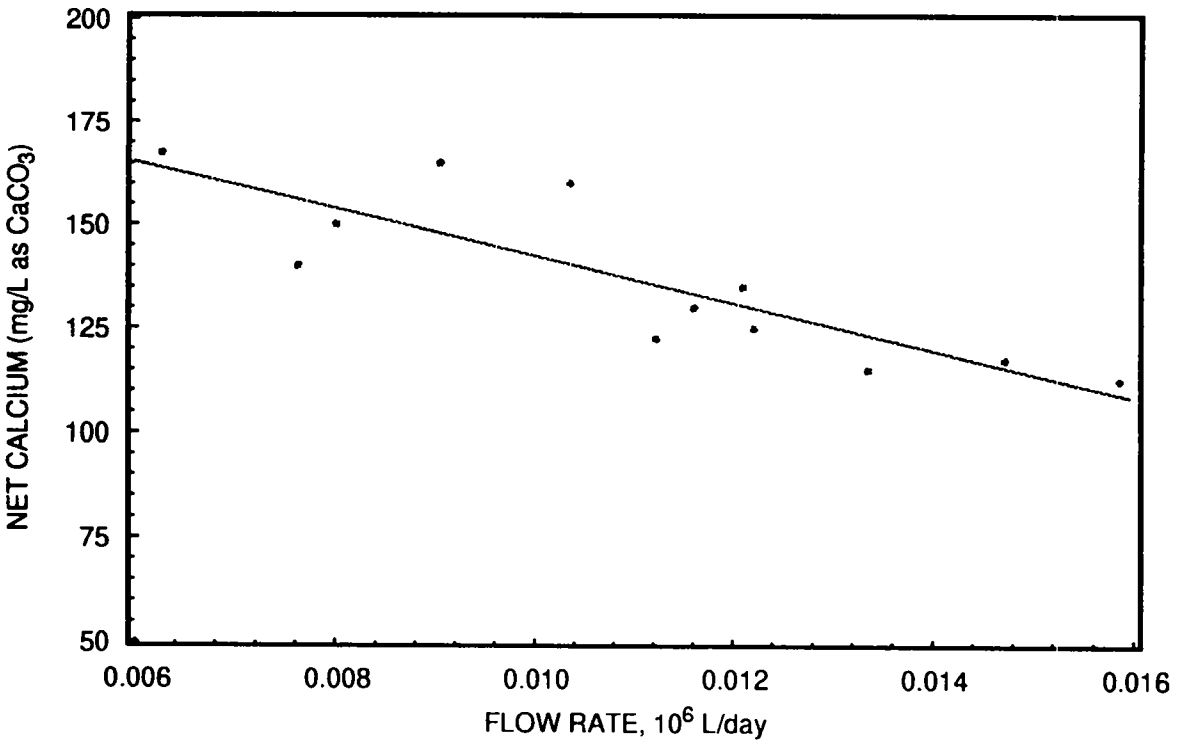


Figure 5. Concentrations of calcium added to effluent from limestone dissolution as a function of discharge rate. Data for samples collected March - August, 1995 from drains 1, 2, and 3.

CONTROLLING AND MONITORING FLOW IN TREATMENT WETLANDS USING WEIRS

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Abstract. Adjustable weirs can be used to control ponding depths and measure flows at constructed wetlands. This paper describes how to design, construct and operate weirs at wetlands.

Background

Constructed wetlands have been used successfully to treat acid mine drainage. Most constructed wetlands do not have a way to easily adjust flow into cells of a multiple cell wetland or to easily modify ponding depth in a wetland cell. In addition, accurate flow measurement to and from wetland cells is often difficult if not impossible. This paper presents practical guidance for using adjustable height V-notch (triangular) sharp crested weirs for flow control and flow measurement at wetland sites.

The paper will focus on practical tips on designing and constructing weirs including:

1. description of weirs,
2. flow equations,
3. using weirs for flow control,
4. using weirs for flow measurement,
5. sizing of weirs,
6. location of weirs,
7. materials,
8. range of adjustment,
9. preventing seepage around and under weir structure,
10. structural integrity,
11. operation of weirs.

In general, adjustable weirs are placed on the upstream end of wetlands to distribute flow between parallel cells, aerate water and measure inflow. Weirs are placed on the outlets of wetland cells to control water levels and measure outflows.

Description of Weirs

Weirs are defined as, "a notch of regular form through which water flows." (French, 1985) Outlet weirs can be thought of as the low spot in the dike around the wetland. Inlet weirs can be thought of as the end of the channel conveying water to the wetland. The top of the weir is called the "crest" and are classified according to the front view shape of the weir

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as rectangular, circular, V-notch (triangular) among others. They are also classified as to the thickness of the weir as broad, short or sharp crested.

For the combined purpose of measuring and controlling flow in constructed wetlands sharp crested, V-notch weirs are the easiest to work with and most accurate. Other types of weirs can be used and the rectangular sharp crested weir is also a suitable choice. For a weir to be considered sharp-crested, the thickness of the plate material from which the weir is made should be significantly less than the depth of water flowing over the weir.

Weir Flow Equation

Equations are available relating the flow rate over a weir to the head upstream of the weir from standard hydraulics references (French, 1985; Grant, 1989; SCS, 1956; Streeter, 1971). The equation for a triangular sharp crested weir can be given by

$$Q=\frac{8}{15}C_e\sqrt{2g}(\tan\frac{\theta}{2})h^{2.5} \qquad (1)$$

where
Q = flow, cfs (m^3/s)
C_e= discharge coefficient
g = acceleration due to gravity, ft/s^2 (m/s^2)
theta = angle of weir opening
h = effective head, ft (m)

Figure 1 defines some of the terms in equation (1). The discharge coefficient depends on the type of weir and the opening angle but is usually around 0.58 (Bos, 1976). A good angle is between 40 and 90 degrees.

The effective head is the height of the water surface above the tip of the V-notch. It should not be measured at the weir but at least several h's upstream of the weir (at least three). Do not measure h at the weir crest because the surface of the water has curved at that point and the potential energy upstream of the weir has already been converted into kinetic energy to some extent, thus violating the assumption under which equation 1 was derived.

Some other considerations regarding use of equation 1 are (see Fig 1 for definition)s:

1. P should be at least 0.3 ft (0.09 m),
2. h should be at least 0.16 ft (0.049 m),
3. T should be at least 2 ft (0.61 m),
4. h/P should be at least 1.2,
5. the depth of water downstream of the weir (tailwater elevation) should be below the vertex of the V-notch.

As one deviates from the above conditions, equation 1 becomes less accurate.

Using english units (g = 32 ft/s^2), equation (1) can be simplified to

$$Q=14.0\tan(\frac{\theta}{2})h^{2.5} \qquad (2)$$

For a 60 degree weir with a head of 4.2 in. (0.35 ft), the flow can be calculated as

$$Q = 14\tan\left(\frac{60}{2}\right) 0.35^{2.5} = 0.58 cfs$$

Using Weirs for Flow Control

If the wetland does not consist of more than one cell in parallel, then weirs cannot be used to control flow between or among cells because all of the water flowing into the wetland must go into the single cell. Raising the weir level for the influent to a single cell wetland, only can temporarily store water upstream as the upstream channel fills.

For wetlands with several parallel cells, the relative height of the weirs determines the relative flow through each cell. Lowering the level of an inlet weir relative to others will increase the fraction of the flow entering a given cell.

Weirs can be adjusted by trial-and-error to obtain the best flow distribution. Alternatively, the notch elevations for each weir can be determined mathematically by replacing the head (h) in equation 1 with

$$h_i = H_u - H_i \qquad (3)$$

where
H_u = upstream water level, ft
H_i = elevation of i-th weir notch, ft

for each of the n weirs. It is possible to set up a separate weir equation for each weir and solve the set of equations for the notch elevations that will give the desired flow distribution.

There are several alternative designs for adjustable outlet controls including telescoping valves and gates. These devices are quite expensive and are only justifiable where water level is changed frequently. For a constructed wetland, water level only needs to be changed occasionally. Therefore the authors have developed an adjustable weir as shown in Figure 2 which consists of three fixed plates (A, B and C) and one movable plate held in place by screws.

Using Weirs for Flow Measurement

Weirs are a very easy device to use for measuring flows over a wide range. The only field measurement is the head. The problem is that the head must be measured relative to the weir notch. For fixed (non-adjustable) weirs this can be done by placing a marked staff in the upstream water. For an adjustable weir the water surface elevation and the V-notch elevation (or the difference in elevation) must be known.

With acid mine drainage a staff placed in the water becomes stained and unreadable over time.

An alternative is a portable device that is attached to the weir and is placed in the water only when the flow is being measured. One such device, designed by

the authors, is shown in Figure 3. This device is attached to the weir such that the zero line on the ruler is at exactly the same elevation as the notch (bottom of V) of the weir. This can be done by installing anything that sticks out of the weir plate at the elevation of the notch.

Another device for measuring flow can be constructed from two carpenter's squares and a carpenter's level as shown in Figure 4. (US DOD, 1982)

In all measurements, the weir plate and the measuring staff must be vertical.

Sizing of Weirs

The two design parameters in weir design are the notch angle and the maximum height of the notch.

The notch angle determines the sensitivity of the weir. The more narrow the angle the greater the change in head for each change in flow. If the angle is too narrow, the head may vary more than desired in the wetland.

The maximum height is determined using the maximum flow rate expected over the weir. The height can be determined by solving equations 1 or 2 solved for h. Once the maximum height is determined the height at the minimum and average flow rates should be checked to make sure the values are not so small that h is insensitive to changes in Q.

For example, consider a weir where the maximum flow rate is 0.5 cfs, the average is 0.2 cfs and the minimum is 0.08 cfs. Equation 2 can be rearranged to give

$$h = \left(\frac{0.714Q}{\tan\left(\frac{\theta}{2}\right)}\right)^{0.4} \tag{4}$$

which yields maximum, average and minimum heads of 0.66, 0.46 and 0.32 respectively. If the heads were too small to measure accurately or below the minimum range for the weir equation, then the notch angle would need to be decreased.

The weir must not be placed in a shallow channel but in front of a pool area to insure that the distance from the bottom of the channel to the notch (P in Figure 1) is great enough to insure that the conditions described earlier with regard to P are satisfied.

The maximum range of adjustment is another design consideration for adjustable weirs. The maximum water level can be given by the value of H_A in Figure 2. The minimum level can be given as

$$H_A - h_D + h_v$$

in Figure 2. In addition

$$h_B > h_D - h_v$$

so that plate D can be slid behind plate B.

