INTERNATIONAL LAND RECLAMATION AND MINE DRAINAGE CONFERENCE

AND

THIRD INTERNATIONAL CONFERENCE ON THE ABATEMENT OF ACIDIC DRAINAGE

Volume 3 of 4: Reclamation and Revegetation

Proceedings of a conference held in Pittsburgh, PA on April 24-29, 1994.
This conference served as the annual meeting for both the American Society for Surface Mining and Reclamation and the Canadian Land Reclamation Association.

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UNIT OF MEASURE ABBREVIATIONS USED IN THESE PROCEEDINGS

Å	angstrom
ac	acre
Bq	becquerel
bu	bushel
cal	calorie
cm	centimeter
cmol	centimole
cps	count per second
°C	degree Celsius
d	day
dS	decisiemen
ft	foot
g	gram
gal	gallon
gpd	gallon per day
gpm	gallon per minute
h	hour
ha	hectare
hL	hectoliter
hp	horse power
Hz	hertz
in	inch
J	joule
K	kelvin
kcal	kilocalorie
kg	kilogram
kHz	kilohertz
km	kilometer
kmt	kilo metric ton
kN	kilonewton
kPa	kilopascal
kV	kilovolt
L	liter
lb	pound
m	meter
M	molar
mA	milliampere
meq	milliequivalent
mg	milligram
Mg	megagram
min	minute
mL	milliliter
mm	millimeter
mmho	millimho
mol	mole
mmol	millimole
MPa	megapascal
mS	millisiemen
mt	metric ton
mV	millivolt
MW	megawatt
μg	microgram
μL	microliter
μm	micrometer
μmho	micromho
μmol	micromole
N	normal
N	newton
nm	nanometer
Pa	pascal
pCi	picocurie
ppb	part per billion
ppm	part per million
ppt	part per thousand
psi	pound per square inch
psig	pound per square inch gauge
r	revolution
rad	radian
rpm	revolutions per minute
s	second
st	short ton
vol %	volume percent
wt %	weight percent
yd	yard
yr	year

INTERNATIONAL LAND RECLAMATION AND MINE DRAINAGE CONFERENCE AND THIRD INTERNATIONAL CONFERENCE ON THE ABATEMENT OF ACIDIC DRAINAGE

Proceedings of a conference serving as the annual meetings of the American Society for Surface Mining and Reclamation, the Canadian Land Reclamation Association, and the West Virginia Surface Mine Drainage Task Force, in conjunction with the Third International Conference on the Abatement of Acidic Drainage.

ABSTRACT

Mine drainage and mine reclamation are topics of major interest to the mining industry, federal and local governments, and the general public. This publication and its companion three volumes are the proceedings of a conference held in Pittsburgh, Pennsylvania, April 24-29, 1994. There were twelve sessions (69 papers) that dealt with mine drainage, including modeling, geochemistry, prediction, treatment, control strategies, characterization, hydrology, and case studies. These sessions comprise volumes 1 and 2. The six sessions (34 papers) that dealt with reclamation and revegetation of disturbed lands are included in volume 3. Volume 4 includes the six sessions (34 papers) that dealt with such topical issues as fires at abandoned mine sites, subsidence, hydrology, mine wastes, and policy. Poster session presentations are represented by 49 papers and 92 abstracts that have been placed in the back of volumes 2, 3, and 4, consistent with the subject of that volume.

Reference to companies and specific products in these papers does not imply endorsement by the Bureau of Mines.

METHODOLOGY FOR MONITORING LAND RECLAMATION OF COAL MINING DUMPS[1]

Concepción Val and Anibal Gil[2]

Abstract: The main objective of reclaiming coal mining dumps is to create a stable and self-sustaining land surface that can, in the long term, be put to some productive use. The relationship that is established between the soil and vegetation is the starting point for the newly created ecosystem to enter into a dynamic evolution. In order to know this evolution, it is necessary to develop a methodology for monitoring systematically the reclaimed surfaces. This monitoring methodology should make it feasible to continuously evaluate the obtained results and serve to clarify the potential uses of the reclaimed lands. This paper explains a monitoring methodology implemented at the mining waste dump at the Puentes Mine in Spain. It consists of the selection of 11 plots on the basis of the time the spoils have been exposed to weathering, the type of reconstructed soils, the reclamation system applied, and the revegetation success. Furthermore, an attempt was made to include every possible situation in the dump. Over a period of 3 years, the evolution of the physicochemical conditions of the reconstructed soils, the soil organisms, the herbaceous species, mycorrhizae, tree species, and vertebrates in these plots were studied. The paper also defines the parameters that need to be controlled within each phase of the study. The results obtained reveal the necessity to place the spoils selectively in the dump, the possibilities offered by the ashes as amendments, and the importance of applying organic fertilizers, seeding herbaceous species as a first phase, selecting tree species, and introducing the vertebrates, soil organisms, and mycorrhizae gradually.

Introduction and Objectives

The subject of this paper is the development of a technique to monitor systematically the evolution of a coal mining waste dump that is in the process of land rehabilitation. The work was conducted from 1990 to 1993 and is part of a research project financed partially by the European Economic Community. At the same time, this project forms part of a global program started by the Empresa Nacional de Electricidad S.A. (ENDESA) at the beginning of the 1980's. The objectives of this global program include recuperating the landscape altered by mining operations and establishing a ground cover that controls erosion as well as the quality of the runoff water and favors the creation of productive land. The results presented in this paper encompass conclusions about the analytical techniques and prospecting methods that should be used to correctly monitor the steps taken in land reclamation, the land reclamation techniques that have been the most appropriate for environmental rehabilitation of the dump, the foreseeable evolution of reclaimed surfaces, and their suitability for different land uses.

The site of the project is the Puentes Mine which is located in the municipality of As Pontes in the province of La Coruña in northwestern Spain. At this mine, there is a combination of circumstances that make land reclamation extremely difficult. The most outstanding of these factors are the extreme acidity and toxicity of the spoils, the scarcity of topsoil due to the time lag between entry into operation of the mining activity and land reclamation work, the high rainfall along with consequent erosion and water contamination, and the absence of methodologies in Europe for the reclamation of surface mined land with these characteristics. Finding a method for the reclamation of surface mined lands with these difficulties is a top priority in Spain.

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[1]Paper presented at the International Land Reclamation and Mine Drainage Conference and the International Conference on the Abatement of Acidic Drainage, Pittsburgh, PA, April 24-29, 1994.

[2]Concepción Val, Biologist, and Anibal Gil, Forest Engineer, Empresa Nacional de Electricidad (ENDESA), Madrid, Spain.

Background

Description of the Exploitation

ENDESA has been exploiting the Puentes Mine since 1976. The brown lignite from this mining operation supplies a nearby 1,400-Mw powerplant. The deposit is approximately 6 km long and 2 km wide. There is a partial narrowing at the center that divides the exploitation area into two fields: west and east. Owing to the characteristics of the deposit, transfer mining, namely, mining in which the spoils are dumped in the area already exploited, has been impossible.

The basement and edges of the coalfield consist of porphyric schists from the Precambrian period. On top of these are materials from the Ordovician period that are separated by a stratigraphic discontinuity. The series is formed by an alternation of phyllites, quartz phyllites, and quartzites, with a clear predominance of the phyllites. On top of these basement rocks is a series of materials from the Tertiary period in erosional unconformity. The bottom of this series consists of a level of plastic clays that are a result of the complete weathering of the phyllites. Above this level, there is basically an alternation in angular unconformity of clays, carbonaceous clays, and lignites with a total thickness of 400 m.

Currently, the minable reserves total 130 Mt, and the rate of extraction is about 12 Mt of lignite annually. As a result, 42 Mm^3 of spoils is excavated and spread annually.

Seven bucket wheel excavators extract the materials, and five pilers deposit the spoils in the dump. The excavated material is carried by conveyor belt to a transfer point, after which the lignite is sent to the powerplant and the spoils to the pilers. The ashes and slag also arrive at this same transfer point. They are then evacuated by dry method and also deposited in the dump. Consequently, the material dumped is made up of tertiary clays intercalated between the seams of minable coal, slates from the edges of the sedimentary basin, which have to be excavated because of the stability of the final slopes of the mine, and the ashes from the powerplant. Furthermore, the fact that all the materials to be dumped come together at the transfer point makes it possible, within the limitations of the mining operation itself, to mix the materials as well as carry out a certain selection of those that will constitute the final surfaces of the dump.

The system for depositing the spoils involves producing final surfaces in the dump at intervals and at an average rate of 60 ha per year. The pilers dump by levels with a height of approximately 20 m, depending on the projected geometry of the dump. Each level consists of a slope with a 20% gradient and a horizontal berm that both constitute the final surfaces of the dump. Once the dump is finished, it will contain 750 Mm^3 of spoils, cover an area of 1,400 ha, and have a maximum height of 200 m.

The mining operation affects several tributaries of the Eume River, which flows in the immediate area, since perimetric channels that flow directly to the waters outside the mining operation have been constructed. Likewise, all the waters from the mine and the dump are canalized to the treatment plant, located near the powerplant, for purification.

Characteristics of the Environment

Surrounding Area

The natural terrain surrounding the mining operation has a curve-backed aspect with elevations between 330 and 540 m. There is a strong anthropic influence. The pasture predominates and is divided into small plots for exploitation, which are demarcated by lined trees and look like a vegetable mosaic. In the areas with steep gradients, shrubs predominate. The different species of *Ulex*, *Cytisus*, and *Erica* predominate. Along the river bank, there is a thickly wooded area with prevalence of *Betula pubescens*, *Salix atrocinera*, and *Alnus glutinosa*.

In addition, there are are small areas with arboreal species characteristic of the region: *Betula pubescens* and *Quercus robur*. Likewise, small reforestations of *Pinus pinaster*, *Pinus insignis*, and *Eucaliptus ssp* can be found.

The geological material of this area is formed mainly by slates and phyllites. The soils developed on top of these materials are characterized by little geochemical evolution due to the high degree of stability of the material. With regard to fertility, these soils have high content of organic matter with an amount of organic carbon between 4.2% and 12.1%, very acidic pH (the most frequent values for pH in water are between 4.1 and 4.6 and never higher than 5.5), predominance of variable charges in the cationic exchange capacity, frequent alic nature (> 60 % saturation of aluminum in the cationic exchange capacity), frequent shortages of Ca and Mg, and low levels of P and K.

Climate

Some of the most outstanding climatic parameters of the area are summarized in table 1.

The extreme monthly thermal amplitude is relatively high throughout the year. The smallest difference is in December (7.5 °C) and the greatest in August (10.9 °C).

The average annual accumulated precipitation is 1,678 mm (table 1). The extreme values were registered in 1951 with a maximum of 2,516 mm and in 1973 with a minimum of 1,178 mm. With regard to its distribution during the year, a marked seasonality has been observed and the following periods are distinguished:

- Period with high precipitation: The rainfall is normally greater than 150 mm per month, and values up to 130 mm/day and 30 mm/h are reached. This period extends from October to March and, as a whole, represents 70% of the average annual accumulated precipitation.
- Period with medium precipitation: This includes September, April, and May and accounts for 20% of the annual total amount.
- Period with low precipitation: During this period, from June to August, the rainfall is less than 100 mm per month and its portion of the average annual accumulated precipitation is 10%.

Table 1. Average monthly values for climatic parameters at the As Pontes weather station.

	Months												
	J	F	M	A	M	J	J	A	S	O	N	D	Annual
T	6.7	6.8	8.5	9.7	12.1	14.9	17.1	17.3	16.1	13.1	9.2	7.3	11.6
MT	10.6	11.2	13.5	14.3	16.8	20.1	22.2	22.7	21.5	18.1	13.6	10.9	16.3
mT	2.7	2.7	3.6	5.1	7.3	10.0	12.1	11.9	10.6	8.2	5.1	3.8	6.9
maxT	16.3	17.9	21.3	22.5	25.7	29.0	31.2	30.8	30.3	25.9	19.4	16.2	23.9
minT	-3.6	-3.7	-2.4	-0.8	1.6	4.6	6.0	6.2	4.2	1.6	-1.1	-3.2	0.9
P	228	188	160	119	112	72	39	51	101	157	200	251	1,678
PET	16	23	47	69	97	108	122	101	58	39	17	15	713
RH	84.3	82.2	79.4	78.5	77.7	77.7	77.7	78.3	79.4	82.3	83.1	84.5	80.0

[1] T=average temperature (°C); MT=average maximum temperature; mT=average minimum temperature; maxT=average absolute maximum temperature; minT=average absolute minimum temperature; P=precipitation (mm); PET=potential evapotranspiration (mm); RH=relative humidity (%).

