THE IMPACTS OF COAL REFUSE/FLY ASH BULK BLENDS ON WATER QUALITY AND PLANT GROWTH[1]

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Abstract: There is considerable interest in the beneficial reuse of coal fly ash as a soil amendment on coal refuse piles. One method of application would be to blend the coal refuse and the fly ash before deposition in a refuse pile. A field experiment was initiated to measure the effects of bulk blending fly ash with coal refuse on water quality and plant growth parameters. Fly ash (class F) from three sources were used in the experiment. Two of the fly ashes were acidic and the third was alkaline. Trenches were excavated in a coal refuse pile to a depth of 2 m and the refuse was blended with fly ash and then returned to the trench. In other plots the ash was applied as a surface amendment. A treatment of a bulk blend of 5% (w/w) rock phosphate was also included in the experiment. Large volume lysimeters were installed in some trenches to collect the leachates. The fly ash treatments appear to improve the quality of the leachates when compared to the leachates from the untreated plots. The fly ash amended treatments have lower leachate concentrations of Fe and Al. Initially the fly ash treatments showed high levels of leachate B, however those levels have decreased with time. Millet (*Setaria italica*) yields from the first year of the experiment were highest on the alkaline fly ash and rock phosphate blended plots. In the second growing season, the two bulk blends with alkaline fly ash had the highest yields. In the third growing season all treatments had higher yield levels than the untreated control plots. The positive effects of the fly ash on leachate quality were attributed to the alkalinity of the ash, and the increase in yield was attributed to the increases in water holding capacity due to fly ash treatments.

Additional Key Words: Acid Mine Drainage, Neutralization Potential

Introduction

Recently, the United States Environmental Protection Agency (USEPA) declassified fly ash as a hazardous waste material (USEPA, 1993). This led to increased interest in backhauling fly ash to the areas where coal is mined. Currently this backhaul is mainly for disposal purposes, however, fly ash (especially alkaline fly ash) may be beneficially reused to control the generation of acid mine drainage (AMD) from coal refuse. Coal refuse is the material removed from coal during the coal cleaning process. Eastern coal refuse usually contains significant amounts of iron pyrite and most refuse piles produce some AMD. The AMD produced must be treated before it is released to local surface waters. Blending an alkaline agent, such as lime or an alkaline fly ash, would keep the refuse at a high pH which will greatly reduce the rate of pyrite oxidation and provide alkalinity to neutralize any acidity produced. To be an effective treatment the ash would probably have to be intimately mixed with the refuse or layered.

The literature on the use of fly ash on mined lands was recently reviewed by Carlson and Adriano (1993). The main benefits of using fly ash are increased alkalinity and improved water holding capacity. Fly ash has also been shown to be effective in controlling AMD generation from some very acidic coal refuse materials in column studies (Jackson, 1993; Stewart et al., 1994). The purpose of this study was to expand upon our ongoing column studies with a field study. The goals have been to assess the water quality impacts of bulk blended fly ash/coal refuse and to determine the effects of these blends on plant growth and elemental uptake.

Background

Currently about 64 million t of fly ash are produced each year in the U.S. (USEPA, 1988). Fly ash makes up about 80% of the wastes associated with the burning of coal. Presently only about 20% of the ash generated is utilized with the remaining material being deposited in landfills and surface impoundments (USEPA, 1988). Disposal of fly ash poses the greatest environmental impact of coal combustion waste at the present time.

When coal is burned in the boiler of a power plant, two residues are produced. Bottom ash, or slag remains in the boiler, and fly ash rises with the flue gases. In many plants, up to 90% of the ash produced is in the form of fly ash. Fly ash is removed from the flue gases by many methods. Bag houses act as filters to collect the larger ash particles as do the mechanical collectors which work by centrifugal force. Neither of these methods results in a sufficient separation to meet EPA requirements, but they are effective in removing larger ash particles. Wet scrubbers remove fine particles by water entrapment. Electrostatic precipatators operate by imparting a negative charge on the ash particles and then collecting them on a positive electrode. After collection most ash is sluiced to a holding pond where it is dewatered and then moved to a landfill.

The physical properties of fly ash depend upon a number of factors, including the type of coal burned, the boiler conditions, the type and efficiency of the emission controls, and the disposal method (Adriano et al., 1980). Certain characteristics tend to be similar in most ashes. Fly ash is mainly composed of silt sized materials having a diameter from 0.01 - 100 μm (Chang et al., 1977). When compared with mineral soils, fly ash has lower values for bulk density, hydraulic conductivity, and specific gravity. Both crystalline (mullite) and amorphous (glass) phases have been identified by X-ray diffraction in fly ash (Mattigod et al., 1990).

One property of some fly ashes which makes them attractive as soil amendments are their liming potentials. In a study of 23 ashes from across the U.S., Furr et al., (1977) found that ash pH ranged from 4.2 to 11.8. Most low pH ashes came from eastern coal sources. Ash pH values of up to 12.5 have been reported (Chang et al., 1977). Many ashes are high in Ca and Mg oxides and have a fairly high neutralizing capacity. Fly ashes with neutralizing capacities of up to 10% calcium carbonate equivalent (CCE) have been reported but CCE values of 1-6% are more common (Aitken, et al., 1984). This means that more than 20 t of most fly ash would be required to provide the same neutralization as 1 t of ground limestone. This would indicate that ash would not be effective in raising a highly buffered, low pH soil due to the large amount of ash needed, but may be effective in poorly buffered systems. Water soluble Ca content was found to be the best indicator of the potential to produce alkalinity (Theis and Wirth, 1977). Unweathered fly ash also has high levels of soluble salts. Ash from lignite and subbituminous coals tend to have the highest levels of soluble salts among fly ashes (Adriano et al., 1980). The application of 80 tons/acre of unweathered fly ash was found to increase soil salinity 500 to 600% and also caused significant increases in soluble B, Ca, and Mg (Page et al. 1979). Values for saturated paste conductivity for fly ash can be as high as 6 S m^{-1}.