Location of Weirs

From the standpoint of measurement accuracy weirs can be placed virtually anywhere in a wetland. However, from a

water treatment standpoint, the wetland designer wants to maximize the average detention time of water molecules travelling through the wetland. This can be done by placing the inlets and outlets (weirs) at the opposite end of the wetland to achieve what process design engineers refer to as "plug flow." (Levenspiel, 1972; Viessman and Hammer, 1993) In some cases it may even be desirable to use several weirs. (Walski and Schroeder, 1978)

Ideal plug flow cannot be achieved in real wetlands due to turbulence and mixing but it can be approached by constructing wetland with a large length to width ratio. If the basin has a good length to width ratio as shown in Figure 5a, one need only insure that the inlet and outlet are on opposite sides. If on the other hand the location of inlet and outlet are constrained Figure 5b, the spur dikes (or baffles) can be introduced into the wetland as in Figure 5c to improve the length to width ratio and hence approach plug flow.

Constructed wetlands for AMD treatment have a great deal in common with dredged material containment areas and some of those references can be helpful for the hydraulics of constructed wetlands. (Shields, Thackston and Shroeder, 1987; Walski and Schroeder, 1978; US Army Corps of Engineers, 1987)

The next consideration is whether the weir should be constructed as a "drop inlet" with a pipe under the dike (embankment) or as part of the dike as shown in Figure 6. Drop inlets have the advantage of not interfering with the design and construction of the dike. However, because they are some distance away from the dike, it is more difficult to reach them to read water levels or make adjustments. On the other hand, weirs in the dike can lead to structural problems in the dike due to differential settlement. (Hammer and Blackburn, 1977)

Materials

Sharp-crested weirs should be constructed of material that is easy to work with, strong even if it is thin, durable and inexpensive. Galvanized sheet metal was selected by the authors. Wood could also be used but it would need to be fairly strong to resist the forces on it. For example plywood would need to be so thick, it might no longer be considered a sharp crested weir at low flow rates.

Where flow measurement is not a consideration and water depth need not be controlled exactly, weirs can be constructed of boards placed vertically inside a metal frame. (US Army Corps of Engineers, 1987)

Preventing Seepage

Seepage under and around weirs can be a serious problem in accurately measuring flow. By using very few parts, seepage through the weir can be minimized.

Seepage under and around the weir can be reduced by

grouting any joints between the weir and the dike or floor or the weir. Clay can be packed along the bottom of the weir to prevent seepage under the weir.

Structural Integrity

Weirs act as part of the dike and can be the weakest portion of the dike around the wetland. (Hammer and Blackburn, 1977) When the foundation material under the dike is a weak soil, the dike can settle faster than the weir structure. This can open up seepage paths between the weir and the rest of the dike. In addition, the material adjacent to the weir needs to be compacted well when it is placed so as not to allow seepage between the weir and the dike to start.

In the authors' weir, the weir plate was extended beyond the side wall into the dike to make it virtually impossible for water to seep around the weir as shown in Figure 7.

The weir acts as a dam and is therefore subject to significant forces due to the hydrostatic pressure of water upstream of the dam. The maximum force on a weir acts at the centroid and can be determined based on the maximum height of water behind the weir and the submerged area of the weir plate according to

$$F=\gamma h_c A \qquad (5)$$

where

F = force on weir, lbs
h_c= depth at centroid of weir, ft
A = cross sectional area, sq ft
gamma = specific weight of water, 62.4 lb/cu ft

The centroid is usually two thirds of the way down the weir. For a two foot wide three foot high weir, the force is

$$F=62.4(2)(2x3)=750\,lb$$

From the above it is clear that even a small weir has significant forces acting on it and must be well supported from the downstream side or else it can fail. The support can come from earth, masonry block or poured-in-place concrete.

Water flowing over the weir must not be allowed to erode or scour the channel leaving the weir and hence weaken the weir structure.

In cold climates where frost can penetrate to a significant depth, the weir structure should be placed on a deep footer to reduce the possible damage due to frost heaving.

Operation

Water level in constructed wetlands should not need to be adjusted frequently. Once the water level is set and flow distribution between cells is fixed they should stay roughly the same over time. However, adjustable weirs provide the

opportunity to experiment with relative ease with alternative flow distributions and ponding depths while knowing the flow and loadings in the wetland.

Adjustable weirs also provide the ability to shut off flow to a cell for a period to perform maintenance and study the cell in some detail. The ability to raise the weir level also provides the opportunity to raise the ponding elevation in the wetland to maintain the same depth even though the wetland begins to fill with sediment.

Summary

Properly designed and constructed adjustable weirs can be an effective way to control and measure flow in constructed wetlands.

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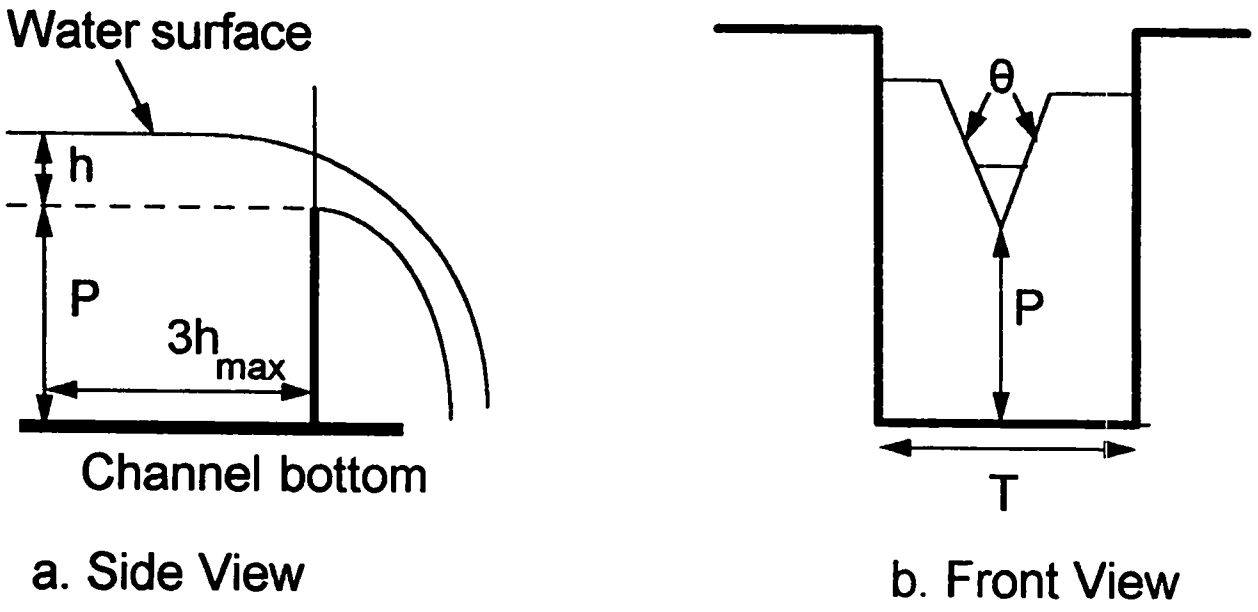