When the periods with minimum rainfall and the maximum values for evapotranspiration coincide, there is a period with a shortage of humidity that begins in May and extends through September.

With regard to the wind, the prevalent components on a yearly basis are northeast and southwest. In the fall and winter, the southern components (south, southeast, and southwest) prevail and in the summer the eastern components (northeast and east) prevail. The average monthly wind rate ranges from 8 to 11 km/h.

Methodology

This consists of the working method followed for the reclamation of the dump surfaces whose evolution is the object of monitoring, as well as the methodology that has been developed to carry out this monitoring.

Reclamation Methodology

In 1980, ENDESA prepared a Land Reclamation Plan (Val et al. 1989) whose aim was to recuperate the surfaces affected by the mining operation, namely, the dump and the edge of the mine, which total 1,800 ha. Upon completion of the mining operation, a large lake will cover the cavity of the excavated mine. Until 1992, 350 ha of final surfaces were generated.

As a step prior to the undertaking of the specific work of revegetation in these surfaces, a series of criteria concerning the course of action to be followed was adopted. Chronologically, they were:

- Integration of the reclamation activity in the normal development of the mine.
- Awareness of the entire mine work force with regard to this new activity.
- Exploitation of all the possibilities of the deposit and work to improve the quality of the substrate of the final surfaces of the dump.
- Preparation of development projects as well as detailed ones for the dump.
- Project for and the carrying out of the infrastructure works needed (accesses and water evacuation and drainage systems).
- Determination of the uses of the land to be achieved in the different surfaces to be reclaimed, namely, forest land and pastureland.

The specific land rehabilitation stage includes the following:

Study of the Spoils at the Mining Faces. Physicochemical analysis (Guitian and Carballas 1975) is conducted of the seams of spoil in the coal deposit to select the most suitable as topsoil substitute material, on the basis of their agricultural capacity, which are deposited later in the final surfaces of the dump. Analysis is also conducted of the series of slates, that crop out above the coalbed and are excavated due to geotechnical requirements, and the series of spoils from the coal deposit.

Analysis of the Final Dump Surface Materials. There are two reasons for taking a second sample and analyzing it. One is the possibility of the final surface being formed by spoils that have not been selected for this purpose; and the other is to know the real quality of the surfaces to be reclaimed. The number of samples per hectare is 8, with a repetition. The following are analyzed according to the methods described by Peech et al. (1947), Gil et al. (1990), and Urrutia et al. (1992): pH, pH oxidation, total S, cationic exchange capacity (exchange capacity, H+, Al, K, Ca, Mg, Na) and Al saturation. The nutrient elements have not been analyzed, since although the spoils have some nutrients, especially assimilable phosphorus content, they have been fertilized as if they did not have any nutrients.

For orientational purposes, table 2 shows some representative results from the analyses of nonselected spoils from the final surfaces of the dump.

Table 2. Chemical analyses in the final surfaces of the dump.

Samples	pH H_2O	% Total S	pH Oxidation	Exchange Cations (mg/100 gr)					% Al Saturation
				Al	Ca	Mg	Na	K	
1	5.30	0.41	2.76	3.44	1.1	3.17	0.25	0.01	43.22
2	6.25	1.69	2.48	3.66	0.7	2.13	0.16	0.02	54.95
3	5.57	0.40	2.55	0.01	3.9	2.09	0.24	0.03	0.16
4	4.60	1.62	2.50	0.33	0.2	1.38	0.20	0.06	15.21
5	6.01	0.27	3.67	0.10	1.6	1.56	0.03	0.22	2.85
6	6.90	0.31	4.34	1.95	2.4	2.26	0.38	-	27.90

Soil Amendments and Topsoil Worth Replacing. Since it is impossible to replace the layer of topsoil on the entire surface of the dump, calcium carbonate and ashes from the powerplant have been applied as modifiers of the acidity in the spoils. On the basis of the tests performed, the following strategy has been established in general:

pH < 3.5: addition of ashes.
5 > pH > 3.5: addition of calcium carbonate in doses of 5-15 t/ha.
pH > 5: addition of calcium carbonate in doses of < 5 t/ha.

The dose of ash has fluctuated between 500 and 1,000 m^3/ha, depending on the acidity and the type of agricultural work that is foreseen.

As an alternative to the contribution of the soil amendments, the spoils are covered with a layer of topsoil whenever it is available. This layer is 10 to 40 cm thick and sometimes has to be treated with small doses of calcium carbonate (3 t/ha).

Chemical and Organic Fertilizers. Once the acidity of the reconstructed soil is controlled, fertilization through pre- and postseeding fertilizers occurs. Initially, the following rates of fertilizers (fertilizer units) have been applied as a nutrient reservoir prior to seeding:

- N: 50-60 fertilizer units (F.U.)
- P: 150-160 F.U.
- K: 110-120 F.U.

starting from complex 8.24.16.

As a postseeding fertilizer, basically nitrogen has been added with a dose of 40-50 F.U. per hectare, starting from nitramon or with the addition of 15.15.15.

The use of organic fertilizers has had excellent results in the pastureland. The following doses of hen droppings have been applied:

- 30 m^3/ha (in the form of liquid manure)
- 20 t/ha (in the form of solid manure).

Likewise, municipal sewage sludge in doses of 50 t/ha has been applied.

Seeding. Irrespective of the land use that has been determined previously, the seeding of pasture species on all the surfaces occurs with the aim of obtaining a rapid vegetation cover. The mixture of species and doses

depends on the land use assigned to each surface (50 to 100 kg per hectare). The species used are *Holcus lanatus*, *Agrostis tennuis*, *Festuca arundinacea*, *Dactylis glomerata*, *Lolium perenne*,and *Trifolium repens*. Occasionally, the shrubs *Cytisus scoparius* and *Ulex europaeus* are incorporated into the mixtures (Jones et al. 1975).

Plantings. Tree species are introduced in the majority of the dump surfaces, fundamentally on the slopes. The distance between trees is 2 x 2 m in quincunx, and there are 3-m-wide pathways every 8 m of separation according to the level lines.

The following species have been planted: *Betula pubescens*, *Alnus glutinosa*, *Pinus insignis*, *Pinus pinaster*, *Eucaliptus ssp.*, *Salix atrocinerea*, *Robinia pseudoacacia*, *Acacia melanoxilon*, *Pseudotsuga douglasii*. In specific dump surfaces, *Castanea sativa* and *Quercus rubra* have been introduced.

Initially, the leafy species were planted with the root exposed and two saps and the coniferous species with a root ball and one to two saps. As a mineral fertilizer, acid calcium phosphate and potassium sulphate in doses of 80 and 50 g per plant, respectively, were applied.

Maintenance Work. Soil analysis campaigns are conducted during the years following the first agricultural year in order to know the chemical evolution of the reclaimed surfaces. On the basis of these analyses, the corresponding amendments and fertilizers are applied to ensure that the soil surface remains in a stable state and there is revegetation success.

Methodology for Monitoring

The results obtained with the application of this line of work are considered important, especially in light of the initial constraints that come together in the land reclamation of this mine. However, due to the quality of the spoils and the limited experience at a national level concerning land reclamation, it was advisable to conduct a detailed monitoring of the behavior as well as the evolution of the reclaimed surfaces. To do this, 11 plots have been selected on the basis of the time the reconstructed soils have been exposed to weathering, the type of reconstructed soils, the handling and the revegetation success. Furthermore, an attempt was made to include every possible situation existing in the dump.

The methodology to evaluate the results and evolution of each of these plots included the study of:

- Physicochemical conditions of the reconstructed soil
- Soil organisms (invertebrates and microorganisms)
- Herbaceous species
- Mycorrhizae (ectomycorrhizal fungus)
- Forest species (trees)
- Vertebrates

Physicochemical Conditions of the Reconstructed Soil. The following have been studied according to the methods mentioned previously:

- The physical properties of the soil: texture, structure, water-holding capacity, hydric balance, and moisture regime of the soil.
- Mineralogy of the clay fraction.
- Chemical properties and fertility: organic matter and C/N ratio, present and potential acidity, base cationic exchange capacity, phosphorus, and heavy metals.
- Composition of the liquid phase of the soil: pH, electrical conductivity and redox potential, anions, basic cations, Si, Al, Fe, and trace elements (Calvo et al. 1987, Kabata-Pendias and Pendias 1984).

Soil Organisms. With special attention paid to the most important groups with regard to the N, C, and S cycles, the microbial populations have been characterized (Jurgensen 1978). On the other hand, the natural colonization of earthworms under different conditions of pH, texture, organic matter, moisture content, and land reclamation activities has been monitored.

Herbaceous Species. With the aim of knowing the potential capacity for pastureland, the number of cuts per year that the corresponding grass cover is capable of tolerating has been estimated. With each cut this production, in green as well as dry material, has been controlled on the basis of the different doses of postseeding fertilizers: without fertilization, 40 F.U. of N, 60 F.U. of N, 40 F.U. of N + 60 F.U. of P. Additionally, analytical determinations have been performed on the samples of the species that have been seeded. The purpose has been to know the assimilation of the soil elements available to the plants and their suitability for consumption by cattle.

Mycorrhizae. An initial prospection has been conducted in the natural zones near the dump and in the monitored plots in order to know the fungus-tree correspondence. Thus, the most adequate ectomycorrhizal fungi for reproduction for each species has been identified. Furthermore, the real spontaneous colonization produced in the existing tree species in the surfaces already reclaimed has been evaluated on the basis of the time elapsed since their planting (Marx 1980).

Tree Species. There has been monitoring and control of a set number of tree species that were selected on the basis of the different types of reconstructed soils, and the relationship with the degree of mycorrhization produced has been determined.

The following parameters have been studied: survival, height growth, diameter, and diagnosis of deficiencies and diseases (Ashby et al. 1988). The tree species studied include: *Pinus pinaster* and *Pinus radiata*, *Betula pubescens*, *Alnus glutinosa*, and *Castanea sativa*. Furthermore, with some specimens the above-mentioned parameters have been monitored on the basis of the application of the different doses of fertilizers.

Vertebrates. Faunistic inventories and censuses, which have coincided with the breeding seasons for the different species of birds and mammals, have been conducted (Majer 1989).

Results

Physicochemical Conditions of the Reconstructed Soils

The obtained results reflect a wide range of variation in all the analyses among the different monitored plots as well as the samples within each of these plots. This is attributed to the heterogeneity of the spoils in the mining face and the additional variability produced when they are spread in the dump. This heterogeneity makes it very difficult to compare the results obtained in the different monitored plots and limits the use of the average values as representative data.

The parameters that have been identified as essential to manage the different dump surfaces are: pH in water and ph oxidation, percentage of S, C, and N, exchange capacity, and available metals.

Soil Organisms

The figures for bacteria, actinomicetos, and fungi reach normal values in undisturbed soils and are similar to those found in other soils from mining operations. The groups that intervene in the carbon and nitrogen cycles show levels similar to those found in the natural soils of the area in which the absence or scarcity of *Nitrobacter spp.*, *Nitrosomonas spp.* and *Azotobacter spp.*, probably related to a low pH (lower than 6.0), is also habitual.

However, there is a small number of species and specimens of earthworms. Only 131 specimens belonging to 4 species, *Allolobophora caliginosa* (8 specimens), *Dendrobaena madeirensis* (2 specimens), *Dendrobaena octaedra* (120 specimens) and *Lumbricus friendi* (1 specimen), were captured. This reveals that the conditions of the plots are unfavorable to the natural invasion of these species. Only two plots have shown a certain diversity and abundance (92.36% of the total of individuals). The most abundant species throughout the dump is *Dendrobaena octaedra* although it is an epigeal species that lives on organic horizons of the soil.