Recent column studies (Jackson, 1993; Stewart et al., 1994) using coal fly ash and acid forming coal refuse indicate that fly ash is an effective treatment for reducing or eliminating AMD from coal refuse, if there is enough alkalinity added with the fly ash. These findings also emphasize the need to balance alkalinity against the acidity potentially produced by pyrite oxidation in the refuse. Columns in which insufficient alkalinity was applied to offset the acidity generated by pyrite oxidation produced leachates with higher metal levels than untreated coal refuse. The higher metal levels in ash treated columns with insufficient alkalinity were attributed to the acid stripping of the metals in fly ash.

Materials and Methods

The field experiment involving bulk blended fly ash was installed at the Westmoreland Coal Co.'s Pine Branch Refuse (PB) area in Wise County, VA. The refuse at Pine Branch is composed of material generated

from cleaned coal from the Taggart, Taggart Marker, and Standiford coal seams. It has a relatively low potential acidity of 38 tons $CaCO_3$/1000 tons refuse by the H_2O_2 method (Barnhisel and Harrison, 1976). Three types of fly ash were used at this site. Clinch River fly ash (CRF) is an alkaline (pH 8.5) ash with a moderate neutralization potential (CCE = 3.0). Chesterfield fly ash (VCF) had a variable pH and the material used in this experiment had a pH of 4.7. The Mead Paper fly ash is a low pH fly ash with a pH of 3.7. The site we are working on is the upper lift of a pile which was idled in the late 1980's. The drainage from this pile has significant amounts of Fe and Mn, and is currently being treated with a NaOH drip system. Large expanses of $AlSO_4$ salts have been observed on this pile. Plots were laid out and sampled March of 1992. The experiment was installed and seeded during late May and early June of 1992.

Plots at PB were laid out in a completely randomized design with 10 treatments and 4 replications (40 plots)(Table 1). The dimensions of each plot were 3 m by 5 m. The treatment area in those plots receiving bulk blended treatments was a 0.8m strip down the middle of the plot. In those treatments which received surface applied amendments the entire plot area was treated. The bulk blended treatments were manufactured by excavating a trench 0.8 m wide 5 m long and 2.2 m deep in the middle of a plot. The refuse was piled between the plots during excavation and then was moved off the experimental area to be mixed with fly ash or rock-P. The fly ash or rock-P and refuse was mixed with a front end loader until well blended and then returned to the trench. Fly ash mixes were based on volume. For a 33% mix, for example one loader bucker of ash was mixed with two buckets of refuse. Rock-P mixes were blended to attain a mixture that was approximately 5% rock-P by weight. Water sampling lysimeters were installed in 17 selected plots in the PB experiment. The lysimeter tops were buried 30 cm deep to allow for tillage of the plots over the lysimeters. The lysimeters consisted of a 1.5 m length of 61 cm diam. smooth bore ABS plastic drainage pipe with a fitted endcap. The endcap was plumbed with a 1.3 cm (diam) nylon pipe T whose ends were covered with 80 mesh polyester sieve cloth. This T was mounted in the bottom of the endcap with the outlet of the T sticking through a 1.3 cm hole in the endcap. A 10 cm length of 1.3 cm (ID) clear tygon tubing was attached to this T on the outside of the endcap. This tubing was threaded into a 18.9 L covered bucket with a 1.9 cm hole drilled in the lid. The bucket was fitted with 4.5 m of 0.64 cm (diam) Nalgene LDPE tubing. This tubing was run back to the ground surface to allow leachate to be pumped out of the bucket. The bottom of each lysimeter was filled with 45.4 kg of a clean sand to act as a filter. The lysimeters were filled with the same mixture that the trench around it was filled with. The exception to this would be those lysimeters which received Chesterfield fly ash. In those plots only the lysimeter received the treatment and mixing was done by hand using shovels. Plots which received the surface applied ash treatments had ash applied in a uniform layer on the top of the plot surface. The ash was mixed into the refuse during the chisel plowing process. The Clinch River ash (CRF) was surface applied at the rate of 672 Mg ha^{-1} while the Chesterfield ash was surface applied at the rate of 605 Mg ha^{-1}. A 15 cm lift of topsoil was placed over 4 of the plots which also were bulk blended with CRF. The topsoil was taken from a hollow near the site.

The plots were chisel plowed on June 3, 1992, and were fertilized and seeded with a millet cover on June 10, 1992. The fertilizer rate was 82 kg N ha^{-1}, 252 kg P_2O_5 ha^{-1}, and 72 kg K_2O ha^{-1}. In September of 1992 the millet crop was mowed down to act as a mulch and the plots were seeded with an acid tolerant mix of grasses and legumes. In September of 1992, 1993 and 1994 the vegetative yield of the plots was measured by weighing a clipped sample from a 30 cm quadrat. In September of 1993 soil samples from the upper 5 cm of material beneath the vegetative sample were also taken. These samples have been analyzed for pH and water holding capacity (WHC). Water holding capacity was determined as the difference between the water retained at -0.03 MPa and -1.5 MPa (Klute, 1986).