Figure 1. Definition Sketch for V-notch Weir

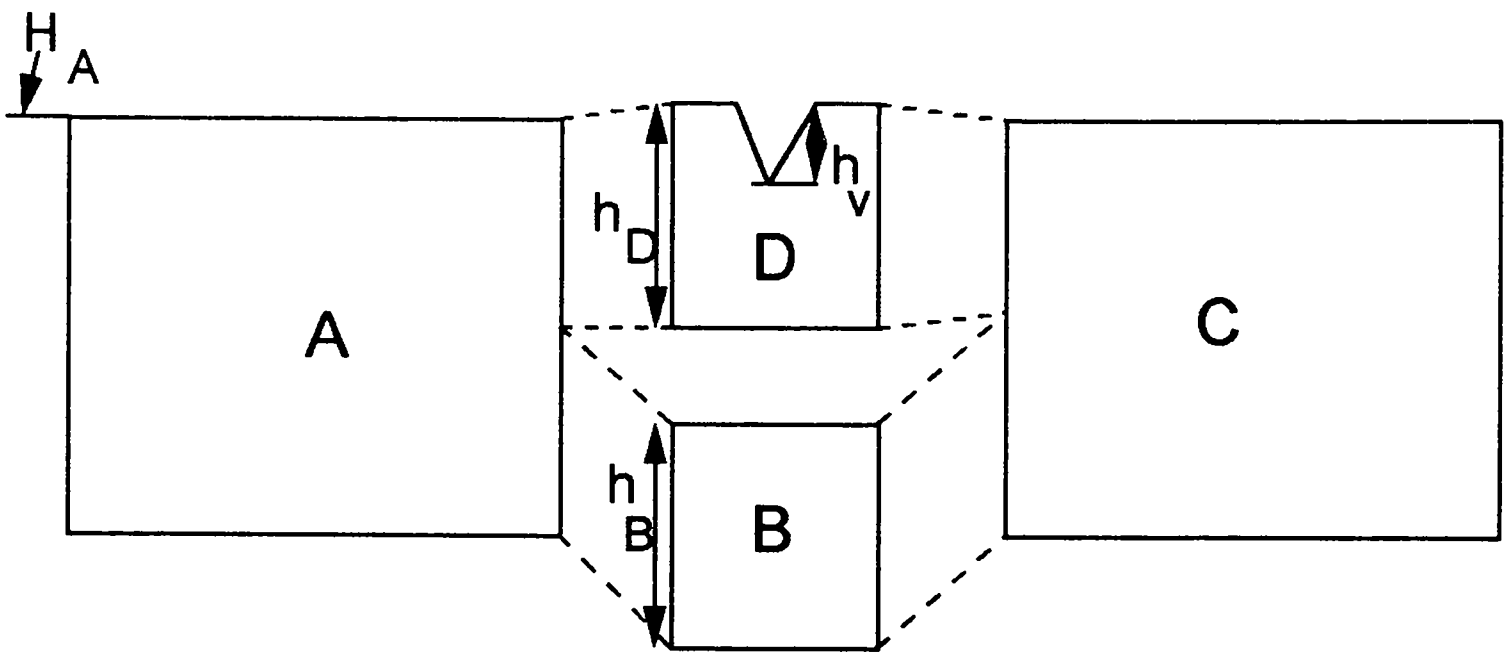

Figure 2. Plates Used in Weir Construction

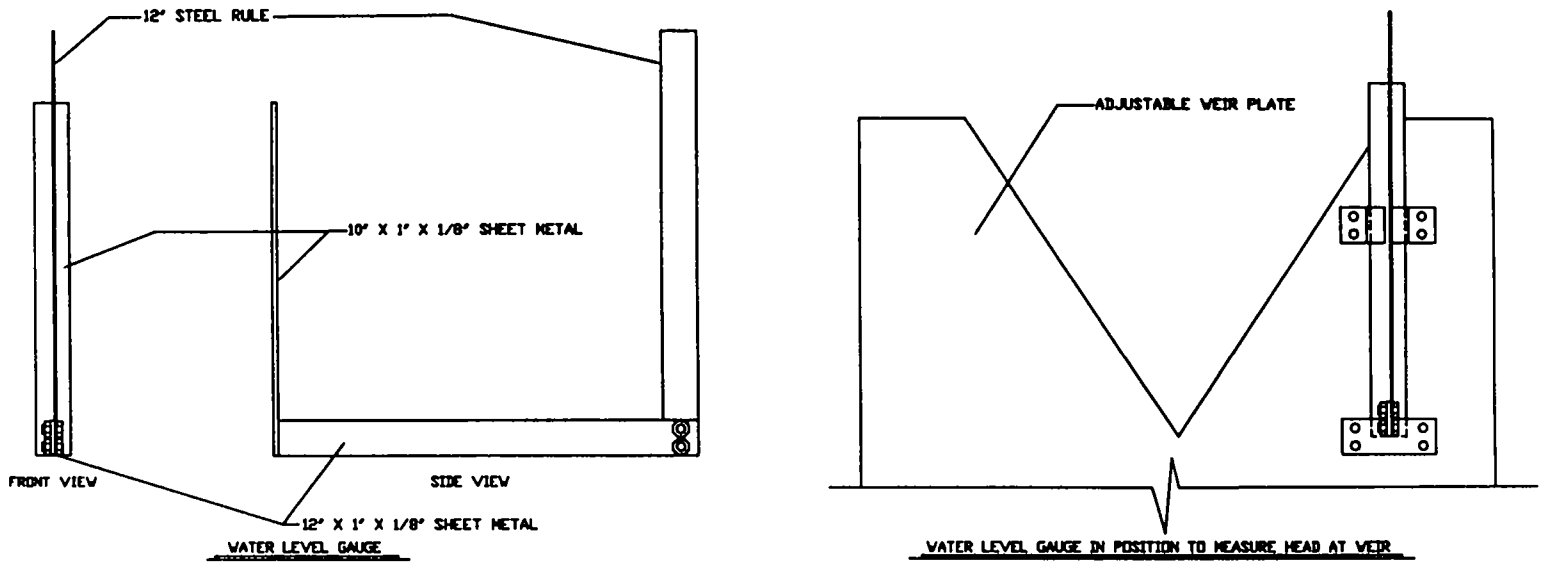

Figure 3. Depth Measuring Device

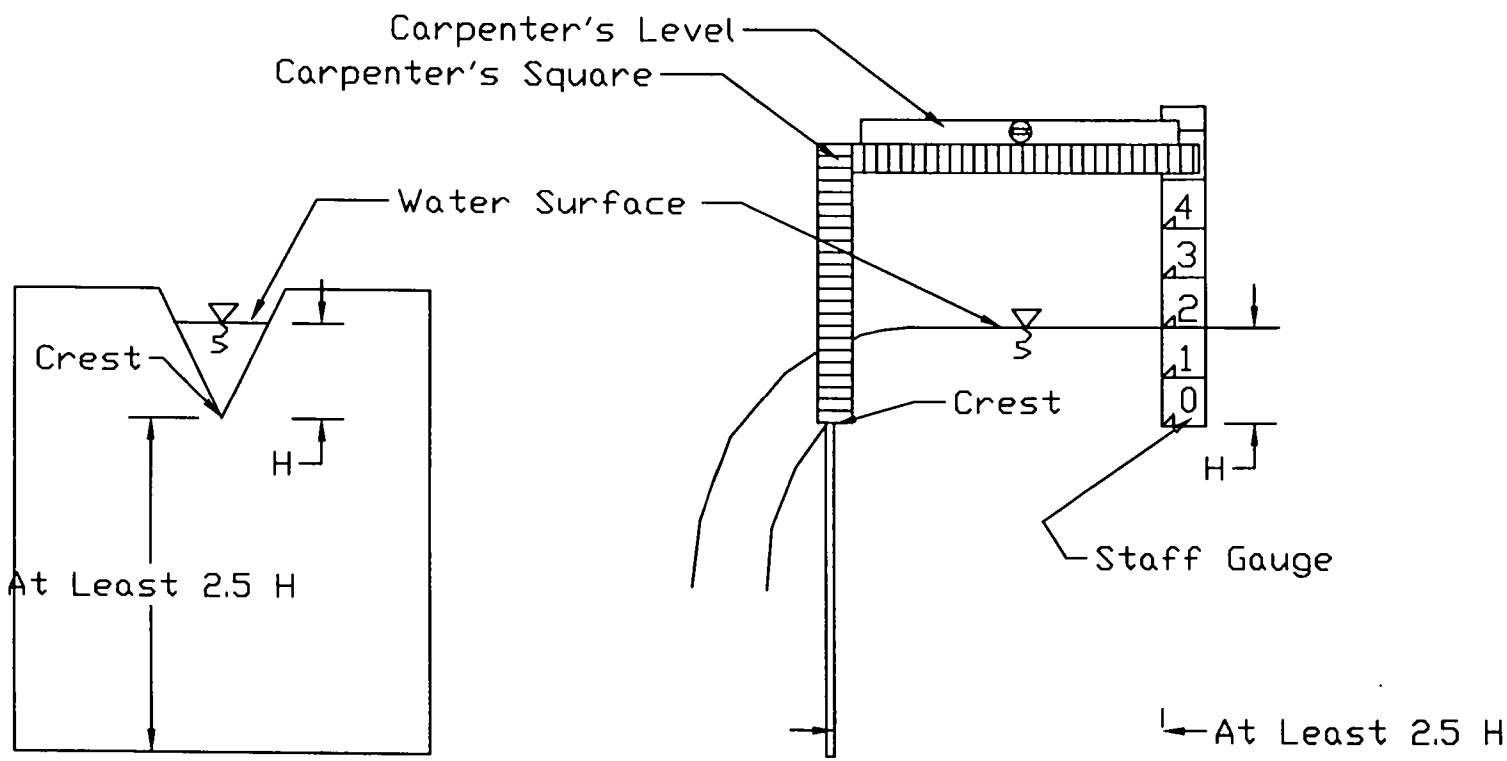

Figure 4. Alternative Method to Measusre Depth

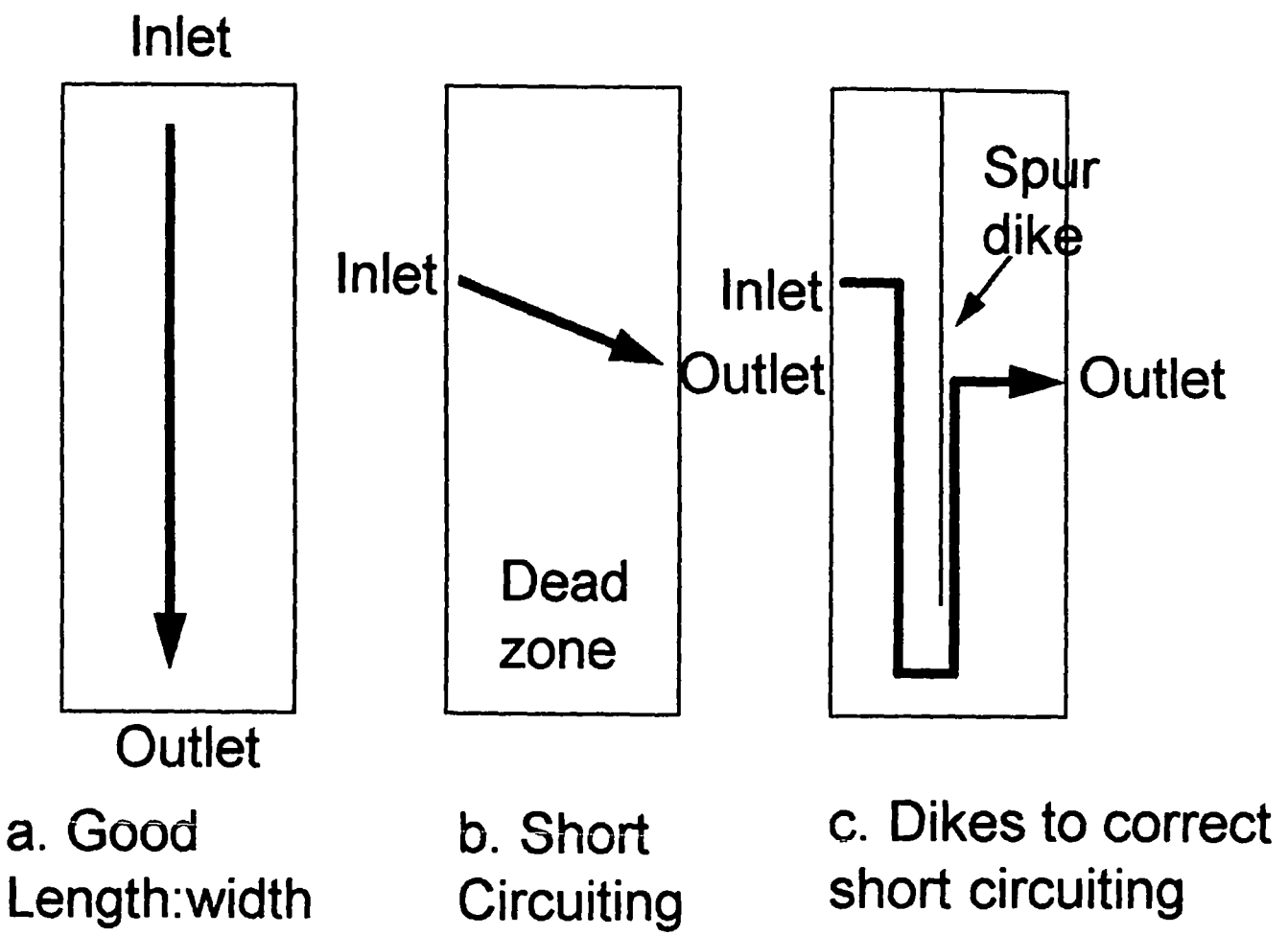

Figure 5. Alternative Flow Paths through Wetland

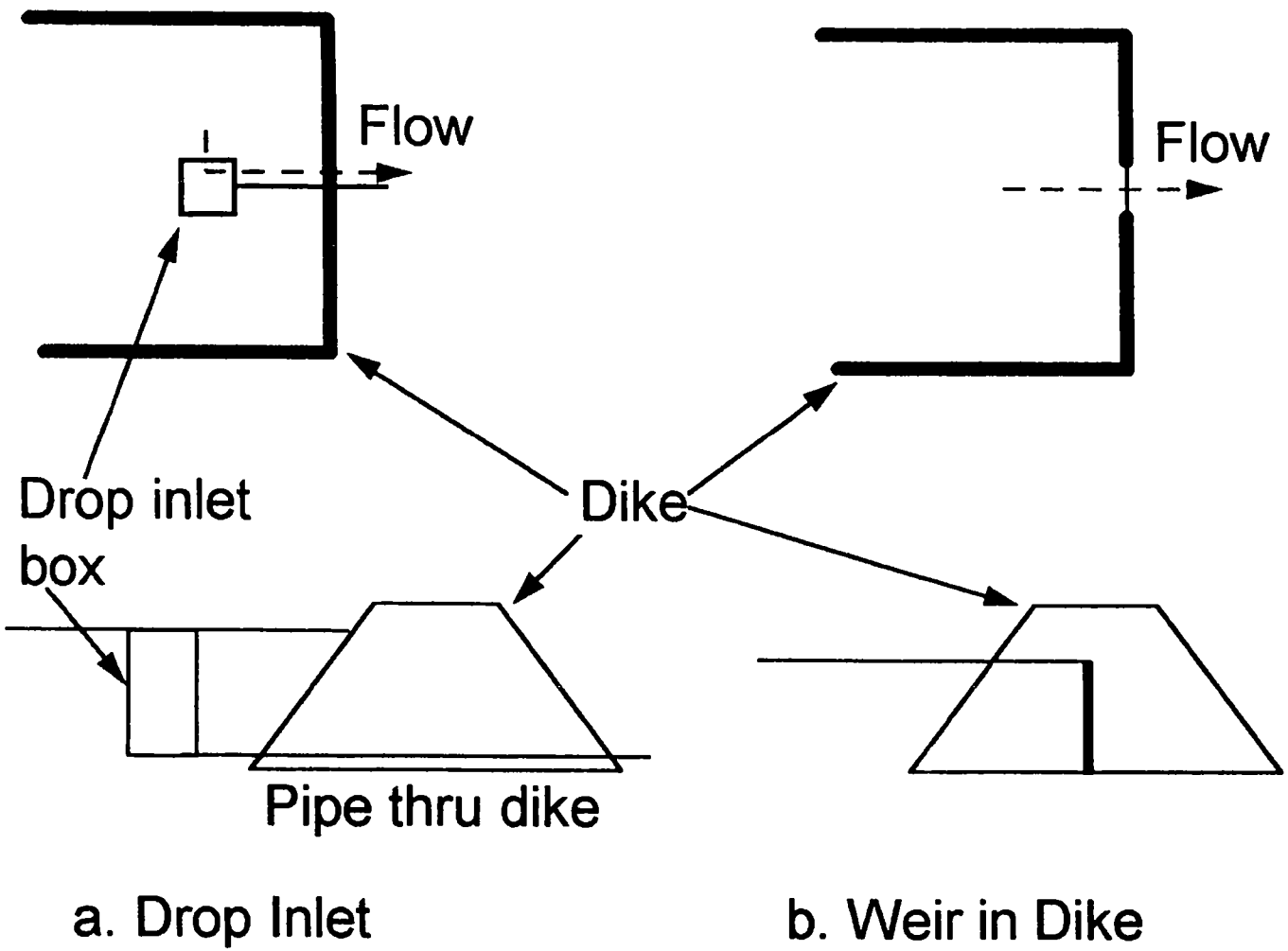

Figure 6. Alternative Location of Weir Structure

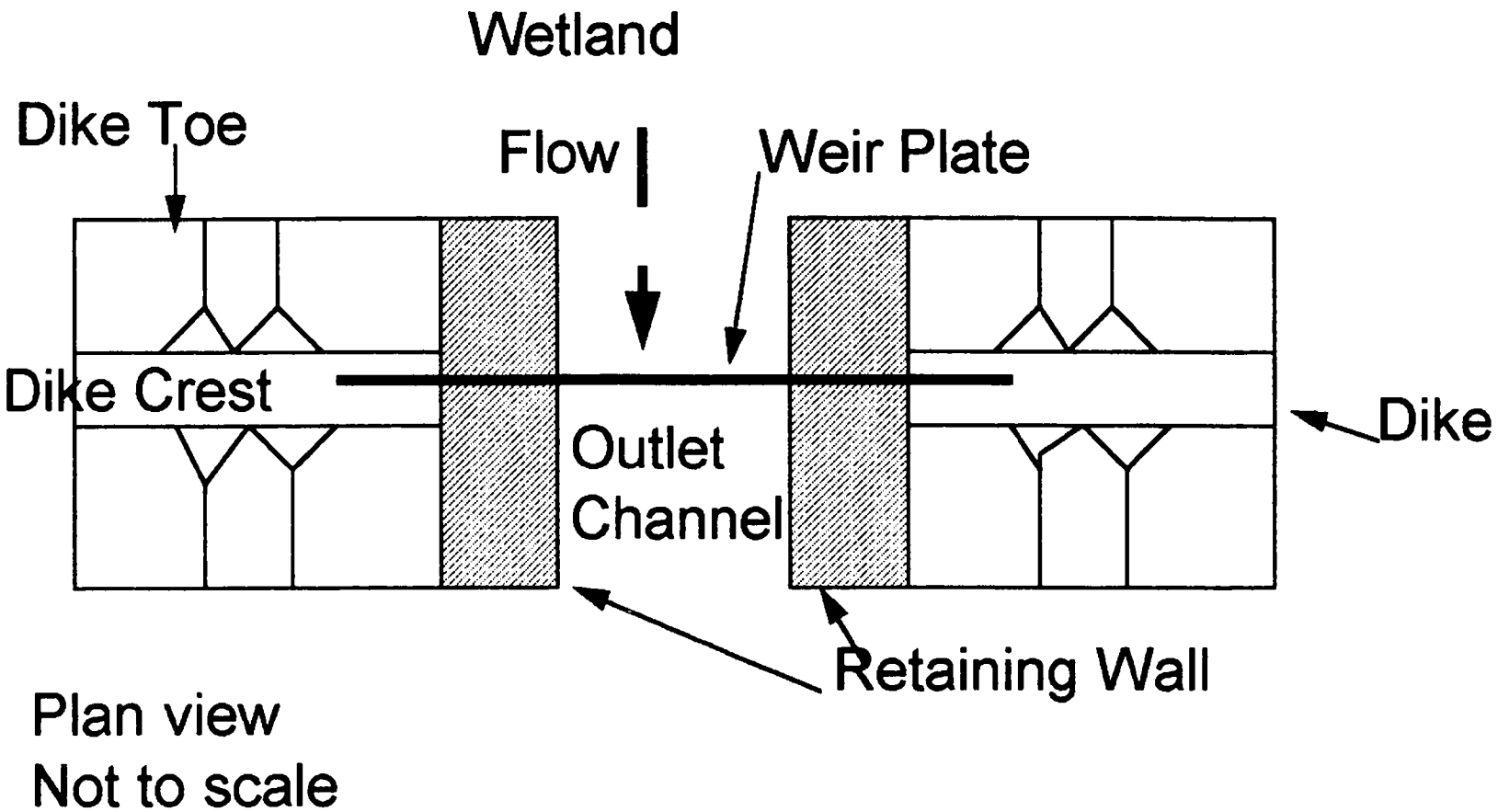

Figure 7. Plan View of Weir Structure

Sulfur Acidic Lakes in Germany: What Has to be Done?
Helmut Klapper, Walter Geller and Martin Schultze. UFZ Centre for Environmental Research Leipzig-Halle Department of Inland Water Research, Magdeburg, Germany. The problem: The rapid decline of brown coal mining, to about 30% in the new "*Länder*" of Germany, has produced many new lakes within a few years. These lakes filled with groundwater very often are affected by acidification from pyrite oxidation. Their water is dark red, the pH 2 to 4 with the consequence of high metal content and absence of bicarbonate. Depending on the base binding capacity of the oxidised rock layers, the acidic conditions may last for decades, and, therefore, countermeasures are needed. Decision criteria for an active water quality management are (1) the urgent demand for fish-waters and recreational areas, (2) the need of new jobs in the post-mining landscape, and (3) the desirability to conserve some of these limnologically unique lakes as objects for science. In nature are also examples of sustainable bacteriological sulfate reduction. One case study was performed on the meromictic lake *Waldsee*. The aerobic epilimnion is acidic, the anaerobic monimolimnion, however, is neutral. The preferred microhabitat is the boundary layer between the two main strata, where plankton synthesise the organic material, necessary for sulfate-reducing bacteria. In another case the carbon source stems from urban sewage. The 11km-long Lake *Laubusch*, serving as a drainage for sewage water, reduces sulfate in the heavy loaded hypolimnion and in the macrophyte stands. The good fish stock of this lake is the basis for sport fishing. The mining lake *Senftenberger See* is an example of neutralisation by river water. After 1.5 exchanges of the lake volume the pH shifted from acidic to neutral in the manner of a titration curve. Experience