Herbaceous Species

The amounts produced on the surfaces without topsoil and organic fertilization have been acceptable (30 t/ha/year of green material) even though they have been lower than the optimum production for the pastureland in the region. The relationship between fertilization and production is clearly revealed on these surfaces. The surfaces with topsoil have produced amounts that have been uniform, very good, and close to the optimum for the region (40 t/ha/year of green material). The surfaces with ashes have produced much lower amounts than the optimum for the region, especially when they receive only mineral fertilization (20 t/ha/year of green material). The amounts produced on the surfaces with mixtures of spoils selected from the upper levels of the excavation have been very good and equivalent to those obtained on topsoil.

The heavy metal contents of the vegetation are within the habitual range for species developed in natural soils. The mineral composition varies according to the species considered. The concentration of the macronutrients, K, Ca, P, and Mg, are normal in all the species and plots studied. On the basis of the obtained productions of herbaceous species, the N content is low although it does not limit vegetal growth.

Mycorrhizae

The specific composition of the fungi associated with each tree species is different. Although some species, such as *Paxillus involutus*, are present in all the plots, others are found only in specific tree species. For example, *Suillus luteus* appears only in the plots with *P. radiata* and *Krombholziella scabra* only below *B. celtiberica*. The younger trees show less diversity and abundance of fungi than the older trees. Species such as *Amanita muscaria* or *Lactarius spp.* are representative of a more evolved state of the fungi succession that occurs below *B. celtiberica*. For *P. radiata*, this state is characterized by species such as *Amanita muscaria*, *Laccaria bicolor*, or *Scleroderma citrinum*.

The considerable abundance of fungi associated with the trees studied could be an important piece of information with regard to the explanation of the adaptation and development of the tree species planted in the environment in question.

Tree Species

- *Betula pubescens*: It has been confirmed that it adapts well to the reconstructed soils of the dump including the acidic, poor, and compacted ones. It shows abundant mycorrhization in the substrates with topsoil.
- *Alnus glutinosa*: It has shown good adaptation to any substrate not excessively acidic (pH $\geq$ 4). It tolerates drought conditions when it is already established.
- *Pinus insignis*: Its development in uncompacted materials has been good, although it is very sensitive to diseases that are common in the region. It shows good mycorrhization.
- *Pinus pinaster*: Its initial development has been mediocre, but it is very well adapted in the region.
- *Castanea sativa*: Its development has been good in adequate substrates and locations.

In general, the growth curves for the species studied have a uniform, upward line. This is a clear symptom that the trees are reaching an important degree of maturity and stability in the dump.

The application of high doses of nitrogen in young trees can lead to a high level of mortality if there is continuous rainfall and the N reaches the roots massively .On the other hand, the effect of the application of different doses of fertilizers in big trees is practically not reflected in their development, especially when they are planted on topsoil. Neither is there a clear response to the application of potassium.

Vertebrates

A positive response of the bird communities to the land reclamation work conducted has been clearly revealed at two levels. On the one hand, it is observed that the oldest land reclaimed surface, where there is already a well-developed vegetation cover and structure (trees, shrubs, and herbaceous cover), is the richest in fauna. Furthermore, the evolution in diversity over the 2 years of the study has shown a considerable increase associated with the changes in the vegetal composition and structure, basically due to the presence of species from the *Cytisus sp.* and tree species. The number of vertebrates found in each sampling totaled 42 species (4 amphibian, 1 reptile, 29 birds, and 7 mammals). This figure can be considered equivalent to that found in the forests in the region.

General Conclusions

- The usefulness of systematically monitoring the evolution of surfaces undergoing reclamation in mining operations in which, like this one, there are serious obstacles to land reclamation and limited availability of topsoil, has been confirmed.
- Under these conditions, the key to obtaining the most adequate technicoeconomic results is to focus on the study of the spoils from the excavation and on their selection for disposal on the final surfaces of the dump as topsoil substitute material.
- The addition of basic amendments is essential. Apart from the classical contributions of calcium carbonate, it has been verified that the use of ashes from the combustion of lignite is appropriate for the reclamation of these surfaces, given the results obtained for the quality of the soil as well as the vegetation.
- The contribution of organic fertilization (manures, composts, municipal sewage sludge, etc.) is decisive in these substrates, which are extremely poor and have very low levels of organic matter.
- Seeding all the reconstructed soils with herbaceous species to control erosion is considered very useful as a first phase of revegetation. Furthermore, it speeds up the natural invasion and succession of the other species of the area, fundamentally the shrubs.
- In very adverse soil quality conditions, the principal effort should be directed toward planting tree species, which, once settled down, show an outstanding capacity for survival, since, even naturally, they have good mycorrhization.
- The species that are considered the most suitable for the conditions of these reconstructed soils are *Betula pubescens*, *Alnus glutinosa*, and the *Pinus sp.* conifers existing in the area of the mining operation. The trees should be planted with a root-ball.
- A variety of habitats (trees, scrub, pasture) should be maintained to increase the possibilities of invasion by wildlife. It is extremely important to keep to a minimum the presence of physical barriers that impede the entry of species from neighboring areas that are rich in fauna.
- The following steps taken in representative plots are considered appropriate for conducting the monitoring:
  - Sampling: eight per hectare in heterogeneous substrates and two per hectare on the topsoil.
  - Analysis of the soil samples: pH in water and pH oxidation, percentage of S, C, and N, exchange capacity and available metals once annually.
  - Analysis of the vegetation samples: observation of any deficiencies of metals and nutrients once annually on the same dates (identical physiological state).
  - Study of the soil organisms and mycorrhizae every 2 years.
  - Study of the vertebrate community every 2 years.

A multidisciplinary team should conduct the monitoring, and the participation of universities and research institutes is also considered very useful.

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COMPARISON OF POTENTIAL SOIL PRODUCTIVITY PRIOR TO AND FOLLOWING MINING[1]

Andrew S. Rogowski[2]

Abstract: In most reclamation operations the productivity of the reclaimed land is controlled by the soil material in the topmost portion of the mined profile. By examining the composition, layering, potential behavior, and properties of the profile prior to mining, or by referring to a county soil survey, it should in theory be possible to reconstitute a similar, or improved profile following reclamation. This however, is not always the case, because soil properties vary greatly over area and in time and when disturbed and reconstituted, are not likely to correspond exactly to the properties of the original profile. An alternative approach is examined which attempts to evaluate potential changes in productivity due to mining on a scale of a watershed, or drainage basin in a regional context. The approach utilizes a productivity index model in a GIS format and three regional databases to simulate distributions of the productivity index and biomass on nine contiguous watersheds within a 12,000 sq.km area in Pennsylvania. Simulations show that a regional map of PI can be constructed based on STATSGO data and that GIS overlays reflecting profile properties may be used to monitor progress of reclamation and status of soil biomass productivity.

Additional Key Words Variability, GIS, Geostatistics.

Introduction

Profitable recovery of a deposit is usually the primary goal of a mining operation. However, it is equally important that following mining, the disturbed land be reclaimed to previous or better use. The productivity controlling zone in most mining operations is the soil. The soil consists of a relatively thin upper crust of a regolith formed by weathering of the underlying rock mantle, sedimentary, or alluvial deposit. By examining the composition, layering, behavior and properties of soil prior to mining, it may be possible to predict the potential environmental impact of mining and to develop a defensible reclamation plan.

Even though one purpose of reclamation is successful revegetation of a disturbed area, the relative impact of soil disturbance on biomass production can seldom be fully evaluated. Equipment used to spread topsoil tends to compact soil layers unevenly, increasing bulk density, decreasing aeration and sealing the surface if moisture contents are high. Overabundance of coarse fragments near soil surface, so often the case on reclaimed areas in central Pennsylvania, makes cultivation and seeding difficult. To minimize recharge, operators construct compacted high density layers below the topsoiled zone. Although these layers slow down percolation and can be effective in controlling acid mine drainage, they will also hinder root growth and development. Most importantly however, original soil has developed in place over time in response to climate, hydrologic controls, landscape position and other factors. When disturbed, repositioned and reclaimed, freshly laid topsoil requires time to reach a new state of equilibrium. Because soil properties can vary over area and in time, (Rogowski and Wolf, 1994) examining and sampling of the profile in place at one or more locations prior to mining, or even referring to soil information in a county soil survey does not necessarily guarantee a similarly behaving profile following reclamation.

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[2] Andrew S. Rogowski, Soil Scientist, USDA-ARS-PSWM, 113 Research Office Building, University Park, PA, USA 16802-4709.

To evaluate soil productivity Kiniry et al. (1983) proposed a simple index relating corn growth to soil properties. Pierce et al. (1983) have used it to estimate crop yields while Rogowski and Weinrich (1987) have adapted and modified the original index to estimate relative biomass productivity of different mined and reclaimed areas.

In this study the index is applied in a Geographical Information Systems (GIS) context. Various GIS methods have been applied recently to display, model and simulate environmental impacts of different activities (Burrough 1986). Standard environmental assessments are usually concerned with an impact of a single activity at a point, or over a small area. GIS applications are more regional in scope and can deal with much larger areas. In this study, GIS was used to evaluate potential impact of mining on nine contiguous watersheds within a 12,000 sq.km area in western Pennsylvania. The objective was to show how a simple model, GIS technology, and available data can be combined to evaluate large scale impacts of mining on regional biomass production.

Materials and Methods

Productivity Index Model (PI)

Relative productivity (Rogowski and Weinrich, 1987) of a soil can be written,

$$PI_{ij} = \sum_{i=1}^{m} W_i \prod_{j=1}^{n} x_{ij} \qquad (1)$$

where,

PI_{ij} = productivity index (dimensionless 0 to 1)
Σ = summation operator over i=1,2,...m horizons
Π = product operator over j=1,2,...n factors
W_i = relative root distribution function
x_{ij} = productivity factors (0 to 1), such as available water, bulk density, aeration porosity and pH

The model assumes that biomass production is a function of root growth modified by certain productivity factors, which depend on soil properties. Root growth density is described by a relative root distribution function (between 0 and 1), subsequently corrected by a product of growth limiting factors, such as lack of available water, poor aeration, high bulk density and low pH, (Rogowski and Weinrich, 1987). All growth limiting productivity factors are scaled between 0 and 1 based on potential impact. For example, relative impact of different levels of the productivity factor pH is shown in Fig. 1.

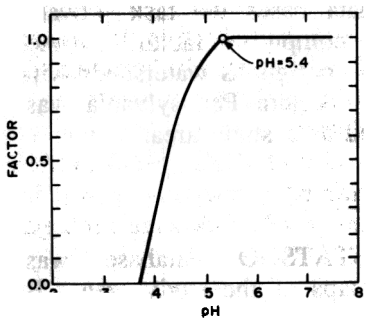

Figure 1. Distribution of the productivity factor pH.

Undisturbed field soil profiles usually consist of several distinct horizons which may be of different thickness. Plant root distributions, and rooting depths may vary. Some plants are shallow rooted, others have roots going down to a considerable depth. The PI model computes the root distribution function based on plant rooting depth and the depth of the soil horizon in the profile. In reconstituted profiles, the topsoil is composed, to a large extent, of the most fertile A-horizon material. In central Pennsylvania the amounts of

A-horizon material are limited and consequently topsoiled profiles are shallow. Therefore both the rooting depth and the depth of the topsoiled layer are the same and the root distribution function, at least initially, may be taken as equal to 1.0 for all reclaimed profiles.

Available Data

To demonstrate the approach three sources of available data were utilized. The major data source was STATSGO, the Soil Conservation Service's 1:250,000 digital soil map of Pennsylvania and the soil interpretations record attribute (GIS term for "property", or "characteristic") data file. Each STATSGO soil unit is composed of different types and percentages of individual mapping units whose properties, descriptions and characteristics are listed in the attribute data file. The STATSGO coverage gives spatial polygon delineations of soil associations and their attribute values identified by a number at a centroid of each polygon. These provided spatial addresses and necessary coordinates for associated attribute values. Continuity and homogeneity of properties within a mapping unit is tacitly assumed. However, at each point within the map there exists a potential for error i.e. the mapping unit, or the percentage composition may not be as listed because of inherent soil variability, (Rogowski and Wolf, 1994).

The second data source was the Pennsylvania State University soil characterization laboratory database system (PSU database), (Ciolkosz and Thurman 1992). The PSU database provides detailed profile descriptions, and attribute information, for over 800 individual soil pedons, which are georeferenced with latitude and longitude identifiers. This data base provides positive identification of mapping units and their locations at a point but lacks information on their distribution and continuity. The data set may also be biased towards modal profiles and easier to reach locations.