The leachate from the lysimeters was collected at approximately a 4-week interval beginning July 3, 1992. A vacuum pump was used to draw the leachate into a 1 L suction flask. The first flask collected was discarded as a rinse and the second 1 L flask was collected as the sample. The lysimeter was then pumped dry and the total volume of leachate was recorded. The samples were transported back to the lab and pH and EC were run the following day. If needed the samples were filtered at this time. The samples were acidified with trace metal grade nitric acid and stored in LDPE bottles in a cold room. The samples were analyzed for Al, B,

Cu, Fe, Mn, S, and Zn by Inductively Coupled Plasma Emission Spectroscopy (ICPES). Selected samples were also periodically analyzed for Cr, Mo, Ni, and Pb.

Table 1. List of treatment abbreviations and lysimeter locations at the Pine Branch site.

Treatment	abbreviation	lysimeters installed
33% Clinch River fly ash bulk blended	33% CRF	3
20% Clinch River fly ash bulk blended	20% CRF	0
33% Clinch River fly ash bulk blended + 30 cm topsoil	TOP	0
672 kg ha^{-1} Clinch River fly ash surface applied	SUR	2
33% Chesterfield fly ash bulk blended (lysimeter only)	VCF	3
605 kg ha^{-1} Chesterfied fly ash surface applied	SVCF	0
33% Mead Paper fly ash bulk blended	MPF	3
5% Rock Phosphate bulk blended	RP	3
Trenched Control	CON	3
Untrenched Control	UCON	0

<u>Results</u>

Water quality

After the plots were seeded in June of 1992, the Pine Branch site received a large amount of rain for the area (27 cm over 60 days in June and July). This caused the first lysimeters installed (CON) to elute much faster than we had anticipated. Due to evapotranspiration and low hydraulic conductivity some lysimeters (RP, 33% CRF) did not yield any leachate until late fall of 1992. Some lysimeters receive inputs of groundwater, particularly in the winter and spring. Overall there was a decline in leachate pH over time, when compared to the first leachates collected (Figure 1). This may be due to the disturbance caused by the trenching and blending processes, which may have exposed new surfaces to pyrite oxidation. This trend may also be due to the rapid movement of water through the profile in the rainy period of June and July of 1992. The effects of the different treatments on leachate pH are illustrated in Figure 1. The CRF (33% Clinch River fly ash, bulk blended) and RP (5% rock P bulk blended) treatments have the highest leachate pH values. The high pH of the CRF leachate was due to the added alkalinity from the CRF ash. The CON (untreated control) and SUR (672 Mg ha^{-1} Clinch River fly ash surface applied) had the lowest pH values. These plots received no treatment below the surface and reflect the quality of the pore water moving down through in these materials. The MPF (33% Mead Paper fly ash bulk blended and VCF (33% Chesterfield fly ash bulk blended) leachates were intermediate in pH.