with anaerobic technology was gained during our development of an in-lake denitrification plant. In a reservoir with nitrate content above drinking water standard, a straw-filled steel cage 20 x 60 x 1.50 m was positioned on the bottom. Nitrate-rich water together with a carbon source was pumped through this *aufwuchs* reactor. After eight weeks the whole hypolimnion was free of dissolved oxygen and free of nitrate because of nitrate respiration. For the task at issue here, the long-term biological desulfurication in mining lakes,

research is likewise focused on the hypolimnion and the sediments as preferable places for sustainable treatment.

Conclusions

- Large lakes threatened with acidification should preferably be filled with surface water.
- In smaller and deeper lakes, conditions suitable for sulfate-reducing bacteria may be created by adding or producing degradable carbon sources.
- In too-shallow lakes, the stratification must be stabilised by installing barriers against wind-induced mixing.
- Besides this final in-lake treatment, all possible means of abating acidification at the sources should be implemented where applicable.

MANGANESE REMOVAL IN SATURATED GRAVEL BEDS: OBTAINING DESIGN CRITERIA.

F.J. Sikora, G.A. Brodie, and L.L. Behrends

Abstract: Manganese is difficult to remove in passive wetland systems due to high pH requirements for rapid Mn oxidation and solubilization of MnO_2 in the presence of Fe^{2+}. To achieve Mn removal below the federal requirement of 2 mg/L, gravel beds placed after wetlands that remove Fe has been proposed. Although some data is available on gravel beds, not enough is available to recommend design parameters for these systems under a range of operating conditions. A study is being conducted to obtain information on the required Mn loading rates and retention times for adequate Mn removal at the TVA constructed wetland research facility. Treatments consist of two Mn loading rates of 1.4 and 3.7 $g/m^2/d$ and two types of gravel, namely limestone and river gravel. The treatments are replicated 3 times resulting in 12 experimental units. After 3 months of operation, limestone is proving to be more effective at removing Mn than river gravel. Compilation of operating parameters for Mn removal in gravel beds will benefit power utility and surface mining industries by yielding information on a passive system that is inexpensive relative to caustic alkaline drip.