The third data source consisted of the 1992-1993 Pennsylvania Soil Test database on land use, pH, acidity, selected nutrients and management recommendations. Land use classification was used to extract spatial locations of "disturbed" land. Disturbed land category, especially in western Pennsylvania, refers primarily to mined and reclaimed areas. The data were summarized by zip code and referenced at the centroid of each zip code zone. By screening zip codes for minimum pH and maximum recommended lime, potential problem areas were targeted. This data base represents user selected field samples extrapolated to the respective zip code areas. Attribute values are correct at a field size scale, but their spatial extent and interpretation of land use may be in error.

Originally the use of data covering much of Pennsylvania was anticipated. Because of the large size of the data bases the task proved beyond the scope of our computing facilities. Accordingly, a subset of nine contiguous watersheds within a 12,000 sq.km area in western Pennsylvania was "windowed out" to be used as a study area.

Computation of PI

The STATSGO database was used to construct PI maps of the study area. Because each STATSGO unit is composed of many individual soil mapping units (Table 1) the PI model was run separately for each of the primary soils comprising a STATSGO map unit. The attribute data file, generally lists data for each soil as a range of values, i.e. pH for the A-horizon of Hazleton soil on 8-15% slope in Table 1 is given as 3.76 to 5.50 pH units. To introduce a degree of variability into STATSGO map, PI was calculated for the low (MIN) and high (MAX) value of each attribute

Table 1. Frequency of occurrence of soils in a STATSGO map unit (PA012) used in weighting.

Soil Name	Slope	Composition
	%	%
Hazleton	8 - 15	14
Hazleton	25 - 60	12
Hazleton	8 - 25	9
Hazleton	15 - 25	5
Cookport	8 - 25	10
Cookport	3 - 8	1
Cookport	0 - 3	9
Cookport	8 - 15	5
Ernest	8 - 25	9
Ernest	3 - 8	5
Wharton	3 - 8	3
Wharton	0 - 3	2
Clymer	8 - 15	6
Udorthents	0 - 50	1

separately and the actual PI assigned to that soil was picked by a random coin toss. The soils, weighted by the frequency of their occurrence were then averaged to give a PI for each STATSGO map unit such as for example PA012 in Table 1.The resulting distributions of the PI were made using ARC-INFO overlay coverage.

Computation of Biomass Productivity

Equation 1 gives relative values of PI based on simulated root distribution modified by soil properties. Rogowski and Weinrich (1987) used Lieth's (1975) Miami Model based on mean annual temperature, or precipitation to estimate biomass productivity. In this study, the temperature portion of the Miami Model was used with PI based on pH, available water, bulk density and aeration porosity to compute potential biomass productivity (P_B).

$$P_B = B \cdot PI_{ij} \tag{2}$$

where,

P_B = biomass productivity (g/sq.m/yr) adjusted for the dimensionless (0-1) PI_{ij}

and

$$B = \frac{3000}{1 + e^{(1.315-0.119T)}} \tag{3}$$

where,

B = biomass productivity level (g/sq.m/yr)

and

T = mean annual soil temperature (°C)

Mean annual soil temperature in the PSU database (Carter and Ciolkosz, 1980) was used in lieu of mean annual air temperature of the original Lieth (1975) model. This was done because in Central Appalachians soil temperatures can differentiate among the soil temperature regimes, which vary with elevation.

Biomass productivity based on STATSGO database and mean annual temperature at Philipsburg, which is located centrally within the subset area, was compared with interpolated values of productivity using soil temperature and pH values from the PSU database. Spatial distribution of PI was evaluated based on the potential inclusion of soil material from other horizons and projected biomass productivity values.

Comparisons

It was anticipated that two productivity limiting factors, pH and available water, would be common to the three data bases and that changes in the distribution of these factors may help identify locations that were undergoing reclamation. Consequently pH, available water and estimated soil temperature values were extracted from the PSU database and the spatial distribution of pH and biomass productivity was interpolated by kriging, (Isaaks and Srivastava, 1989). The distribution of pH was compared with the kriged distribution of minimum pH and maximum recommended lime obtained from the 1992-1993 Soil Test database. In an effort to compare biomass production based on field sampled A-horizon values with profile based estimates, kriged distribution of biomass productivity from the PSU database was compared with the kriged distribution of biomass productivity based on the STATSGO data. The comparisons consisted of subtracting the respective overlays from one another pixel-by-pixel and analyzing the difference relative to the location of disturbed areas.

Results and Discussion

Disturbed Land

Figure 2a shows the disturbed land areas in Pennsylvania based on 1992-1993 Soil Test data land use classification and indicates the boundaries of the subset area. The delineations of disturbed lands are somewhat larger than their actual extent, because each is referenced to a zip code centroid and encompasses the entire zip code area from which soil test samples came. Figure 2a shows that disturbed land is likely to be present through much of the State. Figure 2b shows the "windowed out" subset of the study area in greater detail, including the boundaries of contiguous watersheds.

In western Pennsylvania, the disturbed land consists mainly of stripmined areas, or areas undergoing reclamation. In 1992-93, according to the Soil Test data these areas appeared to be located mainly in the upper half of the study area (Fig.2b). However, particularly elsewhere in the State, disturbed land may also include construction, landfills, and other mining operations, (stone, sand and gravel). This emphasizes the fact that mines are not the only land in need of reclamation and that other areas may need to receive equal attention and scrutiny.

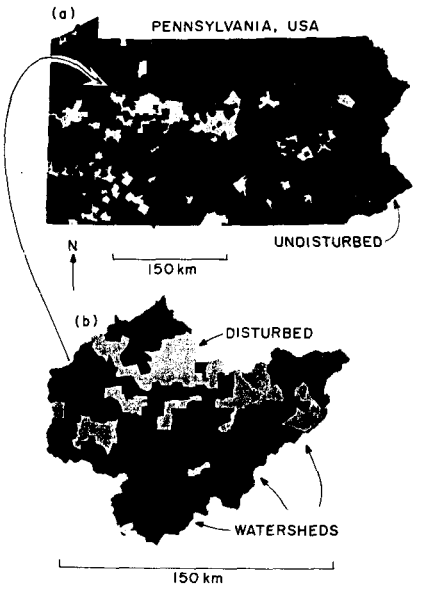

Figure 2. Distribution of disturbed land (a), and location of the windowed out (b) subset used in analysis.

Distribution of the PI

Figure 3 shows the spatial distribution of the PI on the study area and Table 2 gives weighted MIN and MAX values of PI for STATSGO map units and lists contributions by horizon. The PI values are specific to each map unit and the delineations in Fig. 3 will also correspond to soil boundaries. In the light colored areas PI values will generally be <33% of possible values, while dark areas are expected to exceed 66%. Mid-range hues signify areas that are likely to have PI equal to about 50%. Some areas included in this analysis contain old minesoils, or soils that may naturally below in productivity.

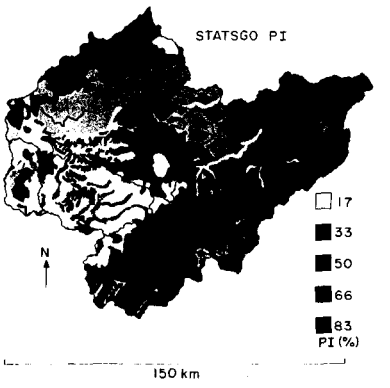

Figure 3. Distribution of the productivity index (PI) on the study area.

A more detailed scrutiny of individual PI calculations (not shown) indicates that differences in PI within the study area result primarily from differences in pH and available water while bulk density and aeration appear adequate, and constant.

Productivity Status

Figure 4 compares the distribution of pH values in the PSU database with distributions of the minimum pH (Fig. 4a) and the maximum lime requirement (Fig. 4b) based on the 1992-93 PSU Soil Test data. All distributions were kriged and scaled to the same number of "equivalent" classes, and the GIS overlays of Soil Test derived values (minimum pH and maximum recommended lime) were subtracted from the PSU derived overlays (pH). The classes were "equivalent" in a sense that the range of values for the three distributions was divided into ten equal increments with numerical values of 1 through 10. Also, for the purpose of this comparison, the lime requirement values were ranked from highest to lowest. Thus the 1st comparison class (Fig. 4) consisted of all locations with the lowest pH and highest lime requirement. All distributions gave well behaved semivariograms (Isaaks and Srivastava, 1989) with approximately a 15 km range of continuity, although structural analysis indicated that the PSU database pH was more variable than Soil Test values.

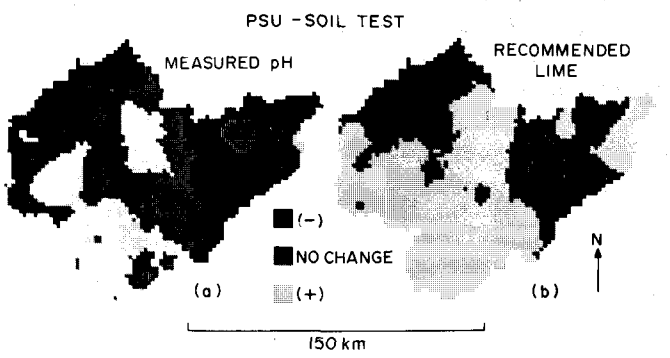

Figure 4. Relative change in pH (a) and lime requirement values (b), between interpolated PSU database values and 1992-93 Soil Test data.

Figure 4 shows locations within the study area where pH and/or lime requirement decreased, increased, or did not significantly differ from that for the original soil. Upper central part of the area designated as (+) in Fig. 4a, which corresponds to disturbed locations in Fig. 2b, may include areas that have undergone reclamation in 1992-93. Similarly, parts of the area (+) in Fig. 4b may also correspond to the area undergoing reclamation. This area is larger in Fig. 4b than in Fig. 4a and may also include soils with low productivity. It should be noted, that changes in pH reflect a comparison of actually measured values, while changes in lime requirement reflect management recommendations. Medium grey areas in Fig 4a and 4b identify locations with no significant change in either observed pH or lime requirement, while black areas (-), where pH of Soil Test data increased and lime requirement decreased, point to areas on which soil productivity has improved. The analysis illustrates a method of monitoring the regional productivity status and potentially, the progress of reclamation operations in time.

Biomass Production

Figure 5 shows differences in the distribution of potential biomass productivity on the study area when an overlay based on STATSGO profile data is subtracted from the A-horizon PSU data based overlay. Black (-) areas indicate locations where profile (STATSGO) derived values were larger, medium grey zones show no change, while light grey areas (+) indicate areas where profile derived values were less than A-horizon derived (PSU data) values.

Figure 5 shows that there was little, or no difference between A-horizon and profile derived biomass productivity over most of the area (medium grey). Areas with negative (black) differences may indicate soils where contribution to PI from deeper parts of the profile may be important. In contrast on areas with positive differences (light grey) suggest sites where contributions to PI from A-horizons appear to predominate. The analysis suggests a method of screening on a regional scale profiles that may be suitable for topsoiling.

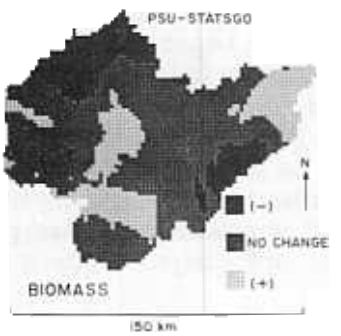

Figure 5. Differences in the distribution of biomass productivity based on the PSU database compared with the STATSGO profile data.

Reclamation Potential

Table 2 shows percent of the area occupied by the soil mapping units, PI values based on MIN and MAX values of input parameters, and the respective contributions (percent) of the A, B and C-horizon materials to the soil PI value. Conventional wisdom holds that if soil horizons other than A are used in the reclamation operation, deterioration of land productivity will ensue. That however, is not always the case. Simulations show that for some map units in Table 2, B-horizon material can contribute much towards the overall productivity of the profile. Since the depth of A-horizons in the study area seldom exceeds 0.30 m and for some soils is between 0.15 and 0.20 m, B-horizon material may at times have to be used in topsoiling. The PI values in Table 2 suggest that for some soils the quality of the B-horizon material may be as good or better than of the topmost layer.

Table 2. Proportion of study area occupied by STATSGO map units, minimum and maximum values of PI, and proportion of PI contributed by different horizons.