Initially the EC (electrical conductivity) of the leachates from the ash treated lysimeters was quite high due to leaching of entrained salts (Figure 2). The RP treatment also had high EC values that were attributed to the salts contained in the rock phosphate. The untreated refuse (CON) leachates also had fairly high EC values. The EC values of the leachates from the control lysimeters tended to increase with time, as pH decreased (Figures 1 and 2). The EC values from the other treatments declined with time as salts were leached. These EC data represent water that is moving down through the profile and salt content is likely to be less concentrated than the water which is held in the pores of the material. This suggests that high salt levels could reduce plant growth in these materials especially early after ash application.
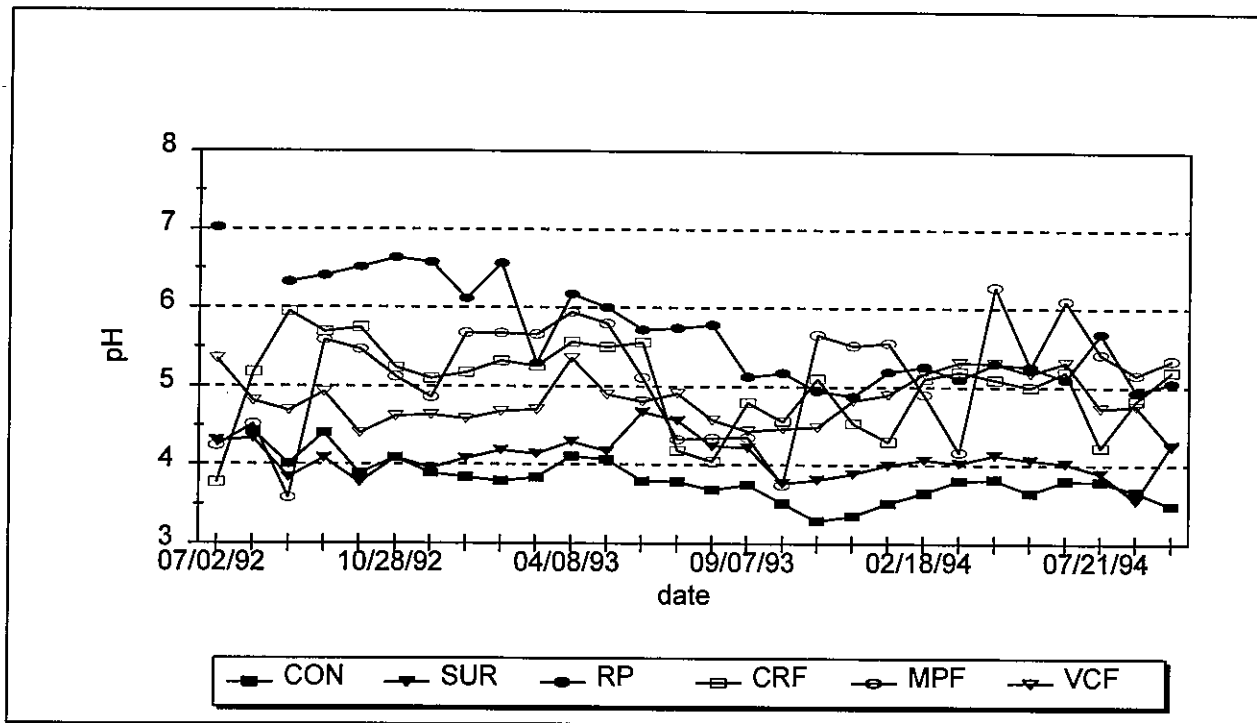
Figure 1. Mean leachate pH values from the lysimeters from July 1992 to September 1994. In some cases a data point may represent a single lysimeter.
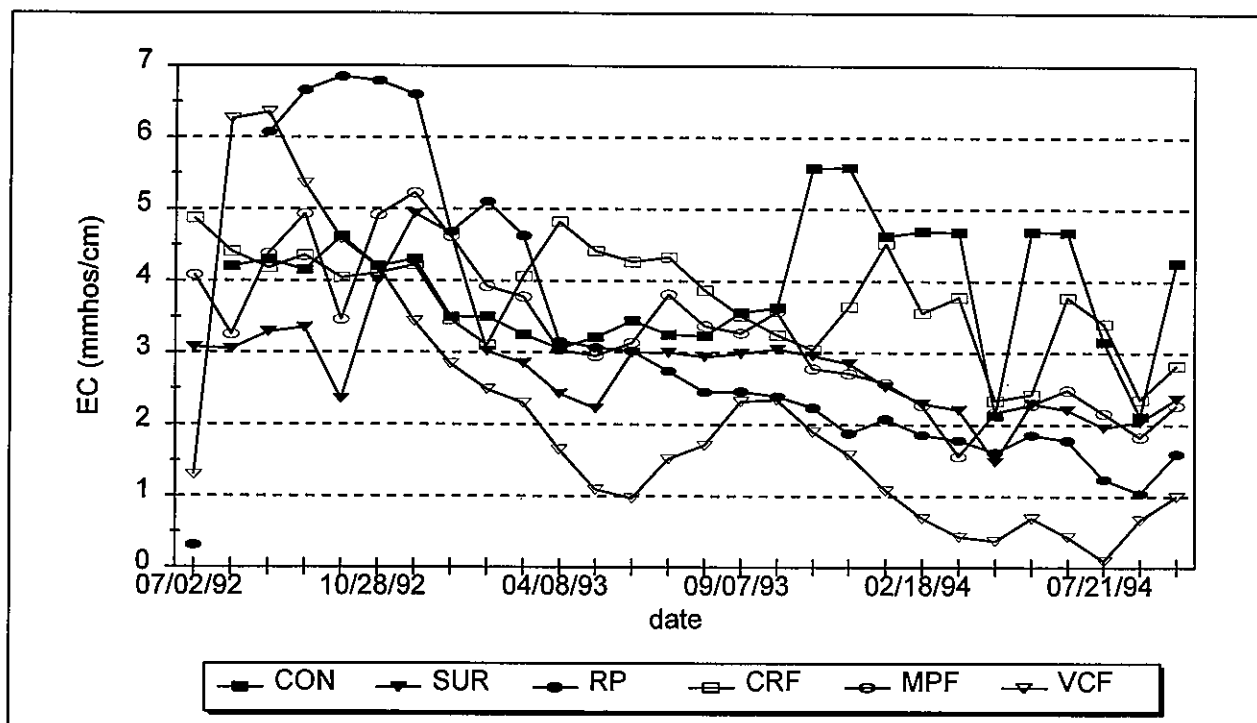

Figure 2. Mean leachate EC values from the lysimeters from July 1992 to September 1994. In some cases a data point may represent the output of a single lysimeter.

The mobility of most metals (Al, Cu, Fe, Mn, Zn) in these systems is largely pH dependent. If the leachate pH remains above 4.5, the levels of these elements remains fairly low (Figures 1, 3 and 4). As the pH drops below 4, the metals become soluble and increasing amounts of the metals are detected. When the pH values drop below 3.5 (CON January 1994 - Figures 1, 3 and 4), metal levels increase dramatically. The Cu and Fe data (Figures 3 and 4 respectively) have been included to illustrate this point. The Zn data is very similar to the Cu data, and the Al data is much like the Fe data. The highest level of Cu in the leachate (21 mg L^{-1} from the control - CON) lysimeters concurs with the lowest pH value. This is also true of the Al, Fe, Mn, and Zn data. The SUR treatment also has a low pH and relatively high Cu levels. Other surface chemistry adsorption mechanisms may also be involved in the binding of metals in some of the ash treatments. In the RP treatments the metals are bound by the high levels of phosphate present. The large differences in the Fe values between the CON treatment and the other treatments makes it difficult to interpret the Fe data for the other treatments (Figure 4). We believe that the low pH, high metal, high EC conditions that are present in the control (CON) lysimeters around January of 1994 were the result of the leaching of a build up of salts that accumulated during the dry fall of 1993. The Mn data is the most variable of the metal data (Figure 5), and is likely due to the many oxidation states of Mn. Leachate Mn contents as high as 25 mg L^{-1} were observed. Leachate Mn content seems to be higher in the winter months than in the summer months in the CON and SUR treatments.