Additional Key Words: iron, wetlands

Acidic Stream Mitigation by Limestone Sand Addition. David L. Brant[1], Anthony J. Marich Jr.[2], Keith L. Largent[3] (1-WVU National Research Center for Coal and Energy, Morgantown, WV, 2-PA Department of Environmental Protection, Markleton, PA, 3-Somerset Conservation District, Somerset, PA).

Abstract: The Town Line Run watershed comprises an area of 3,600 wooded acres. The tributaries feeding the stream consist of sandstone springs that do not contribute alkalinity to the watershed, leaving the stream susceptible to acid precipitation. This has a negative affect on Iser's Run, a native brook trout fishery above the confluence with Town Line Run. The objective in stream liming is to improve water chemistry by increasing pH, alkalinity, and reducing acidity, aluminum, and iron. Introducing crushed limestone directly into a stream from a dump truck is an inexpensive but temporary solution to accomplish this objective. In this type of liming operation, a bed of limestone is spread down the stream channel by the momentum of the stream from the introduction point, rather than manually. Water moving across this bed dissolves the limestone, increasing the pH, alkalinity, and calcium while decreasing the acidity, iron, and aluminum concentrations of the water. The size of the limestone particles is important for this purpose because particles that are too small (<150 microns) will carried away, while particles that are too large (>1000 microns) will remain at the introduction point. Our study placed 80 tons of sand-sized limestone (85% calcite) in the stream channel at a single point. Water samples were collected monthly at the following sites (1) directly upstream of the addition site, (2) 100 yards downstream of the site, and (3) 2500 yards downstream of the site. Other sample locations include (4) upstream and (5) downstream of the Town Line Run-Iser's Run confluence and the Casselman River upstream (6) and downstream (7) of Town Line Run. The samples were analyzed for pH, Specific conductivity, Alkalinity, Acidity, Iron, Manganese, Aluminum, and Sulfate. The first liming produced a pH of 6.6 that slowly declined to the baseline pH of 4.8 after 10 months. The alkalinity increased to 10 mg/l $CaCO_3$ eq. and also decreased to 6.0 mg/l $CaCO_3$ eq. after 10 months. The acidity was eliminated after the addition and gradually increased to 12.6 mg/l $CaCO_3$ eq. in the same time period. The second addition of 180 tons of the same limestone yielded similar initial results. The pH increased to 6.5, the alkalinity increased to 18.6 mg/l $CaCO_3$ eq., and the acidity was eliminated. In general, this low cost (<$2,000.00) project demonstrated that a single point application of limestone can increase the water quality of marginal acid sensitive streams for a limited time period. Careful selection of limestone particle size and amount of limestone can reduce the addition interval to annual or bi-annual additions.

Additional Key Words: Acid Rain, Lime-sand.

Performance of a Wetland/Anoxic Limestone Drain Treatment System at the Douglas Abandoned Mine Land Project. P.L. Sterner, J.B. Cliff, J.G. Skousen and A.J. Sexstone

Abstract. Acid Mine Drainage (AMD) exiting West Virginia's Abandoned Mine Land (AML) sites often is remediated using passive treatment systems. A new passive treatment technology which employs a combination wetland/anoxic limestone drain (WALD) is currently being evaluated at one AML site near Thomas, WV. The WALD utilizes a three cell design to complete treatment. Cell 1 (dimensions 365.8m x 2.5m x 1.8m- L x W x D) employs a 0.6m base of limestone overlain by 1.2m of organic material. Cell 2 (dimensions 457.2m x 9.1m x 2.4m) employs a 1.5m limestone base overlain by 0.9m of organic material. Cell 3 (dimensions 25m x 9.1m x 1.2m) is a sedimentation basin which allows deposition of precipitating metals. Thirteen polyvinyl chloride (PVC) cylinders with multiple ports were installed throughout the system so that water sampling at various locations and depths can be conducted. The working hypotheses to be tested is whether microbial iron reduction in organic sediments is sufficient to prevent limestone coating by iron oxyhydroxides, thus allowing continuous limestone dissolution and subsequent pH increase of the treated AMD. Influent water enters the system via surface flow at a rate of approximately 240 gpm from pipes draining an underground coal mine. Average influent AMD parameters over the study period were: pH= 3.0, total acidity= 426 mg/L, total alkalinity= 0 mg/L, total iron= 20 mg/L, total manganese= 6 mg/L and aluminum= 31 mg/L. Results based on effluent water quality indicate that AMD is being treated by the WALD. Average water quality parameters exiting the WALD over the course of the study period were: pH= 6.6, total acidity= 25 mg/L, total alkalinity= 152 mg/L, total iron= 0.1 mg/L, total manganese= 1.5 mg/L, and aluminum= 0.8 mg/L. Despite dramatic improvement in effluent water quality parameters, the organic matter in cells 1 and 2 does not currently function as desired. The organic matter has limited hydraulic conductivity so less AMD infiltration takes place than is desirable, resulting in surface rather than subsurface flow. Therefore minimal treatment occurs because of limited contact between AMD and the WALD. Dissolved oxygen values within the organic matter are not conducive to Fe^{3+} reduction, therefore Fe^{3+} is likely precipitating. Limestone is dissolving sufficiently to raise pH of the AMD allowing hydrolysis and precipitation of Fe^{3+} and Al^{3+} as shown by absence of total iron and aluminum and increased Ca^{2+} in the effluent water. Considering these results, the WALD treatment system may prove to be an effective retainer of metals that contaminate mine waters. However, retention of metals coats limestone, inhibiting dissolution, which leads to decreased functionality of the WALD. The pH of the effluent water has declined since AMD was introduced, accompanied by an increase in total acidity.

Additional Key Words: acid mine drainage and wetland/anoxic limestone drain.

Advanced Design and Construction Techniques for Staged Passive Wetland Treatment Systems for Coal Ash Leachate

by
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ABSTRACT. Passive wetland treatment systems represent a viable alternative for the chemical treatment of leachate waste waters from coal ash disposal. Ash resulting from the burning of coal commonly contains many trace metals. Leachate from active and closed disposal areas is often regulated as a solid waste discharge requiring the removal of trace elements to meet water quality standards in the receiving stream. For many of these sites, topographic constraints limit the availability of land for large passive systems. To meet these conditions, the Springdale Passive Wetland Treatment System represents a more engineered configuration with advanced design and construction techniques, with specific design considerations for future maintenance access. The Springdale facility is a passive wetland system designed to treat leachate from a closed fly ash disposal site near Pittsburgh, Pennsylvania. The project is jointly funded by the Electric Power Research Institute and Allegheny Power Service Corp. The project was developed to provide initial compliance with iron, pH and TSS under a current NPDES permit for a toe of fill discharge. To meet pending discharge criteria, the system was designed as a multiple stage system for research and development of passive design criteria for the removal of trace metals. Flows average 45 gpm, and the discharge is characterized by circumneutral pH, elevated iron, manganese, boron, and other trace metals typical of fly ash leachate. The leachate is collected at the toe and pumped to a flow equalization basin for initial iron oxidation and precipitation prior to entering a series of eight passive treatment units, each approximately 30 feet in width by 150 feet in length. The first four units are vegetated aerobic wetlands to remove residual iron. The fifth and sixth units are two rock drains for bacterial manganese removal. The seventh unit is a sulfide-generating organic upflow bed for trace element removal. System discharge occurs through a final algal and sand filter unit. Construction was completed in October 1995.

Performance of the system is monitored by water quality sample points located at the discharge of each type of passive treatment, so as to identify the most effective removal environment for each target parameter. The system is being monitored for 38 water quality parameters of interest on a semi-annual basis and monthly for selected parameters relating to current compliance objectives. Average removal efficiencies for Aluminum, Total Iron, Dissolved Iron and Manganese have been 99, 92, 96, and 66 percent respectively since the system was completed in 1995. A comprehensive monitoring program for water quality and substrate analyses will continue for at least 24 months. Detailed plant uptake and metal cycling studies will be undertaken with the University of California, Berkeley, through 1998. The ultimate goal of this work is to provide improved sizing and design standards for passive removal of trace metals.

Passive Treatment of Acid Mine Drainage in Systems Containing Compost and Limestone: Laboratory and Field Results. G. R. Watzlaf and D. M. Pappas.

Abstract: Passive, down-flow systems, consisting of compost and limestone layers, termed successive alkalinity producing systems (SAPS), may be well suited for treatment of mine drainage containing ferric iron and/or aluminum. A column, simulating a SAPS, has been operated in the laboratory for 52 weeks. The 0.16-m diameter column consisted of a 0.30-m thick layer of limestone, a 0.76-m thick layer of spent mushroom compost and 0.91 m of free standing water. Actual AMD (pH = 3.02, acidity = 218 mg/L (as $CaCO_3$), SO_4 = 600 mg/L, Fe = 16.0 mg/L, Mn = 12.1 mg/L, and Al = 17.1 mg/L) was applied to the column at a rate of 3.8 mL/min. Effluent pH has remained above 6.2 (6.2-7.9) in the column system. A SAPS located in Jefferson County, PA has been monitored for the past 4.5 years. The SAPS has an approximate area of 1000 m^2 and contains a 0.4-m thick layer of limestone, a 0.2-m thick layer of spent mushroom compost, and 1.5 m of free standing water. Mine water (acidity = 335 mg/L (as $CaCO_3$), SO_4 = 1270 mg/L, Fe = 246 mg/L, Mn = 38.4 mg/L, and Al = <0.2 mg/L) flowed into the SAPS at a rate of 140 L/min. Water samples from the field and laboratory systems have been collected at strategic locations on a regular basis and analyzed for pH, alkalinity, acidity, Fe^{2+}, total Fe, Mn, Al, SO_4, Ca, Mg, Na, Co, Ni, and Zn. Alkalinity has been generated in both field and laboratory systems by a combination of limestone dissolution and sulfate reduction. The column generated an average of 378 mg/L of alkalinity; 74% due to limestone dissolution and 26% due to bacterial reduction of sulfate. The field SAPS generated an average of 231 mg/L of alkalinity and exhibited seasonal trends. Sulfate reduction was responsible for 70% of the alkalinity production in the summer and decreased to as little as 30% of the alkalinity production in the winter.

Additional Key Words: compost wetlands, sulfate reduction.