Soil Map Unit	PI			Contribution to PI							
	AREA	MIN	MAX	MIN				MAX			
				A	B	B	C	A	B	B	C
					%						
Hazleton/Dekalb/Laidig	23	0.9	52.1	42	50	0	8	42	50	0	8
Gilpin/Wharton/Ernest	16	20.1	70.6	60	15	23	2	56	39	3	2
Gilpin/Cavode/Ernest	15	20.4	70.5	51	25	20	4	59	36	2	3
Gilpin/Weikert/Ernest	14	16.4	70.4	66	1	32	1	63	32	4	2
Hazleton/Cookport	13	17.3	58.7	76	15	8	2	54	40	1	5
Udorthents/Gilpin/Ernest	6	23.0	47.4	36	37	20	6	50	41	3	6
Monongahela/Philo/Atkins	5	39.0	76.9	57	25	16	2	53	36	8	3
Cavode/Brinkerton/Gilpin	4	39.9	68.3	57	21	15	7	57	30	7	6
Average		17.4	63.3	56	25	15	4	53	40	3	4

Conclusions

Productivity index (PI) and biomass production distribution were examined on a 12,000 sq.km area in western Pennsylvania. The area was reasonably typical of the of the coal mining region where stripmining operations necessitate subsequent reclamation. Three separate sources of regional data were used to construct and manipulate regional GIS overlays of PI and biomass production. A regional PI map was constructed, based on STATSGO profile data. Data analysis indicated that low values of PI within the study area were due to low values of soil pH and available water, while both bulk density and aeration porosity appeared adequate. GIS overlays of PI based on differences between 1992-93 Soil Test and PSU data could be used to locate some ongoing reclamation operations and check on progress. The analysis also suggested a method of monitoring the productivity status of soil and the progress of reclamation operations in time. Differences in the distribution of biomass productivity when STATSGO overlay was subtracted from the PSU database overlay, suggested a method of screening potential contributions of soil profile to the productivity. A closer scrutiny of simulations showed that when PI values were calculated separately for different horizons, sometimes the quality of a deeper horizon material was as good or better than that of the topmost layer.

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THE USE OF PRODUCTIVITY INDEX TO PREDICT CORN YIELDS ON RESTORED PRIME FARMLAND[1]

Richard I. Barnhisel and Judith M. Hower[2]

Abstract: A 4-year study was conducted to determine if corn yields could be predicted from a productivity index (PI) derived from physical and chemical properties of soil, including bulk density, cone penetrometer resistance, water-holding capacity, texture, pH, and other soil fertility factors. In our studies, correlation coefficients (r) between PI and yield varied from near 0 to 0.76 from one field or mining method to another. Such a wide range in values of r may be largely attributed to sampling variation over a short distance, whereas corn plants can explore a larger volume of soil thus compensating for poor physical conditions that by chance was collected to represent sampling points within a field. The site from which the highest r value was obtained had been planted to soybeans prior to planting corn. This area did not have the lowest bulk density nor did it have the highest corn yield, but it was the treatment that the model most nearly predicted the yield of corn. Bulk densities varied over time and in general decreased, but the correlation with corn yield also decreased. Corn yields may be achieved (i.e., target levels for bond release) even though PI values may indicate remedial treatments were necessary.

Introduction

The idea of using a productivity index (PI) to predict yield of crops is not new. In recent years, interests have been generated to test this concept of restored prime farmland following surface mining. An index based on the physical and chemical properties of reclaimed land would be useful to both the mining industry and regulatory officials as it could be used to predict the release of Phase III bond soon after Phase I requirements have been met. Phase III bond release is primarily contingent on demonstrating that crop yields have met standards for 3 years. As an evaluation criterion, crop yield is not only dependent on the quality of soil reconstruction, but also on weather conditions and on several options that can be classified under "management practices."

Therefore, it has been proposed here that a productivity index might

1) More accurately evaluate the adequacy of soil reconstruction, and
2) Provide a mechanism that could potentially allow bond release by the fifth year after Phase I bond release, provided adequate ground cover has been established, and reclamation standards other than yield have been met.

If a PI approach could accurately predict crop yields, then coal companies could "prove" that the productivity was returned, without actually growing the crops. On the other hand, if it indicated restoration would not likely be successful, remedial action to correct the problems could be taken earlier in the reclamation process. For "intermediate" situations, the company could either take a chance that Phase III bond will be achieved or "restart the clock" which results when remedial actions are taken. If Phase III bond is not met for these intermediate situations, remedial work such as deep ripping may be required, then bond release will be delayed.

The basic model from which we started was that of Wollenhaupt (1985). Various parameters of the original model were evaluated as well as unmodelled factors. This included topsoil and subsoil fertility (P and K), soil acidity (exchangeable Al), and cone penetrometer resistance. In their selection, consideration was given to incorporating the

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[2]Richard I. Barnhisel, Professor of Agronomy and Geology, and Judith M. Hower, Research Associate, University of Kentucky, Lexington, KY, USA.

fewest number of parameters that are not only predictive of root growth, but would not become cost-prohibitive to the users of model.

The model is being tested for soil profiles and corresponding yields taken from data obtained on the River Queen Mine in Muhlenberg County in western Kentucky. Several additional sites were evaluated in other areas of Kentucky and Indiana, but due to manuscript length restrictions, these data will not be reported here.

Literature Review

Productivity Indices

Over the past 60 years in the United States, a variety of quantitative expressions have been proposed to compute a soil productivity index from soil characteristics. Indices have been developed in response to specific needs in agricultural land use planning and in the equalization of land values and tax assessments (Huddleston 1984).

In 1979, Neill proposed an index such that yield was assumed to be a function of root growth, which in turn, was a function of soil environment (fig. 1). Her model was designed to evaluated soil horizons within soil profiles in terms of their potential available water storage capacities, aeration, bulk densities, pH's, and electrical conductivity's and the sufficiency level of each horizon to promote root development. The product of these five sufficiency factors was then multiplied by a depth-dependent root weighting factor for each horizon based on an ideal corn root development model. The sum of these products over a given soil profile was defined as the productivity index.

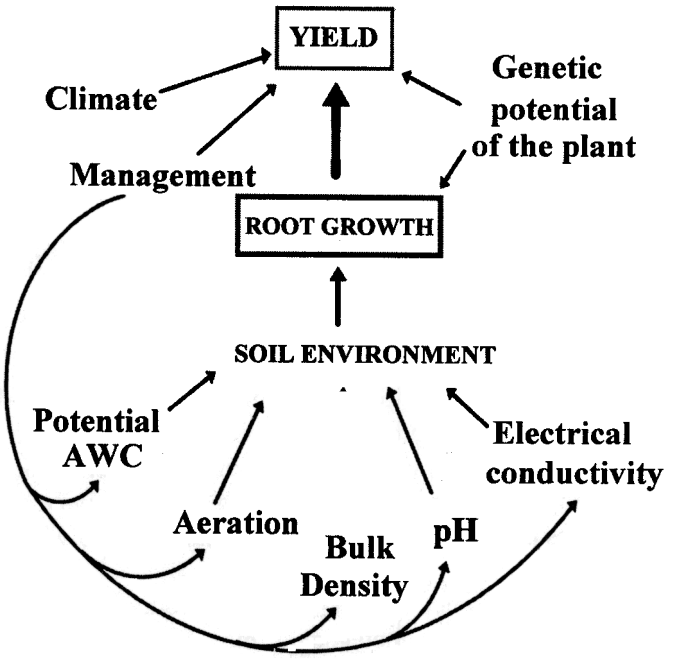

Figure 1. Conceptual PI model after Kiniry et al. (1983)

Pearce et al. (1983) built from the concept proposed by Neill (1979) and streamlined it to consider only three major soil characteristics: bulk density, available water, and pH. Bulk density sufficiency was redefined to reflect various soil texture classes. While Neill assumed an ideal rooting depth for corn of 200 cm, Pearce et al. (1983) used a critical soil rooting depth of 100 cm. Both models assumed that nutrients are not limiting to plant growth.

In 1985, Wollenhaupt made further refinements and made adjustments to the sufficiency equations. He normalized the root distribution curve between the upper and lower horizon boundaries, solved the integral for centimeter increments, and found that relationship approached a power function.

Soil Factors Introduced in This Study

The soil characteristics of P and K concentrations, total acidity, and cone penetrometer resistance were introduced as sufficiency factors in the model developed herein because these factors may vary independently from physical properties, and thus we did not assume they have an index of 1. Subsoil fertility was considered essential to adequate root growth and beyond the control of surface management techniques. The model allows cone penetrometer data to be used as a substitute of bulk density.

Total acidity, especially exchangeable Al, of the subsoil has been demonstrated to be a significant factor that can potentially impair crop yields (McKenzie and Nyborg 1984). Acidity can be a consideration in soils and overburden materials associated with coal surface mining in the Midwest. Another parameter we are considering in place of, or along with exchangeable Al is pH measured in a 1:1 M KCl to soil suspension. In Wollenhaupt's 1985 model, the pH was measured in dilute $CaCl_2$ (Hoyt and Nyborg 1972) or it was estimated from the pH measured in water by subtracting a constant (0.5).

Newly constructed prime farmland soils commonly exhibit excessive soil strength and an absence of continuous macropore networks. Consequently, root growth is often severely inhibited because provisions for water movement, aeration, and root system extension are lacking (Hooks and Jansen 1986).

Data presented by Thompson et al. (1987) from three mine soils, indicate that within each soil horizon, as cone penetrometer resistance increases, average root density decreases. This was a significantly (negative) correlation between the average root length density of the 67-88 cm and 89-110 cm depth segments to the cone penetrometer resistance over the 23-110 cm depth averaged across soil treatments. This effect appears to be most profoundly noted for soils at the Captain Mine in Perry County, Illinois on fields where soil reconstruction was achieved using the mining wheel-conveyor spreader system.

Methods

Soil Sampling

A tractor-mounted Giddings coring machine was used to collect soil samples from various field locations. In general, two samples, approximately 1-foot apart, were taken at each location; all cores were at least 30 in. (76 cm) in length. One core was used to independently evaluate bulk density; the other core was used for other physical and chemical properties. In general, sampling locations were on a 50- by 50-ft (15.2- by 15.2-m) grid. The number of sampling sites per field varied, but always exceeded 25 within an individual data set.

Cone penetrometer resistance was introduced into the PI model as an alternative to bulk density. Since bulk density or cone penetrometer data may not always be available, a working relationship between the two measurements was proposed to increase the application of the PI model to a wider set of conditions. Bulk density data may be obtained over a wider range of soil moisture conditions, whereas cone penetrometer resistance data are most accurate when the moisture is near field capacity. Both penetrometer resistance and bulk density have been demonstrated as adequate predictors of root system performance, especially in the deeper regions of the root zone (Thompson et al. 1987).

Soil Testing

Soil tests for available plant nutrients were performed by the Department of Agronomy and the Soil Testing Laboratory of the University of Kentucky. Bulk density was determined using a procedure modified from the core method described by Blake and Hartge (1986). Water and KCl salt pH's were determined using a glass electrode (Van Lierop 1990, Eckert 1988, and the Council on Soil Testing and Plant Analysis 1980). Exchangeable Al was determined employing the titration method of Yuan (1959). Available P and K were determined by the Mehlich III extraction (Tucker 1988a, Tucker 1988b). Sufficiencies for P and K were based on data and recommendations of Semalulu 1992, and University of Kentucky's AGR-1 Lime and Fertilizer Recommendations 1990.

Percent sand, silt, and clay were determined by the pipette method of Gee and Bauder (1986). In the PI model, texture is used in a subroutine to "weight" the contribution of each particular depth increment. It also was used with bulk density to calculate hydraulic conductivity. Water-holding capacity was determined using a standard

method developed by Klute (1986). Water-holding capacities were calculated from moisture values obtained at field capacity and wilting point.

A continuous-recording cone penetrometer was used on a tractor-mounted Giddings coring machine at a constant rate of 2.9 cm/s. A 30° right-circular cone, 6.45 cm^2 (1 in^2) in cross-sectional area, was employed after the methods of Hooks and Jansen (1986). Measurements were recorded on a system developed in Illinois, similar to that described by Hooks et al. (1993). Replicated measurements at each sample location were taken. Areas under the recorded curves were integrated as representative of pounds per square inch of resistance.