Boron is a problematic element in some fly ash materials and some of the leachates from the ash treated lysimeters contained relatively high levels of B (Figure 6). This is especially true of the early leachates. The VCF ash was known to be high in soluble B and the leachates from these lysimeters had the highest levels of B observed. These levels declined sharply as the borates were leached from the materials. The CRF ash was also known to contain high levels of B, and the B content of these leachates remained elevated through June of 1994. These trends can be attributed to reduced solublility of B as pH increases (Kurier, 1994). The low pH of the VCF bulk blend released B rapidly, while the high pH CRF blend released B to solution more slowly. The untreated control (CON) lysimeter leachates had low levels of B. The leachates from the blended ash treatments can easily be separated from the other treatments on the basis of their B content. There may be some B leaching out of the surface (SUR) treatments. Boron toxicity in plants may occur immediately after ash application, but with leaching, B toxicity appears to become less of a possibility. The aquatic effects of B do not seem to be well documented and it is unclear what effects drainages with elevated B levels will have on surface waters.

Plant Biomass Yield

Growth of the initial millet crop was uneven and was most vigorous in the furrows left by the chisel plow. The chisel plowing loosened the soil and the furrows were sites of water collection. The RP treatment had the highest millet yield in 1992 (Table 2). This indicates that although the conventional fertilizer applied had a large amount of P, the addition of more P would improve yields. The 33% CRF plots also had a high yield, likely due to the increased water