EVALUATION OF SELECTED GRASSES FOR THE REVEGETATION OF A COAL SLURRY LAGOON IN WESTERN KENTUCKY BY DIRECT SEEDING.

by

T.P. Carter, R.I. Barnhisel, B. Gray and J.R. Nawrot

The research was conducted on the intermediate zone of a 23 ha slurry impoundment located on the Gibraltar Mine, near Central City, Kentucky. Two separate experiments were established. In the first, cultivation was carried out using a conventional disc and in the second, an 'Aer-way' attachment was used. The experimental plots, each measuring 4 m x 3 m, were prepared and seeded in June 1993. Percentage cover results were estimated by field observation in September 1995. In the experiment where cultivation was carried out using a conventional disc,. the greatest mean percentage cover levels with sideoats grama 'El Reno' (55%) being the best entry. For the second experiment, the highest percentage cover was obtained for switchgrass 'Alamo' (38%). These values reflect do not include weeds which averaged in the 40-50% ranged and most plots had 90-95% vegetation cover. Results for the cool-season species were generally disappointing and may have been due to late seeding-date. This study revealed that warm-season grasses, especially the tall-grass prairie species, produced reasonable stands on coal slurry after just two years.

KEY WORDS: direct seeding, coal slurry lagoon, warm season grasses, grass seed stratification

Constructed Wetland Treatment System for Upper Blackfoot Mining Complex. F. S. Sanders

Abstract: A combined passive oxidation/sedimentation (pre-treatment) and constructed wetland (polishing) system has been designed and installed to treat mine drainage at the inactive Upper Blackfoot River Mining Complex northeast of Lincoln, MT. The system is designed to treat up to 100 gpm flow with moderate heavy metal concentrations (25 to 95 mg/L Zn; lower concentrations of Pb, Cu, and As), moderate Fe concentrations (< 100 mg/L), and periodic high acidity (pH 2.6 to 3.5). The treatment scheme is flexible to accommodate a wide range of flows, metal concentrations, and acidity in order to gain cost-efficiencies and to meet restrictive discharge standards for the environmentally-sensitive Blackfoot River watershed. The wetland treatment system presently is undergoing start-up testing and initial tuning and will be operational during summer, 1996. Conceptual and engineering designs are presented along with preliminary data.

Additional Key Words: constructed wetland, acid mine drainage, metals

Reclamation of degraded areas in Eastern Amazonian: the potential of *Sclerolobium paniculatum* Vogel

by

Silvio Brienza Junior[1]
Luiz Eduardo Dias[2]
Cassio Alves Pereira[3]

Abstract. *Sclerolobium paniculatum* Vogel (taxi-branco) is a legumenous tree native to the Brazilian Amazon region. It occurs in different types of soil and fix atmospheric nitrogen. The mechanical dormancy of the seeds may be overcome by immersing in boiling water and then removing them from the heat until the water cools to room temperature. The seed germination occurs in approximately 30 days. In greenhouse conditions, taxi-branco does not respond to the application of Ca and S. The critical levels in the soil of these two nutrients were 0.37 meq/100 cm^3 and 5.10 mg/cm^3, respectively. The silvicultural performance of taxi-branco may be considered satisfactory when compared to other native tree species of the Amazon. In homogenous plantations, taxi-branco trees produce about eight tons of litter per hectare. Its rapid growth accompanied by a high production of litter and its N fixation qualify this species as potentially suitable for the recuperation of degraded soils by human actions.

Additional Key Words: Amazon region, nutrients, critical levels, legume tree, etc

Introduction

The economic activities developed in the Brazilian Amazonian have caused different levels of degradation. From an environmental point of view, exist effects on biodiversity involving the losses or damages to the animal and/or plant populations, as well as alterations in the critical functions of the natural ecosystems, modifications in the carbon stores, quantities of water transpired and the retention of nutrients. Estimates shows 25 to 34 million hectares of native forest, have already been altered. (INPE, 1990; Skole & Tucker, 1993 and Feanrside, 1993).

The search of environmental conservation practices must be based upon more sustainable production systems, involving biological, social and economic aspects. Restoring the productivity of areas considered altered or degraded, must be the object of continuing research. The use of species more adapted and production systems more efficiently in the humid tropical areas are the key for sustainability.

Reforestation programs in the Brazilian Amazonian are limited to a few regions such of Para and Amapa States. The mainly species used in plantations programs are non-native, e.g. *Pinus caribea*, *Eucalyptus urophylla*, *E. deglupta*, *E. pellita* and *Gmelina arborea*. Despite the high production of the introduced species, the potential of native species has not been exploited and it is rarely considered in reforestation programs.

This paper presents and discusses a mineral nutrition experiments for seedlings of taxi-branco carried out in glass house and silvicultural field information obtained in the State of Para in Brazilian Amazonian.

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Material and Methods

- Mineral nutrition of seedlings

Sub-superficial (0.4 to 0.8 m depth) samples of an Alic Red yellow Latossoil with a clay texture were used from Viçosa, State of Minas Gerais, Brazil (Table 1). The samples were dried, ground to pass through a 4 mm sieve and divided into subsamples of 3.6 kg. The treatments composed by nine combinations of liming and N, P, K and S doses levels were compared to a control that did not receive any liming materials or fertilizers (Table 2). The levels of corrective material were based on soil liming demand (LD) (LD = 2 x Al^{3+} + 2 - (Ca^{2+} + Mg^{2+}).

Table 1. Physical and chemical properties of the an Alic Red yellow Latossoil used in the glass house experiment

Parameter	Unit	Value
pH (1)	-	4.60
Al^{3+}(2)	$cmol.kg^{-1}$	0.78
Ca^{2+} (2)	$cmol.kg^{-1}$	0.04
Mg^{2+} (2)	$cmol.kg^{-1}$	0.02
P (3)	$mg.kg^{-1}$	0.40
K (3)	$mg.kg^{-1}$	5.00
S (4)	$mg.kg^{-1}$	5.10
Organic carbon (5)	%	1.60
Sandy (6)	%	31.00
Silt (6)	%	14.00
Clay (6)	%	55.00
CEC (7)	$cmol.kg^{-1}$	6.57

1) water (1:2.5)
2) Extract. KCl 1.0 N
3) Extract. Mehlich-1
4) Extract. $Ca(H_2PO_4)_2$, 500 mg P kg^{-1} in HOAc 2N
5) Walkley & Black procedure
6) Granumetric fractions
7) Cation-exchange capacity

As a liming was used a mixture of commercial $MgCO_3$ and $CaCO_3$. The N, P and K were applied as salt solutions of $NH_4H_2PO_4$, NaH_2PO_4 H_2O, KH_2PO_4, KCl and NH_4NO_3. The S source was $CaSO_4$ $2H_2O$.

Table 2. Liming demand levels and doses of N, P, K and S applied to soil samples

Liming Demand	Doses Applied ($mg.Kg^{-1}$)			
	P	N	K	S
0.175	180	120	150	60
0.175	420	120	150	60
0.455	180	120	150	60
0.455	420	120	150	60
0.315	300	120	150	60
0.000	180	120	150	60
0.630	420	120	150	60
0.175	30	120	150	60
0.455	570	120	150	60
0.315	300	0	150	60
0.315	300	30	150	60
0.315	300	60	150	60
0.315	300	120	150	60
0.315	300	180	150	60
0.315	300	300	150	60
0.315	300	120	0	60
0.315	300	120	75	60
0.315	300	120	225	60
0.315	300	120	300	60
0.315	300	120	150	60
0.315	300	120	150	30
0.315	300	120	150	90
0.315	300	120	150	120
0	0	0	0	0

In the study of liming and P doses the plants received 80 mg $N.kg^{-1}$, 100 mg $K.kg^{-1}$, and 60 mg $S.kg^{-1}$, while in the study of the response to N, K and S the plants received 300 mg $P.kg^{-1}$ and 0.315 of the liming demand (Table 2).

After the application of the treatments, each experimental plot, consisting of six plastic bags containing 0.6 kg of treated soil, received 18 taxi-branco seeds. Forty-five days after

planting was done a pruning, leaving only one plant per bag.

The assays were carried out in a glass house with three replicates. After 190 days of planting the tops of the plants were harvested and the leaves separated from the branches and stems. Soil samples of each plot were taken to evaluate P and K (extraction with Mehlich-1), Ca and Mg (extraction with 1N KCl) and S (extraction with $Ca[H_2PO_4]_2$, 500 ppm of P in HOAc 2N). The plant samples were dried at 70°C, then weighed and ground. From the extract obtained by the nitre-perchloric digestion, K was determined by flame photometry (A.O.A.C., 1975), S by turbidimetry (Blanchar *et al.*, 1965), P by a colorimetric procedure (Braga and Defelipo, 1974), Ca and Mg by atomic absorption and N by Kjeldhal method (Bremner, 1965).

Regression analyses were carried out for leaves dry matter production (LDM), twigs + stems (SDM) and total (TDM) as a function of liming levels and P, N, K and S doses. With these equations, the necessary doses to reach 90% of maximum production were estimated. By substituting the necessary doses of P and K in the regression equation for the recovered P and K in relation to the added P and K doses were determined, the critical levels of these nutrients in the soil. The foliar N, P and K critical values were obtained by substitution of the necessary N, P and K doses in the regression equations adjusted to the foliar N, P and K content, as dependent variables of the doses added to the soil. To choice the equations by the regression analysis, were tested the linear, quadratic and root-quadratic models. The coefficients of the models were examined by F test, using the mean square error of the variance analysis of the experiment.

- Silvicultural information

Silvicultural information were taken from different places in State of Para under homogeneous plantations where the soil was recovered after mined of bauxite and in natural soil conditions.

Results and Discussion

- Mineral nutrition of seedlings

The fertilizers applied resulted in considerable growth gains to the taxi-branco plants and no response to the addition of lime (Table 3). The absence of response may be related to the fact that these treatments received 60 mg $S.kg^{-1}$ as gypsum. The Ca content in this salt (0.37 $cmol.kg^{-1}$) was sufficient for the requirements of taxi-branco and, as the soil used contained low levels of Ca (0.04 $cmol.kg^{-1}$), it follows that seedlings of taxi-branco have a low requirement for this nutrient.

Increments of P doses resulted in increments of P foliar contents (Table 4). The soil critical level of P was 26.06 $mg.kg^{-1}$ using the model adjusted for TDM, and 0.12% of the critical foliar P content with the model adjusted for LDM. These values were close to those observed by Novais *et al.* (1982) for *Eucalyptus* sp.