Experimental Sites

Several sites have been used to collect data for the PI model. We chose sites to represent soils reclaimed with scraper pans and/or end-dump trucks. These sites also had a range of reclamation practices, including different soil depths, soil chemical and physical properties, time following soil replacement, ripped versus nonripped, cropping sequence or crop rotation, and test crops. However, since corn is required as one of the test crops in Midwestern States, corn yield measurements may have been separated in time, and perhaps space, for some of these locations.

PI Model Development

The PI model tested is a sum of products. Each soil test value is assigned a sufficiency factor ranging between 0.1[3] and 1.0, which indicates adequacy to promote root growth. The product of these sufficiencies for each soil layer in a profile is multiplied by a root weighting factor, which is a function of soil depth. The PI for a single soil profile is the sum of the PI's for the soil layers that make up that profile. Conceptually, the model is depicted in fig. 2.

The equation for the conceptual model is

$$PI = \sum_{i=1}^{i=n} [BD_i \cdot AWC_i \cdot pH_i \cdot EC_i \cdot P_i \cdot K_i \cdot EA_i \cdot CPR_i \cdot WF_i]$$

where

- n = number of layers in the soil profile,
- BD_i = bulk density sufficiency for the ith soil layer,
- AWC_i = available water holding capacity sufficiency for the ith soil layer,
- EC_i = electrical conductivity sufficiency for the ith soil layer,
- P_i = phosphorus sufficiency for the ith soil layer,
- K_i = potassium sufficiency for the ith soil layer,
- EA_i = exchangeable aluminum sufficiency for the ith soil layer,
- CPR_i = cone penetrometer resistance sufficiency for the ith soil layer,
- WF_i = root weighting factor for the ith soil layer.

Sufficiency equations[4] developed by previous workers were adopted for: bulk density (Wollenhaupt, 1985), available water (Kiniry et al. 1983, Wollenhaupt 1985), pH (Pearce et al. 1983, Wollenhaupt 1985), and electrical conductivity (Kiniry et al. 1983, Wollenhaupt 1985). For this application, electrical conductivity and hydraulic

[3] Since the model is still undergoing refinement, the lower limit has not been fixer at this time. It is likely to reside between 0 and 0.1.

[4] These equations are imbedded into the computer program and affect the final PI value as various limits are meet. This program is being developed using a spreadsheet format. Copies should be available after the paper is given orally by writing the senior author.

conductivity (in relation to its effect on bulk density sufficiencies as calculated in the model) were assumed to be nonlimiting in their effect on root growth. The above model should be considered as a "working model"; some changes have occurred since we first introduced it (Barnhisel et al., 1992). The rationales for changes in the various equations that are used in the subroutines of the working model are presented there. The model is likely to be revised further when the remaining soil samples have been processed.

Results and Discussions

Productivity indices generated by the model were tested for their predictive capability by determining their correlation coefficients with corn yield at both the 5% and 10 % confidence limits. Erratic results have been obtained for such correlations. For some sites, medium to high r values have been obtained, whereas for other sites, r is essentially 0. Data will be presented from only two of the sites used for model verification, one in which poor results were obtained and one in which some treatments gave very good results.

A relatively large data set has been analyzed for the "Prime-on-the-Gob" site on the River Queen Mine. At this site, 30 locations for samplings of all parameters were established in a 5 by 6 matrix with 50-ft (15.25-m) a spacing between rows. Two approaches have been used, one using bulk density data, the other using cone penetrometer values. Calculated PI values and the corresponding corn yield for both approaches are given in table 1. In general, the overall size of the PI value was higher when the bulk density was used in the model than when cone penetrometer data was used. Correlation coefficients between the PI's computed using both bulk density and cone penetrometer data with their respective yield for this data set have been very disappointing and were 0.05 and 0.06, respectively. These poor correlations occurred even though there was a relatively wide range in yields and PI values occurred. It is likely that improvements in the ability of the model to predict the PI when cone penetrometer data are used PI can be made by incorporating changes in the sufficiency component of the equation, but at the present time, rational for making such changes has not been developed.

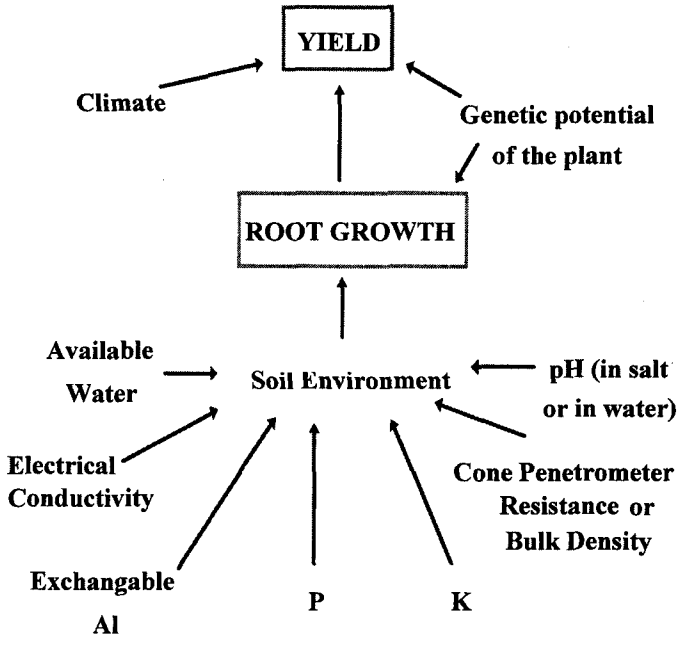

Figure 2. Revised conceptual model after Barnhisel et al. 1993.

At a second location on the River Queen Mine, PI values were calculated using only the model with the bulk density term for four cropping and compaction removal treatments. At this site, strips of 100 ft (30.5 m) wide by 450 ft (137 m) were initially established in various crops, including soybeans, corn, black locust, and alfalfa. In addition, each strip was divided into four blocks that were ripped under three moisture conditions with one of these subplots being a nonripped control. After two years of the initial crops, the entire area was planted to corn. Correlation coefficients between corn yields from these plots and PI ranged from 0.06 to 0.76, and these data are shown in table 2.

The r values between corn yield and PI were highly correlated for plots that had been previously planted to soybeans, intermediate for black locust and alfalfa, and poor, if any, correlation between corn yield and PI for the plots that had been planted to continuous corn through 1989. The PI data given in table 2 were calculated for two sets of bulk density measurements, one taken in 1984 at the beginning of this study and the one in 1991. In all cases except for the continuous corn plots, the correlation coefficients decreased when PI was regressed against the 1991

bulk density data. The corn yields regressed against the two PI's were collected in 1987 midway between the times the bulk density data were collected. Weather can affect corn yields from one year to the next. However, correlations between corn yields made in 1988 versus the two PI data sets, corresponding to bulk densities collected in 1984 and 1991, had about the same range 0.07 to 0.79 (data not shown).

Table 1. River Queen Prime-on-the Gob corn yield and their respective PI's using cone penetrometer versus bulk density data.

Corn Yield, bu/a	Soil PI[5]	Soil PI[6]
108	0.557	0.653
116	0 223	0.346
118	0.258	0.375
120	0.192	0.422
128	0.427	0.530
131	0.466	0.630
132	0.475	0.672
134	0.233	0.573
138	0.279	0.550
140	0.230	0.544
140	0.300	0.476
145	0.247	0.373
147	0.411	0.532
150	0.233	0.567
150	0.222	0.458
151	0.398	0.543
152	0.189	0.352
153	0.243	0.437
154	0.269	0.439
156	0.446	0.465
160	0.235	0.462
162	0.235	0.362
165	0.278	0.499
168	0.480	0.568
170	0.237	0.396
170	0.409	0.495
180	0.324	0.446
Mean 146	0.305	0.488

[5] PI computed using the modified model and cone penetrometer data.

[6] PI computed using the modified model and bulk density data.

Table 2. Correlation coefficients (r) between PI and corn yield, River Queen Mine—Coleman Area.

	--------- Initial crop ----------			
Year[7]	Soybean	Corn	Alfalfa	Black Locust
1984	0.76	0.06	0.50	0.59
1991	0.70	0.14	0.13	0.31

[7] Year in which bulk density measurements were made.

It may not be much of a surprise that plots having been in continuous corn had the lowest correlation between PI and yield, but there were areas within these plots that also had good yields (data not shown). The predictive capability of the model increased, especially when the 1984 bulk density data were used, and when a legume had been grown the first 2 years.

For some unknown reason(s), the model based on the 1991 bulk density values had much lower r values, especially for alfalfa, as compared with the r value based on the 1984 bulk density. This poorer correlation was in spite of a slight decrease in the overall bulk density between the two years. A relatively small change in the negative direction occurred for soybeans.

Conclusions

Although some of the PI measurements were highly correlated with corn yields at the River Queen—Coleman site, it was poorly correlated at the other site on this same mine ("Prime-on-the Gob"). Even for the Coleman site, correlations between corn yield and PI ranged from 0.06 to 0.79 depending on the the cropping sequence, year the corn yield was collected, and the year the bulk density was measured.

We are hopeful that by adjusting the components within the PI equation, better and consistent correlations can be obtained. If this is not possible, then using PI to determine if Phase III bond release criteria for the return of the surface-mined land to it's original productivity have been meet by measuring soil properties will not be

recommended. This is in spite of having met these yield criteria by actually growing various crops at all locations tested.

Acknowledgment

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IMPROVING LAND RESTORATION IN BRITAIN BY BETTER TIMING OF SOIL MOVEMENTS[1]

Malcolm J. Reeve[2]

Abstract: The soil moisture content at plastic limit is shown to be a suitable threshold below which there is likely to be less compaction and damage to soil structure when soils are being moved on surface mine and quarry sites. However, in wetter parts of England and Wales use of this threshold will result in a very short stripping season for stripping upper layers of soil and, in many years, deeper subsoil layers may never dry out sufficiently to resist damage. Direct use of plastic limit to control earthmoving involves continual on-site checking of soil moisture conditions, but it is feasible to predict suitable soil conditions from a remote location using models of soil moisture deficit. Maps of long-term moisture deficit can be used to define no-go periods and optimum soil stripping seasons, while real-time use of moisture deficit data in conjunction with rainfall measurements can aid the scheduling of site operations.

Additional Key Words: soil handling, plastic limit, soil strength.

Introduction

Effective revegetation of mining and quarrying sites is dependent upon the replacement of soils in the best possible physical condition. They are more likely to be in good condition if they have been stripped and handled in good condition. When soils are wet or very moist they are weak; handling inevitably results in smearing, compaction and loss of structure (i.e. deterioration of physical condition). As soils dry they develop greater strength and are more resistant to physical deterioration.

In England and Wales some 2,000 ha of land were restored each year of the 1980's after sand and gravel or clay workings and 700 ha after opencast coal mining. In total more than 20,000 ha of mineral workings were reclaimed between 1982 and 1988 (DoE 1991). Much of this land was in agricultural use prior to working and 90% was returned to a "green" afteruse (mainly for agriculture or amenity) which required careful attention to soil restoration. To achieve this, it was important that the most suitable soils were retained for restoration, that the most appropriate machinery was used to strip, store and respread them, and lastly (and perhaps most importantly) that soils were handled only when they were sufficiently dry to resist damage through compaction and smearing. Continuing examples of poor restoration show that these key criteria are not always being met, and this paper presents the results of research into improving the timing of soil handling.

Part of the work reported herein was carried out when the author was a senior research scientist with the Soil Survey and Land Research Centre.

Timing of Soil Handling

An early attempt at controlling the timing of soil handling was used in a series of agricultural land restoration experiments carried out in eastern England in the late 1970's (DoE 1982). Separate rules on soil movement, based on preceding rainfall, were given for soil stripping and direct reinstatement, reinstatement from storage heaps and restarting work after rain. An example is given in table 1.

There was a general proviso that soil-moving operations should normally only take place between May and October. However, as McRae (1989) points out, restricting soil handling to ordinarily dry times of the year takes no account of abnormally wet summers. Also, rainfall on its own is not a sufficient control as soil moisture

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[2]Malcolm J. Reeve, Principal, Land Research Associates, 153 Swarkestone Road, Derby DE73 1UD, England.

Table 1. Impact of rainfall on stripping and direct reinstatement by earthscaper (after DoE 1982).