holding capacity, and increased pH due to the fly ash treatment (Table 3). All other ash blends had intermediate yields. The trenched controls had relatively high yields indicating that compaction was one factor limiting plant growth on these plots. The surface applied ash treatments (SUR, SVCF) had low yields which were attributed to the high levels of salts in the ash.

The biomass yields of the mixed grass legume stand harvested in 1993 were lower than those of the millet harvested the previous year (Table 2). The trends in the yields are a response to both pH, water holding capacity (WHC), and the salt concentration in the ash. The surface applied ash treatments (SUR, SVCF) have the highest water holding capacities and fairly high pH values, yet they do not have the highest yields. This is likely due to salts contained in the ash that have not leached out of the surface applications. The highest yields were observed in the blended CRF ash treatments. The RP treatment, which had the highest millet yields the previous year maintained fairly high yields although this treatment is somewhat limited by low WHC. The lowest yields where observed in the control plots (CON, UCON). It is interesting to note that some of the ash/refuse blends have higher WHC than the native sandstone derived topsoil (Table 3). Other researchers have also observed increases in water holding capacities of coarse textured materials by adding fly ash (Salter, et al., 1971; Jacobs et al, 1991) Biomass yield is a combination of many factors. In general, these yields were low
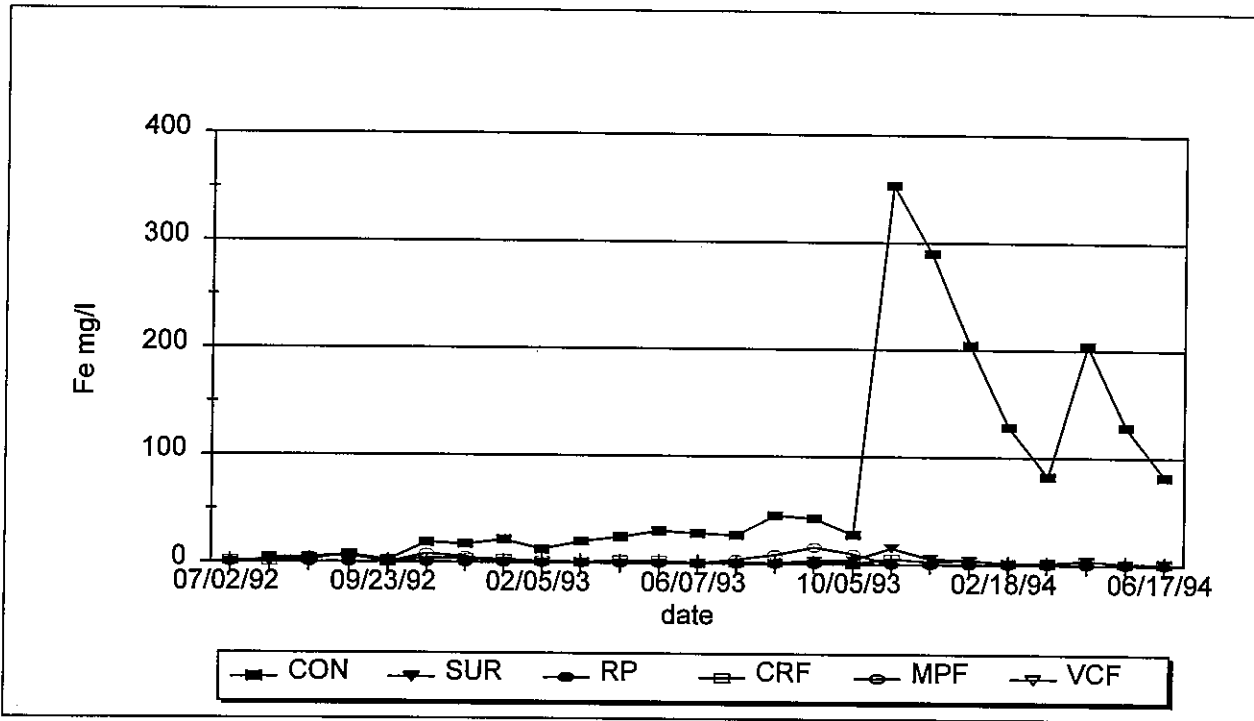
Figure 3. Mean Fe content of lysimeter leachates from July 1992 to September 1994. In some cases a data point may represent a single lysimeter.
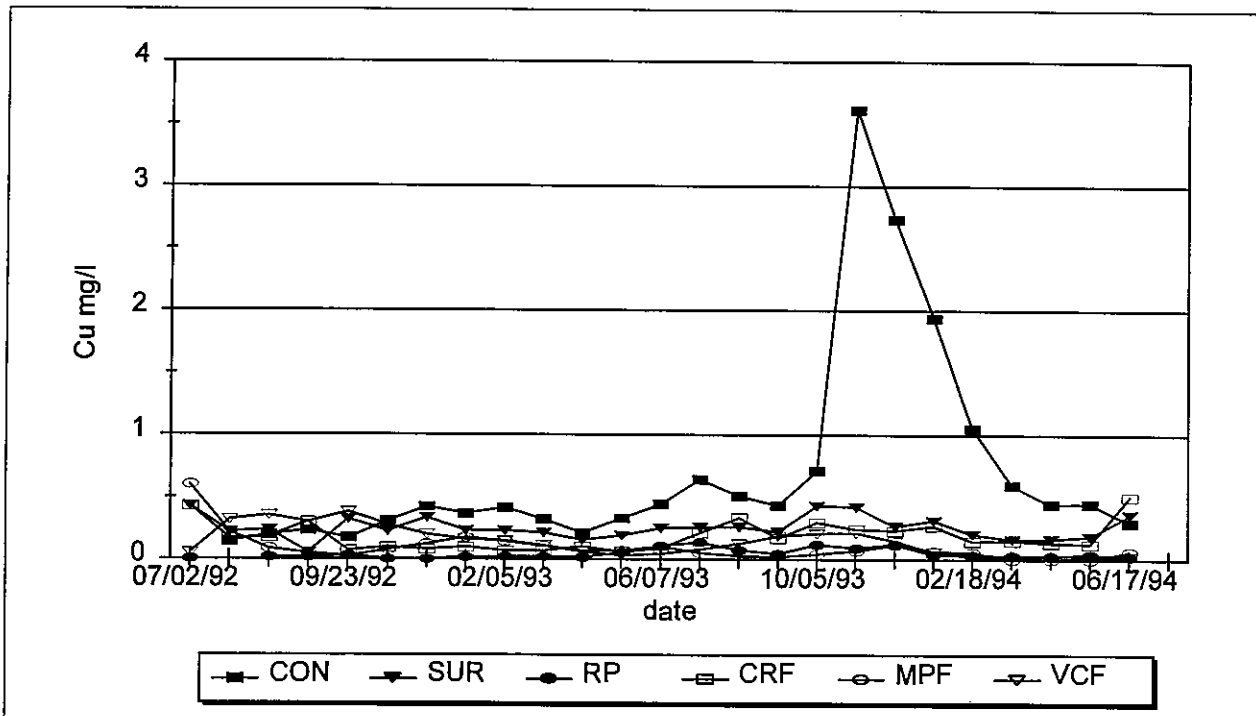