Table 3. Regression equations adjusted for the production of total dry matter ($g.pot^{-1}$) of the aerial parts (TDM) as dependent variables of levels of liming and P, N and K doses

Regression Equation	R^2
Response to liming and P applied	
$TMD = 3.71699 + 0.05134^{***} P - 0.000059^{*} P^2$	0.9444
Response to N applied	
$TMD = 9.40775 + 0.05735^{**} N - 0.000166^{*} N^2$	0.8390
Response to K applied	
$TMD = 9.92005 + 0.04669^{*} K - 0.000125^{*} K^2$	0.9088

° Significant at 0.1 probability level
* Significant at 0.05 probability level
** Significant at 0.01 probability level
*** Significant at 0.005 probability level

The positive response of N addition can be confirmed by the equation for TDM as variables dependent on the N doses added to the soil (Table 3). In accordance with this model, 90% of maximum production (12.92g) obtained can be

obtained by one dose dose around 79 mg of N kg^{-1} of soil.

Table 4. Regression equations adjusted for the P, Ca, Mg, N and K concentrations (%), in the dry matter of leaves and twigs + stems, as dependent variables of the applied doses of N and K

Regression Equation	R^2
$P= 0.05855 + 0.000335^{***} P - 0.00000042^{*} P^2$	0.8825
$Ca= 0.1745 + 0.00103^{*} P + 1.16715^{***} NC - 0.00000187^{*} P^2 - 0.95461^{*} NC^2$	0.9088
$Mg = 0.0802 - 0.00011^{***} P + 0.08953^{***} NC + + 0.00000016^{**} P^2 - 0.07326^{*} NC^2$	0.9725
$N= 1.41731 + 0.011643^{***} N - 0.0000213^{*} N^2$	0.9616
$K= 0.28103 + 0.004501^{***} K + 0.0000093^{***} K^2$	0.9582

° Significant at 0.1 probability level
* Significant at 0.05 probability level
** Significant at 0.01 probability level
*** Significant at 0.005 probability level

In spite of a quadratic response to the N addition, taxi-branco is regarded as a legume tree with the capacity to associate with atmospheric N_2-fixing bacteria. Therefore, if Rhizobium inoculated seeds were to be used, the plants may exhibit a different response to the addition of these nutrients.

According to the adjusted model for TDM as the dependent variable of K doses (Table 3), the soil critical level obtained for this nutrient was 27.4 $mg.kg^{-1}$. This value is close to that obtained by Barros *et al.* (1982) for *Eucalyptus grandis*.

The N and K contents in the tops of the plants increased with increasing doses of these nutrients (Table 4). On the other hand, this was not observed for the S contents in the aerial parts of the plants, since they did not alter significantly with the different S doses applied.

As the recommended N and K doses (79 mg $N.kg^{-1}$ and 80 mg $K.kg^{-1}$) are substituted in the adjusted equations for N and K foliar contents as dependent variables of the applied doses (Table 4), the critical foliar levels of 2.2% of N and 0.7% of K were obtained.

The absence of response to the S doses can be seen by the lack of adjustment of significant mathematical models for the different analyses of variables. This indicates that the original S content in the soil was higher than the critical level for taxi-branco. On the other hand, the adjusted equation for the S recovered from the soil by the extractant, as a function of the applied doses (Table 5), shows an increase of the availability of this nutrient with the doses. Therefore, it can be suggested that the critical S level for taxi-branco is lower than 5.1 $mg.kg^{-1}$ of soil.

With the same extractant from a soil with a clay content similar to that used in the present study (53%), Alvarez *et al.* (1983) obtained a critical S level of 5.3 $mg.kg^{-1}$ of soil for 90% of maximum production of *Eucalyptus grandis* seedlings.

Table 5. Regression equations adjusted for the P, Ca, Mg, K and S recovered from the soil by the extractants, as dependent variables of the applied levels of liming and P, K and S doses

Regression Equation	R^2
$P= - 5.62804 + 0.114598^{***} P$	0.9198
$Ca= 0.24512 + 1.51045^{***} NC$	0.9899
$Mg= - 0.023941 + 0.46401^{***} NC$	0.9590
$K= 4.4099 + 0.2362^{***} K + 0.00064^{**} K^2$	0.9986
$S= 4.5943 + 0.0680^{*} S + 0.00058^{*} S^2$	0.9899

* Significant at 0.05 probability level
** Significant at 0.01 probability level
*** Significant at 0.005 probability level

- Silvicultural information

Sclerolobium paniculatum Vogel has four varieties (paniculatum, subvelutinum, rubinosum, and peruvianum) with the difference found mainly in the characteristics of the leaflets. The States of Pará and Amazonas in Brazil are the main regions where the paniculatum variety occurs (Pereira, 1990).

In homogenous plantations taxi-branco has an architecture similar to eucalyptus plantations. The adult trees

found near Santarem, possess heights varying between 20 m and 30 m and a Diameter at Breast Height (DBH) of 70 to 100 cm. In secondary succession occupies open spaces and is characterised as a light demanding species, with a great capacity for adaptation in unfavourable soil conditions (Carpanezzi et al., 1983; Erfurth & Rusche, 1976; Lemeé, 1956; Ducke, 1949 and Correa, 1931).

Phenology observations in Trombetas National Forestry indicated that flowering occours between may throught july and frut production in August until November (Mineração Rio do Norte, sn). In Santarem region, near Trombetas National Forestry, fruit matures and the dissemination of the seeds occurs from January to March.

The taxi-branco seeds are small and hard. It was observed about 15,000 seeds per kilo (Mineração Rio do Norte, sn).

The mechanical dormancy of seeds can be overcome with the application of adequate technologies (Carpanezzi et al., 1983 and Carvalho & Figueiredo, 1991) and its germination occurs in approximately 30 days.

Field experimentation has shown good silvicultural performance in relation to other species considered pioneers in secondary succession (Tables 6 and 7). The analysis of the performance of taxi-branco with other native species must consider the fact that, seeds from mother trees, chosen in locals of natural occurrence were used and therefore there is a low index for genetic selection. So, the development of studies for genetic improvement to the determination of better provenance and adequate silvicultural practises, may improve, even further, the performance of taxi-branco.

Taxi-branco has showed its potential for reclamation of degraded land in a trial held in abandoned pasture in Paragominas (Table 8). This behavior can be due to its capacity to fix nitrogen and to associate to endomicorizal fungi (Table 9).

Table 6. Increments average values of plant height (m) and Diameter at Breast height (DBH) (cm) for taxi-branco at different ages (months) planted in Belterra

Age	Plant height	Plant DBH
24	1.00	2.80
60	2.40	3.70
72	1.90	3.80
108	1.80	2.80
180	1.10	2.50

Adapted from Matos (1993)

Table 7. Average values for survival (%), height (m), DBH (cm) and volume (m^3/ha) in different ages for some native and exotic species of rapid growth planted with a 3 m x 2 m spacing at Belterra, State of Para

Species	Age	Survival	Height	Dbh	Volume
*Vismia*sp	66	91,7	8,6	7,6	31,9
Laetia procera	66	96,7	7,6	8,6	36,9
Acacia mangium	30	97,0	7,4	10,7	58,2
S. paniculatum	66	94,7	12,2	9,1	105,7
E.urophylla	78	85,5	13,4	13,2	167,2
Jacaranda copaia	78	94,7	12,4	14,5	175,2
Eucalyptus grandis	78	74,7	16,0	16,0	257,2

Adapted from Yared et al. (1988)

Table 8. Growing of height (m), DBH (cm) and dossel area projection (m^2) of eight forestry species planted directly in abandoned pasture in Paragominas

Specie	Height	DBH
*Tabebuia serratifolia**	2.80	1.80
*Bertholletia excelsa**	3.30	5.10
*Dipteryx odorata**	3.60	3.30
*Cedrela odorata**	4.00	4.70
*Swietenia macrophylla**	6.30	6.30
*Sclerolobium paniculatum***	6.40	8.70

* plants with five years old

** plants with three years old

Adapted from Pereira & Uhl, in press

Table 9. Root infection of endomicorize fungi of fast growth tree species planted in abandoned pasture in Paragominas

Specie	Root infection (%)
Swietenia macrophyla	90.0
Cecropia palmata	86.0
Cordia multispicata	82.0
Solanum crinitum	80.2
Anacardium occidentale	80.8
Bertholletia excelsa	79.8
Dipterix odorata	74.2
Acacia mangium	65.7
Sclerolobium paniculatum	58.3
Stryphenodendrum pulcherrimun	57.8
Cariocar vilosum	56.2
Dydimopanax morototoni	56.0
Schyzolobium amazonicum	49.7

Adapted from Oliveira Júnior (unpublished data)

The biomass of taxi-branco trees is proportionally greater for the trunk (247.01 kg), branches (97.81 kg) and leaves (22.13 kg). The greatest concentrations of nutrients were found in the leaves followed by the branches and trunk. Nitrogen was the nutrient observed in the greatest proportion in the different parts of the plant (Matos, 1993).

In homogenous plantation at the Experimental Station in Belterra, the litter production of taxi-branco was about 2.7 times greater than *Eucalyptus citriodora* (Table 10). Besides this, the chemical composition of the litter incorporated to the soil is another important characteristic to be considered. This has immediate implications in relation to its ability for decomposition and augmentation of organic material to the soil, especially when one is dealing with recuperating degraded soils. In this context, when comparing the chemical composition of the litter of taxi-branco with that of *E. citriodora*, one can verify the advantage of using a leguminous specie for the recuperation of soil fertility (Table 10). Although taxi-branco has low ratio of C/N, that facilitates decomposition, the high quantities of N, P, K, Ca and Mg on litter that reach the soil are superior to those of eucalyptus (Table 10).

Table 10. Production of litter ($t.ha^{-1}.yr^{-1}$), C/N ratio in litter and quantities ($t.ha^{-1}.yr^{-1}$) of N, P, K, Ca and Mg in litter of homogenous plantations of taxi-branco and *E. citriodora* at Belterra, State of Para

Parameter	Species	
	Taxi-branco	E. citriodora
Production of litter	7.7	3.3
C/N ratio	40.0	69.0
N	32.0	28.0
P	2.3	1.0
K	3.9	2.3
Ca	13.9	13.7
Mg	5.4	3.3

Brienza & Yared (unpublished data)

The potential of taxi-branco for the recuperation of degraded soils has been confirmed with plantations in areas where bauxite has been mined by Mineração Rio do Norte, in Porto Trombetas. Among the different native species used, taxi-branco has been prominent (Table 11). The rapid formation of litter may provide favourable conditions for establishment of other pioneer species in order to promote the increase of biodiversity in an imitation of the natural succession process.

Table 11. Growth in height (m) and diameter at breast height-Dbh (cm) at seventy month of different native tree species planted in area with superficial top soil after buxite mined

Specie	Height	Dbh
Himenolobium excelsum	5.20	8.0
Miconia longifolia	9.40	11.0
Bowdichia sp	6.50	8.0
Sclerolobium racemosa	10.60	26.6

Adapted from Knowles (1994)

A study of undisturbed soil samples, collected under different vegetation after the exploitation of bauxite in Porto Trombetas, using *Acacia auriculiformis* as an plant indicator, suggest that the *A. auriculiformis* growth was probably favoured by the better soil conditions promoted by taxi-branco (Table 12).