Rainfall, mm	For rainfall reading taken at: 0730 h	1800 h
0-1	May be ignored	May be ignored
1.1-3	Delay operations to midday	May be ignored
3.1-5	Delay to next day	Delay to midday next day
5.1-7.5	Delay to midday next day	Delay to day after next
7.6-10	Delay for 48 hours	Delay for 48 hours
>10	Delay for 3 days	Delay for 3 days

contents will vary with rainfall and evapotranspiration. Thus, rainfall in summer will normally be less restrictive than rainfall in early spring.

Because of the shortfalls of this system, criteria based on soil conditions were introduced during the 1980's and applied by local government through legal conditions attached to permissions to work minerals. Some site-specific planning conditions referred to the soil moisture content at plastic limit, either indirectly or directly. For example *"the stripping and movement of topsoil and subsoil shall only be carried out when the material to be moved is in a dry (and friable) condition"* (1991 planning condition for opencast coal mining in north-east England) and *"soil handling, cultivations and trafficking..... shall only take place when the moisture content of the soil is 5% or more below the lower plastic limit"* (1989 planning condition for sand quarrying in south-east England).

The use of plastic limit has some merits because it reflects the reaction of soil to specific forces. For example, above the plastic limit the compressibility of a soil increases rapidly and smearing occurs (Boekel 1979). The plastic limit has been reported as corresponding to a shear strength of about 170 kPa (Wood and Wroth 1978). This implies that at moisture contents near and below the plastic limit a soil will exhibit sufficient strength to resist the force exerted by the wheels of an agricultural tractor, bulldozer or small dumper truck, but may not have sufficient strength to bear the larger vehicles used for earthmoving without some deformation (table 2).

Table 2. Static contact pressures exerted on soil by earthmoving and cultivation equipment.

Vehicle	Maximum static tire/ soil normal stress, kPa[1]	Reference
Wheeled agricultural tractor	54-95	Blackwell et al. 1985.
Medium tracked-bulldozer	45-70	Manufacturer's literature.
Low-ground-pressure bulldozer	30-45	Manufacturer's literature.
23-mt 3-axle dumper truck	[2]145-170	Manufacturer's literature.
40-mt 2-axle dumper truck	[2]220-250	Manufacturer's literature.
330-HP wheeled tractor-scraper	[2]160-185	Manufacturer's literature.
550-HP wheeled tractor-scraper	[2]260-300	Manufacturer's literature.

[1]Dynamic stresses can be much larger than static stresses as a result of slewing (tracked vehicles) and bouncing (wheeled vehicles).

[2]Calculated as 50% to 60% of tire/hard surface stress.

Exclusive use of the plastic limit (or a moisture content related to it) to control soil movements on mineral sites has caused problems, however. Subsoils on clayey substrates or in wet parts of England and Wales may never dry sufficiently to reach the plastic limit, in effect precluding continued working of the site or, more likely, resulting in a default of the planning condition because it is too stringent.

For this reason a research program was undertaken to study changes in soil moisture and plasticity in specific soils over an 18-month period. The aims were to see whether it is realistic in practice to use a plastic limit-related threshold to control earthmoving, to investigate the use of meteorological soil moisture deficit data as a partial control of earthmoving, and to develop a Code of Practice that would ensure that a consistent and practicable approach is adopted in future permissions to work minerals.

Site Monitoring

A range of agricultural sites were monitored for a 15-month period from July 1990 to January 1992. Soils ranged from fluvic ustochrepts and haplaquepts (typical of sites from which alluvial gravels are quarried) to ochric and mollic haplaquepts on boulder clay (typical of open pit coal mining areas). Soils were divided into three main textural categories:

- Sandy loams and sandy clay loams (12% to 24% clay in topsoils, 10% to 23% in subsoils)
- Clay loams and silty clay loams (26% to 35% clay in topsoils, 26% to 37% in subsoils)
- Clays (32% to 49% clay in topsoils, 41% to 51% in subsoils)

In each category 25 m x 25 m monitoring plots were established under grassland and replicated between three climatic regions defined by the magnitude of the average annual potential soil moisture deficit. Some of the soil and climate combinations were replicated under a wheat crop or left unvegetated. Moisture contents and shear strengths were measured weekly (twice weekly during the spring period of rapid drying) at 10-cm intervals to a depth of 0.5 m and at depths of 0.75 and 1 m. The measurement method involved augering a 6-cm diameter vertical hole in stages, measuring shear strength with a vane tester 5 cm below the base of the hole, and then taking a loose soil sample by soil auger from the same zone. Moisture contents of the samples were determined gravimetrically in the laboratory. On any visit, measurements were taken from three auger holes distributed across each plot and the results for any depth averaged. The soils of each site were fully characterized physically, including analysis of the bulk density and plastic limit moisture content at each monitoring depth.

Results

Soil Strength and Plastic Limit

Some results of this study are presented by Reeve and Davies (in press) and show that the plastic limit of remoulded soil is an approximation to an undisturbed shear strength of 100 to 200 kPa, dense soils having a greater strength than others. This range can be compared with the equipment ground pressures in table 2 and suggests that at the plastic limit careful use of the lighter items of earthmoving equipment will result in little damage to soils.

Opportunities for Soil Stripping Using a Plastic Limit Threshold

The opportunities that are provided by the use of a plastic limit threshold for soil stripping can be seen in figure 1. In this, data are presented for a zone in the upper subsoil (Bw or Bg horizon at 40-cm depth) in each soil. This is a soil layer that often has a better structure than layers below but one that is very vulnerable to damage while topsoils are being stripped from above it. The main year of the study, 1991, was unremarkable climatically but had a drier than average autumn. Figure 1 shows that a long soil-stripping period would be obtained by using the plastic limit as a threshold in the drier parts of eastern England (right column of graphs). Stripping of loamy soils could continue from early spring to late autumn and of clayey soils from June to late autumn (with the exception of a wet spell in early July). In any texture, however, the stripping season shortens markedly from the drier climatic zones to the moist ones (right to left in figure 1). In the moister areas of western and northern England an

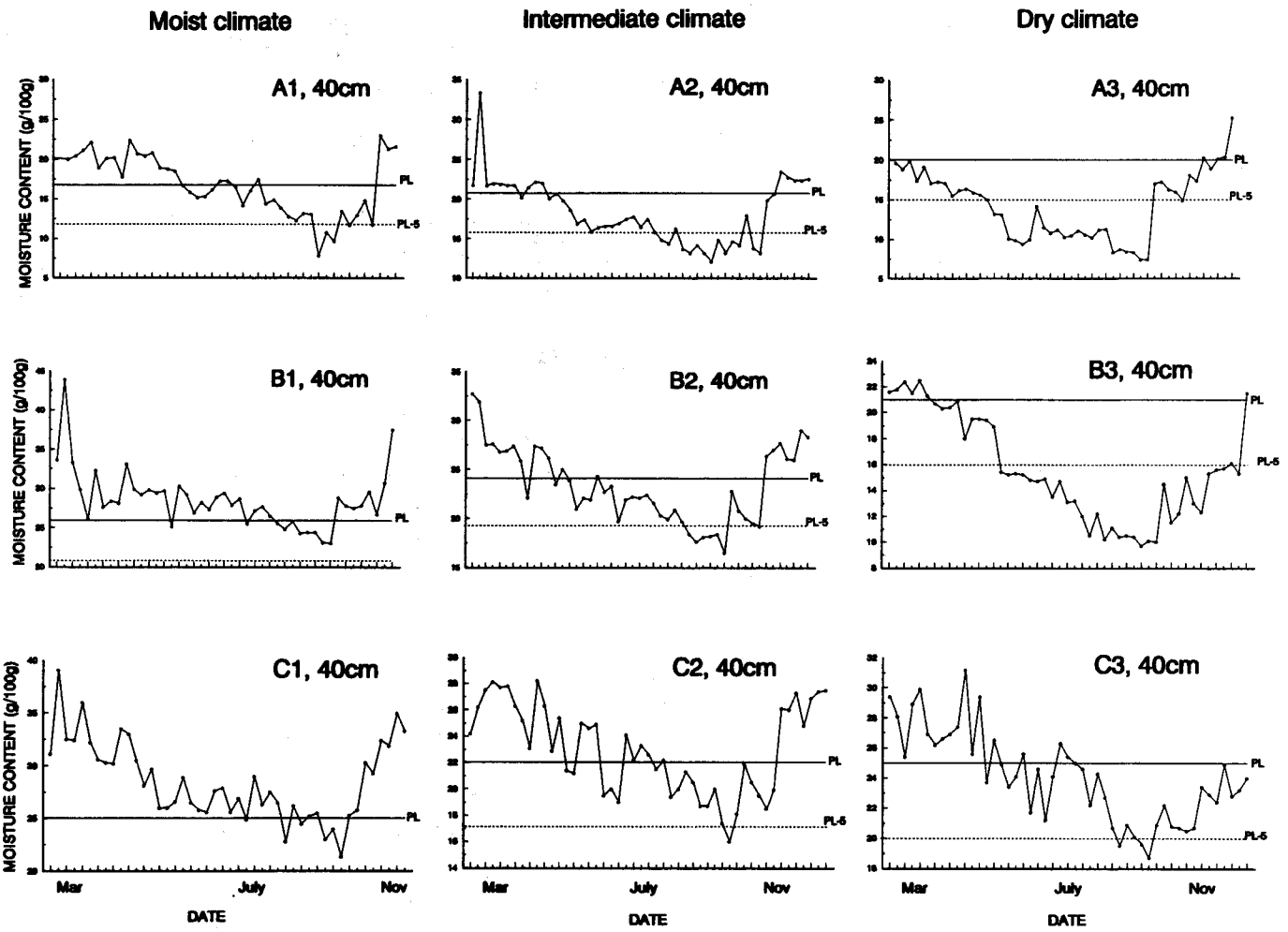


Figure 1. Changes in moisture content and consistency during 1991 for upper subsoils at 40-cm depth. Sites A1-A3 are light loams, sites B1-B3 heavy loams and sites C1-C3 dense clays. (PL = plastic limit moisture content; dotted line = plastic limit moisture content minus 5 g/100g)

acceptable period would exist for the stripping of the lighter loamy soils, but suitable conditions for stripping the clay loams and clays would have been confined to the months of August and September in 1991. A wetter-than-average year would thus completely preclude the stripping of certain soil textures if earthmoving was legally controlled by use of the plastic limit moisture content. The graphs also show that while it may be desirable to delay soil-stripping operations to await a moisture content of 2% to 5% lower than the plastic limit, during which time bearing capacity will have increased further, this is only likely to be practicable in the drier parts of England and for loamy soils.

The main problem with using plastic limit as a controlling moisture threshold for soil movements is that it involves regular site measurements of moisture conditions. For the quarrying operator wanting to plan his next season's soil stripping or the earthmoving contractor wishing to make best use of manpower and equipment, there is a need for a system to better define normal soil stripping periods for specific soils in specific climatic regions.

Soil Moisture Deficit and Attainment of Plastic Limit

The research was extended to develop relationships between plastic limit attainment dates and soil moisture deficits. Modeled soil moisture deficits throughout the year are available in Britain from a service run by the Meteorological Office. Actual soil moisture deficits were calculated for all sites and compared with the

modeled estimates and with the dates at which individual soil layers reached plastic limit. Using this approach, a model was derived that gave target soil moisture deficits for attaining plastic limit for specific layers of the soil (table 3).

Soil-Stripping Seasons

Through applying this model to long-term average moisture deficit data for England and Wales it is possible to produce a series of maps defining the soil-stripping season. An example is given in figure 2, in which median dates for attainment of a specified deficit (in the case of figure 2A, a deficit of 50 mm) are used to indicate the average start to the soil-stripping season for different parts of England and Wales. Because rewetting of soils does not necessarily occur evenly, or from the surface, a poor correlation was found between whole-profile deficits and the return of individual soil layers to plasticity in the autumn. For this reason a date related to the return to meteorological field capacity is used to define the average end of the soil stripping season (figure 2B).