Figure 4. Mean Cu content of lysimeter leachates from July 1992 to September 1994. In some cases a data point may represent a single lysimeter.
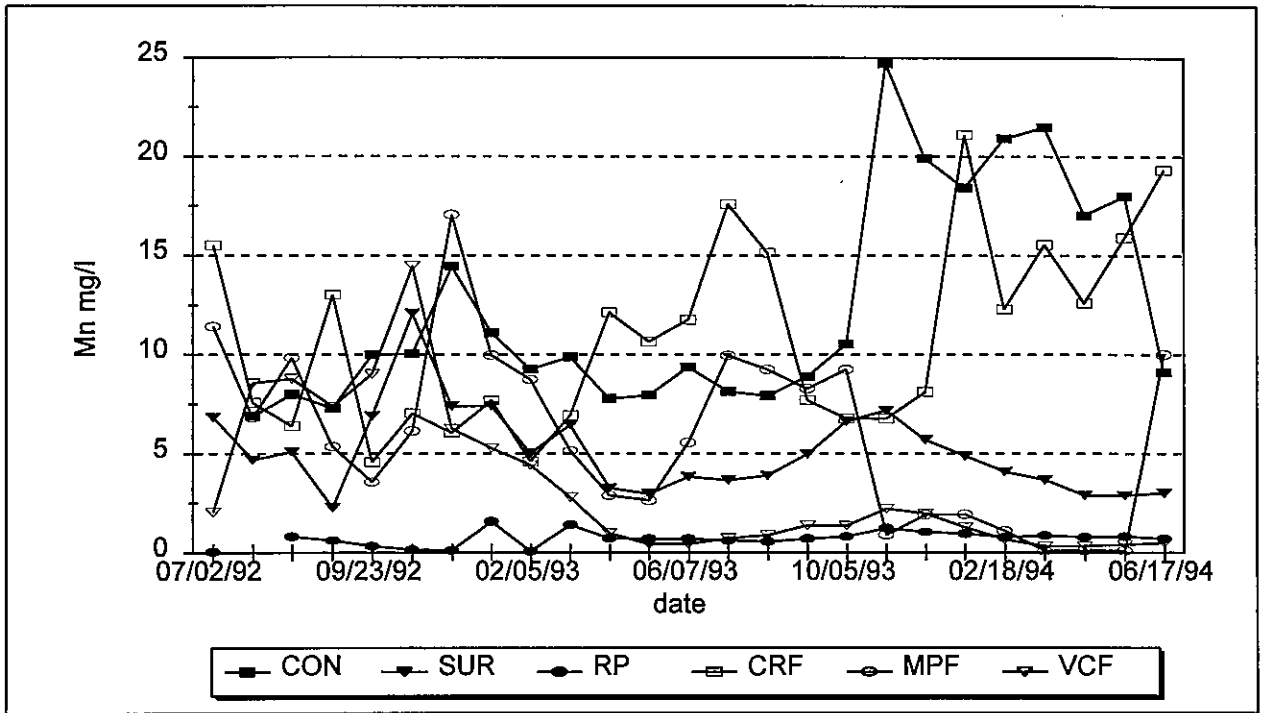
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Figure 5. Mean Mn content of lysimeter leachates from July 1992 to September 1994. In some cases a data point may represent a single lysimeter.
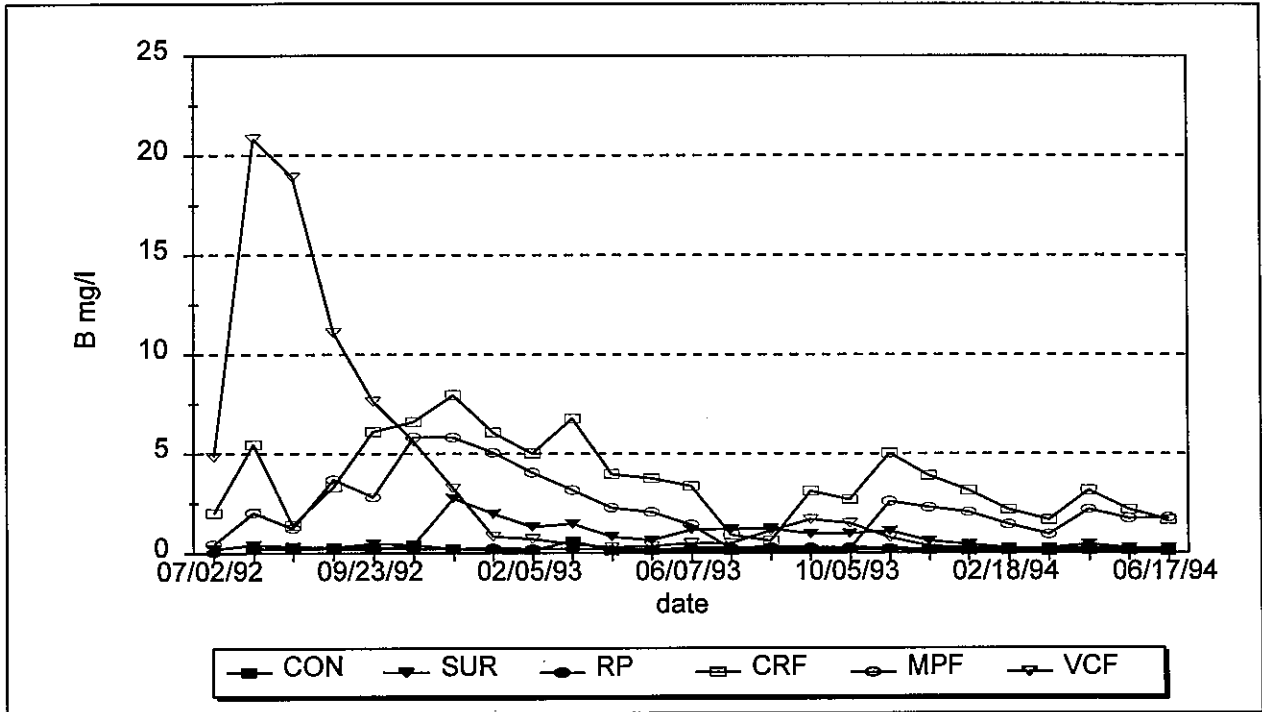