Table 12. Height, roduction of dry matter of roots and aerial parts of *Acacia auriculiformis* planted in no-modified soil collected under: natural regeneration (NR); homogeneous stands of taxi-branco; mixed plantings of native species (MPN); mixed plantings with exotic species (MPE) and homogeneous stand of *Acacia mangium*, at the Mineração Rio do Norte in Porto-Trombetas

Soil type	Height (cm)	Aerial dry matter (g)	Root dry matter (g)
NR	17.3	2.6	0.6
Taxi-branco	16.5	2.5	0.4
MPN	14.3	2.2	0.3
MPE	13.6	2.2	0.3
Acacia mangium	8.7	1.5	0.1

Conclusion

The results of this work permitted the determination of the critical levels of P and K in the soil for taxi-branco, as being 26.1 and 27.4 $mg.kg^{-1}$ of soil respectively. The N dose required to obtain 90% of maximum production of total dry matter was 79 mg kg^{-1} of soil. The N, P and K critical foliar contents were 2.20%, 0.12% and 0.70% respectively. Because of the absence of response to liming and S addition, the Ca and S critical levels were estimated as being lower than 0.37 $cmol.kg^{-1}$ and 5.1 $mg.kg^{-1}$ of soil respectively.

The low Ca requirements of taxi-branco, its tolerance for exchangeable Al in the order of 1 meq/100 cm^3 of soil, its capacity fix N and associate with micorizal fungi are certainly factors which contribute to its success as a pioneer in secondary succession, especially in conditions of low soil fertility, as was confirmed based on Trombetas, Belterra and Paragominas plantations and experimentations.

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Field Inoculation Rates of Mycorrhizal Fungi in Revegetation of Abandoned Coal Mine Lands. Robert K. Noyd and F. L. Pfleger, Department of Plant Pathology, University of Minnesota, St. Paul, MN 55108

Abstract: Abandoned coal mine land (AML) sites in southern Illinois and western North Dakota contain areas that are difficult to revegetate due to low fertility (1-3 mg kg-1 N and P), little organic matter, and acidic (3-4, Illinois) or alkaline (~8, North Dakota) pH. Areas such as these may benefit from inoculation with arbuscular mycorrhizal (AM) fungi to assist in the establishment of vegetative cover. Potential sources of adapted mycorrhizal inoculum were found in reclaimed overburden sites with large AM fungal spore densities (100 and 33 spores g^{-1} Illinois and North Dakota, respectively). Soils from these locations were used to determine an infective inoculation rate by a mycorrhizal inoculum potential (MIP) bioassay. Inoculum, consisting of rhizosphere soil and dried roots, was mixed into overburden in proportions of 0, 1, 2.5, 25, 50 and 100% (w/w), placed into containers, and sown with a single 12-day-old seedling of *Andropogon gerardii* Vitm. (big bluestem), a native prairie species known to respond favorably to AM fungi. After 14 days, shoots were dried and weighed and the root system was collected, cleared, stained, and assessed for percent root length colonized by AM fungi. An inoculum proportion of 1% in Illinois and 2.5% in North Dakota overburden produced moderate (16%) root colonization. These inoculum proportions were selected for rates of field inoculation because they were the lowest proportions that were both infective and effective in increasing shoot biomass of *A. gerardii*. In both soils, this level of root colonization was about one-third of the maximum colonization (50%) obtained with 25, 50, and 100% proportions of inoculum. Using adapted AM fungi and *A. gerardii*, MIP bioassays can be used to determine a field inoculation rate that has the potential to establish populations of beneficial mycorrhizal fungi and enhance chances of successful revegetation.

Additional Key Words: arbuscular mycorrhizal fungi, mineland reclamation, native prairie grasses, mycorrhizal inoculum potential, *Andropogon gerardii*

Lateral Thinking on Minesite Rehabilitation in Australia. D.J. Williams

Abstract: The Australian mining industry has disturbed less than 0.03 % of the land mass in 200 years, compared with 65 % used for agricultural activity (mostly for sparse grazing). About 0.1 % of Australia's land is cleared every year, removing native habitat. The rehabilitation of mined land conventionally involves smoothing of the disturbed land, followed by revegetation. The end land use is typically designated as cattle grazing, even though this is unsustainable in the more arid climates. Open-cut mining results in an elevated land form. Smoothing generates a mounded final land form of enlarged area, which tends to direct increased rainfall runoff, erosion products, and any contamination offsite. An alternative strategy is to minimise the area of disturbance and to contain any potential offsite impacts within the mined area. This can be achieved with far less earthworks than is required for smoothing, and reduces offsite impacts. A dished plateau land form allows grazing, while retaining steeper slopes is compatible with native habitat re-construction, reflecting the sharp relief which often dominates in nature. These alternative strategies are being trialed at a number of operating Australian mines.

Additional Key Words: cattle grazing, erosion, rainfall runoff.

Reclamation of the Wahsatch Gathering System Pipeline in southwestern Wyoming and northeastern Utah. D. Strickland, G. Dern, G. Johnson and W. Erickson.

Abstract: The Union Pacific Resources Company (UPRC) constructed a 40.4 mile pipeline in 1993 in Summit and Rich Counties, Utah and Uinta County, Wyoming. The pipeline collects and delivers natural gas from six existing wells to the Whitney Canyon Processing Plant north of Evanston, Wyoming. We describe reclamation of the pipeline, the cooperation received from landowners along the right-of-way, and mitigation measures implemented by UPRC to minimize impacts to wildlife. The reclamation procedure combines a 2 step topsoil separation, mulching with natural vegetation, native seed mixes, and measures designed to reduce the visual impacts of the pipeline. Topsoil is separated into the top 4 inches of soil material mixed with ground up vegetation and the remaining 8 inches of soil material, when present. The resulting top dressing is rich in native seed and rhizomes allowing a reduced seeding rate. The borders of the right-of-way are mowed in a curvilinear pattern to reduce the straight line effect of the pipeline. Monitoring of reclamation success illustrates the effects of landowner cooperation on revegetation. Specifically, following 2 years of monitoring, significant differences in plant cover ($0.01<P<0.05$) exist among regions of the pipeline. Observations suggest that revegetation may be heavily influenced by grazing management by individual landowners. Observations also suggest that growth of sagebrush plants from seed germination is exceeding growth from sagebrush plants planted as tublings.

Additional Key Words: erosion potential, grazing, stratified sampling.

Case Studies of Ecosystem-Based Approaches To Remediation. K. Trimble.

Abstract: Applications of the ecological sciences to site remediation have becoming increasingly common, as objectives have expanded from surface stabilization and aesthetic improvement to actual ecosystem reconstruction. In the fields of surface mining reclamation, specific techniques are often applied to common problems such as slope instability and erosion. The influence of larger scale physical and biological pressures on a site from the surrounding ecosystem, such as vegetation succession, is usually ignored. These processes affect the success of reclamation techniques, the management effort required to achieve success, the appropriateness of choices where alternative techniques exist, and the long term ecosystem sustainability. We stress a need for design approaches that examine the broad ecological context of site specific projects. Using cases study examples, we discuss cost-effective considerations including successional trajectory, bioregional wildlife and vegetation management criteria, and large scale biodiversity targets. Such considerations are used in establishing goals for site specific projects, and as tools in choosing appropriate techniques. In one example, the rehabilitation design for a limestone quarry in southern Ontario addressed regional aquatic habitat requirements, wildlife and forest community targets, and bioregional populations of internationally significant species, while at the same time minimizing approval and maintenance issues.
Additional Key Words: ecosystem restoration, biodiversity.

Abandoned Mined Land Impacts on Water and Sediment Quality, and Invertebrate Assemblages in Two Virginia Watersheds. J.L. Yeager, J. R. Bidwell, D. S. Cherry and C. E. Zipper. The constituents of abandoned mined land (AML) discharges (acidic pH, metals, dissolved solids, total suspended solids) can be toxic to aquatic life. Studies were undertaken to determine environmental impacts of acid mine drainage(AMD), a component of AML, in the Black Creek and Ely Creek watersheds, Wise and Lee Counties, Virginia. Conductivity and pH in the stream were measured to survey the magnitude of AMD discharge within each system. Water, sediment and water/sediment mixtures that simulate storm events were analyzed for metal content (Al, Fe, Mn, Zn, Cu, Mg). Benthic macroinvertebrates were collected seasonally using D-framed nets to determine AMD effects on relative abundance and taxon richness. Acidic pH ranged from 2.15-3.30 at three AMD-influenced seeps and varied from 6.40-8.00 at reference stations. Conductivity (µmhos/cm) ranged from 32-278 at reference sites and from 245 to >6000 at AMD-impact sites. Benthic macroinvertebrate abundance and taxon richness were notably lower in the seeps having only 1-3 taxa totalling < 10 organisms as compared to reference areas where richness values were 12-17 and comprised 300-977 organisms. Sediments from selected areas within Black Creek caused significant reductions in Daphnia magna reproduction relative to reference site sediments in 10 day chronic toxicity test. Concentrations of Fe, Al, Mg, Cu, and Zn were highest in the AMD influenced stations with low pH and high conductivity.

Additional Key Words: Acid Mine Drainage, Metals

Revegetation Standards for Floodplain Forest Ecosystems in Western Washington, USA. B. Zamora (Natural Resource Sciences, Washington State University, Pullman, WA 99164-6410).

Abstract. Mining activity within floodplain landforms of western Washington, USA, presents unique problems in terms of approaches to revegetation and the success standards to be use to quantitatively evaluate revegetation success. Persistent historical disturbance of floodplain areas of the region has left little undisturbed natural vegetation to use as reference sites for development of success standards. A strategy is proposed for use of an ecological model of succession within floodplain vegetation to both identify revegetation options and provide a quantifiable and ecologically dynamic framework of success standards for revegetation evaluation. The floodplain forest mosaic of mined lands in western Washington is a combination of (1) aquatic sites of open surface, impounded or flowing waters, (2) minerotrophic wetlands, and (3) xeroriparian sites between wetlands and uplands. Seven distinct and persistent plant communities of three physiognomic types (herb-dominated, shrub-dominated, and tree dominated community types) are present are common. These physiognomic groupings are strongly related to the degree of seasonal flooding and are successionally linked as habitats change from one flooding regime to another. Remnant stands of floodplain vegetation were used to construct a successional model which will provide for revegetation guidelines and a framework of success standards.

Additional Keywords: ecological classification, wetlands