Figure 6. Mean B.conent of lysimeter leachates from July 1992 to September 1994. In some cases a data point may represent a single lysimeter.

and the plant cover would not have met bond release criteria.

Table 2. Mean biomass yields from the Pine Branch plots for 1992, 1993, and 1994. The crop grown in 1992 was millet. The crop in 1993 and 1994 was a mixed grass, legume stand. Values followed by the same letter are not different at the p=0.05 level.

Treatment	1992		1993		1994	
	----------------------------------Mg ha⁻¹----------------------------------					
33% CRF	11.2	a	5.2	a	5.7	abc
20% CRF	7.6	a	5.2	a	6.5	ab
TOP	6.3	ab	2.5	bc	5.2	abc
SUR	2.8	ab	2.6	bc	5.2	abc
SVCF	7.1	ab	4.3	abc	3.6	c
MPF	3.8	ab	3.9	abc	4.8	bc
RP	12.6	a	4.7	ab	7.1	a
CON	7.4	ab	1.9	cd	1.4	d
UCON	1.3	b	1.2	d	1.0	d

Table 3. Soil pH and water holding capacity (WHC) values for the Pine Branch site. Values followed by the same letter are not different at the p=0.05 level.

Treatment	pH		WHC (g H2O kg⁻¹ soil)	
33% CRF	6.1	b	153	bc
20% CRF	5.3	bc	119	cd
TOP	4.9	c	100	d
SUR	7.5	a	232	a
VCF	3.7	d	98	d
SVCF	5.5	bc	273	a
MPF	5.0	c	189	b
RP	5.5	bc	87	d
CON	3.3	e	94	d
UCON	4.1	d	40	e

The biomass yields from the 1994 growing season were higher than the previous growing season for all treatments except the controls (CON,UCON) and the surface applied nonalkaline ash (SVCF) (Table 2). The declines in the control yields may be attributed to the lack of nutrients (mainly N and P) in the refuse. These plots were fertilized and planted more than 2 years ago and these treatments may have run out of nutrients.

Some of the other treatments have produced enough plant growth that nutrients are likely being cycled. The very sparse plant growth on these plots (CON, UCON) have not allowed nutrient cycling to begin. Most of the applied N has likely been lost to leaching and most of the applied P is likely in a plant unavailable Fe or Al phosphate. The low yield of the SVCF treatment is likely due to the high levels of salts in this ash. All plots which received amendments had significantly higher yields than the control plots.

The RP plots had the highest yields indicating that the other treatments may respond to P application. The past summer (1994) was one of the wettest on record in Wise Co., VA. The RP treatment yields indicate that the yields of this treatment the previous year may have been limited by lack of rainfall. The 20% CRF treatment had the highest yields among the ash treatments. There was little difference among the ash amended treatments other than the low yield of the SVCF plots. The SUR plot (surface applied CRF fly ash) had an intermediate yield indicating that the salt levels may be declining in this treatment. The high rainfall and overall good growing conditions in 1994 improved the quality of the vegetation at PB. Most of the blended CRF, RP, and MPF treatments would have met bond release cover criteria.

Conclusions

The ash treatments have had no consistent negative impacts on the leachate water quality with respect to the parameters observed. If the pH drops to < 4.5 elevated levels of Mn and other metals are detected in the leachates of the ash treated lysimeters. Negative water quality effects have been shown from ash/refuse mixtures when ash alkalinity is inadequate to offset the acidity generated from pyrite oxidation (Jackson, 1993; Stewart et al., 1994). Improvements in water quality were observed when there was enough alkalinity in the ash to buffer the pH of the refuse above pH 5.5. The refuse/ash mixtures showed consistent increases in plant biomass production due to increased water holding capacity and increased pH when an alkaline ash was used. The effects of the salts in the refuse seem to decrease with time as leaching takes place. Both the refuse and the fly ash are known to be low in fertility. The use of an organic amendment is likely to be very effective in improving the quality and quantity of vegetation at this site.

Acknowledgement

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INFLUENCE OF TILLAGE AND AMENDMENTS
ON VEGETATION ESTABLISHMENT IN PHOSPHOGYPSUM [1]

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Abstract: Chemical and physical characteristics of phosphogypsum (PG) must be ameliorated for successful vegetation establishment. Therefore, a three-year field study was initiated in 1991 to compare the effects of disking vs. rototilling on nutrient content and dry matter yield (DMY) of browntop millet (Panicum ramosum) and bermudagrass (Cynodon dactylon [L.] Pers.) grown in PG having a pH of 4.82 and an electrical conductivity (EC) of 2.29 dS/m. The PG was amended with 0, 52, 104, 208, and 416 metric tons/hectare (mt/ha) of phosphatic clay (PC), and 0, 29, 58, 116, and 232 mt/ha of sewage sludge (SS). Fertilizers at 112 kg/ha N, 56 kg/ha K, and 28 kg/ha Mg were selectively added in 1991, but were added to all plots in subsequent years. Tillage influenced pH and DMY in 1991 and 1992, but was not influential in 1993. Tillage did not affect EC in any year. Rototilling resulted in greater DMY in the first two years than did disking. Rates within each amendment affected pH, EC, height, DMY, and tissue nutrient concentrations. Sludge application resulted in much higher Zn concentration in bermudagrass tissue, but levels were not sufficient to cause phytotoxicity. In establishing vegetation on PG, both PC and SS have potential as nutrient sources, but no seed germination was observed in SS-amended plots in 1991 and its detrimental effects need to be investigated further. The data suggest that disking, less energy intensive compared to rototilling, would be more economical in initially alleviating the surface hardness problem on PG to establish vegetation.

Additional Key Words: reclamation, phosphogypsum, phosphatic clay, sewage sludge, disking, rototilling.

Introduction

For each metric ton of P_2O_5 produced from phosphate rock, there is co-production of approximately 5 mt of phosphogypsum (PG) as a byproduct. The PG is pumped as a slurry to an impoundment where a dragline is used to move some of the settled PG to raise dike height, resulting in a "pile" or "stack" up to 60 m in height. Currently, stacks in Florida range from 20 ha to 200 ha and occupy over 2,000 ha of land. Chemical processing results in 33 million mt of PG annually (May and Sweeney, 1980), with over 600 million mt already accumulated in stacks. Inherent low-level radioactivity present in PG, mostly from radium-226, has resulted in restrictions on the use of PG (Federal Register, 1992). With this final ruling on PG removal and uses, most PG will remain stockpiled in approximately 20 stacks throughout Florida. When PG stacks cease usefulness as deposition sites, they must be reclaimed in an environmentally acceptable manner.

Establishing a vegetative cover will undoubtedly be part of any reclamation effort. Along with improving aesthetics, vegetative cover should reduce erosion and radon-222 emanation (Richardson, 1994), improve surface runoff quality, and possibly enable use of PG stacks as wildlife habitat. Sparse natural vegetation on PG surfaces indicates that establishment of a vegetative cover is possible. Reasons for sparse vegetation on PG surfaces include residual acidity, nutrient deficiencies or imbalances, low nutrient-holding capacity, and surface hardness.

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