Table 3. Soil moisture deficits for attaining plastic limit during soil drying

Soil layer	Deficit mm
All topsoils	25
Subsoils above 50-cm depth:	
Light and medium loams (10% to 27% clay)	25
Heavy loams and clay (>27% clay)	50
Subsoils below 50-cm depth[1]:	
Light loams (<18% clay)	25
Medium loams (18% to 27% clay) and dense heavy loams and clays (>1.5 g/cm^3)	50
Heavy loams and clays (>27% clay)	100

[1] layers at 100 cm may never become friable except in dry climatic zones or in very dry years

Use of the whole suite of maps generated by this study has enabled England and Wales to be divided into three climatic zones (figure 3) for which the best periods for stripping soil layers of various texture and depth below the surface can be derived (table 4).

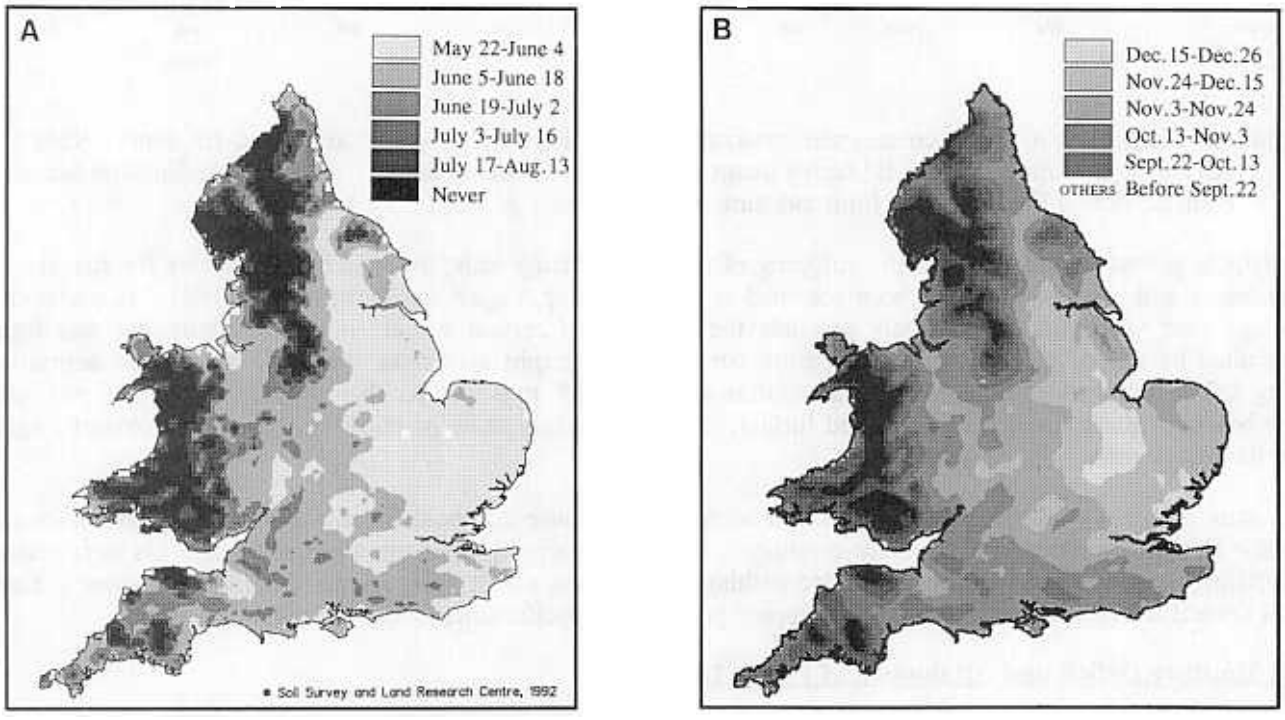


Figure 2. Maps showing in an average year (A) the dates of attainment of a 50-mm soil moisture deficit under grass and (B) the date of return to field capacity.

Soils with <10% clay that do not have a plastic limit are assigned a provisional moisture deficit threshold of 5 mm for the start of stripping. A date of 2 weeks before the return to meteorological field capacity is used to define the average end date of the stripping season for all soils except those having groundwater within 1-m depth during some part of the growing season; a date of 4 weeks before the return to field capacity is used for these.

No-Go Periods for Soil Stripping

While information on the best periods for stripping in a year of average climate is generally useful in defining normal earthmoving periods, it does not allow the optimum use of "shoulder" seasons in drier than average years. Using data for the driest standard deviation, it is possible to define no-go periods, though accepting that rigorous use of such might preclude earthmoving at certain times in exceptionally dry years. Figure 4 presents two of the maps used to derive the start and end of the no-go periods shown in table 5. The start is defined in relation to the dry standard deviation date for the end of field capacity (one week after for most soils, but 3 weeks before for difficult soils) and the end by the dry standard deviation date for the attainment of the moisture deficits in table 3.

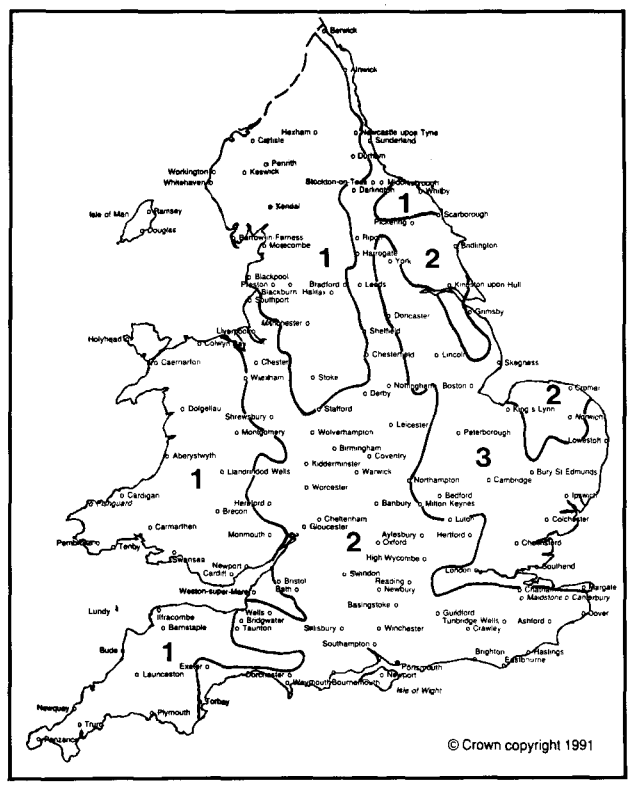


Figure 3. Climatic zones for soil stripping in England and Wales.

Table 4. Best periods for stripping vegetated soils in an average year

Strip depth[1]	Soil[2]	Climatic zone 1 [3]	Climatic zone 2 [3]	Climatic zone 3 [3]
<30 cm	VL[4]	Mid Apr.-early Oct.	Late Mar.-early Nov.	Late Mar.-early Dec.
	L,M,H,D	Late May-early Oct.	Early May-early Nov.	Early Apr.-early Dec.
30 to 60 cm	VL[4]	Late Apr.-early Oct.	Mid Apr-early Nov.	Early Apr.-early Dec.
	L,M	Late May-early Oct.	Early May-early Nov.	Early Apr.- early Dec.
	H,D	Late June-early Oct.	Early June-early Nov.	Late May-early Dec.
>60 cm	VL[4]	Late Apr.-early Oct.	Mid Apr.-early Nov.	Early Apr.-early Dec.
	L	Late May-early Oct.	Early May-early Nov.	Early Apr.-early Dec.
	M, H	Late June-early Oct.	Early June-early Nov.	Late May-early Dec.
	D	Mid July-mid Sept.	Early July-mid Oct.	Late June-mid Oct.

[1] Total depth from surface of all soils to be stripped and retained for reuse as a growing medium in restoration.
[2] The texture of the heaviest layer to be stripped for reuse. VL=very light (<10% clay), L=light (10% to 18% clay), M=medium (18% to 27% clay), H=heavy (>27% clay), and D=difficult (heavy soils affected by groundwater during the growing season).
[3] From figure 3.
[4] Opportunities for stripping very light soils occur during dry weather throughout the year but are best during the periods indicated.

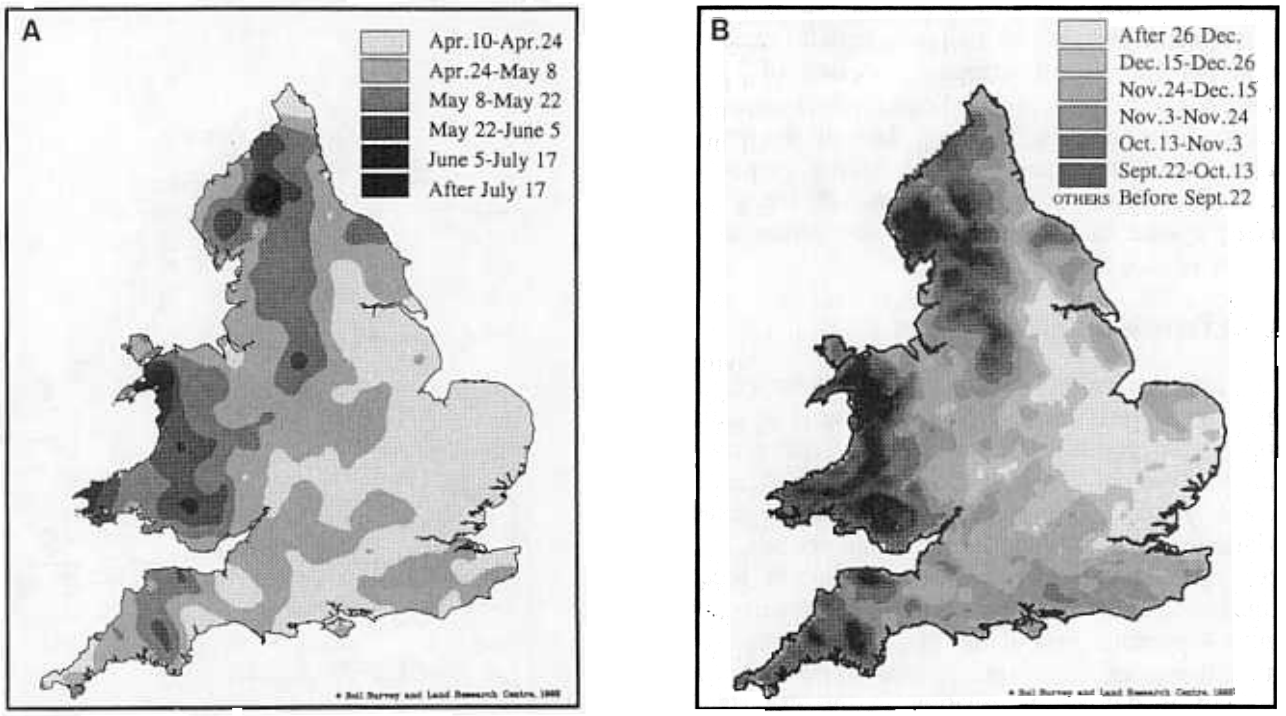


Figure 4. Maps showing in a dry year (A) the dates of attainment of a 25-mm deficit under grass and (B) the date of return to field capacity

Table 5. No-go periods for stripping light, medium, heavy and difficult soils

Strip depth[1]	Soil[2]	Climatic zone 1 [3]	Climatic zone 2 [3]	Climatic zone 3 [3]
<30 cm	L,M,H,D	Nov.10-Apr.10	Dec.1-Apr.1	Jan.1-Mar.20
30 to 60 cm	L,M	Nov.10-Apr.10	Dec.1-Apr.1	Jan.1-Mar.20
	H,D	Nov.10-May 1	Dec.1-Apr.20	Jan.1-Apr.20
>60 cm	L	Nov.10-Apr.10	Dec.1-Apr.1	Jan.1-Mar.20
	M,H	Nov.10-May 1	Dec.1-Apr.20	Jan.1-Apr.20
	D	Oct.10-June 10	Nov.1-May 20	Nov.1-May 20

[1] Total depth from surface of all soils to be stripped and retained for reuse as a growing medium in restoration.
[2] The texture of the heaviest layer to be stripped for reuse. L=light (10% to 18% clay), M=medium (18% to 27% clay), H=heavy (>27% clay), and D=difficult (heavy soils affected by groundwater during the growing season).
[3] From figure 3.
NOTE: Opportunities for stripping very light soils (<10% clay) occur during dry weather throughout the year and can occur in winter.

Day-to-day Control of Soil Stripping

While it is clearly useful to be able to plan in advance the opportunities for soil stripping, once earthmoving equipment is on-site it is important that soil stripping should continue only while soil conditions are

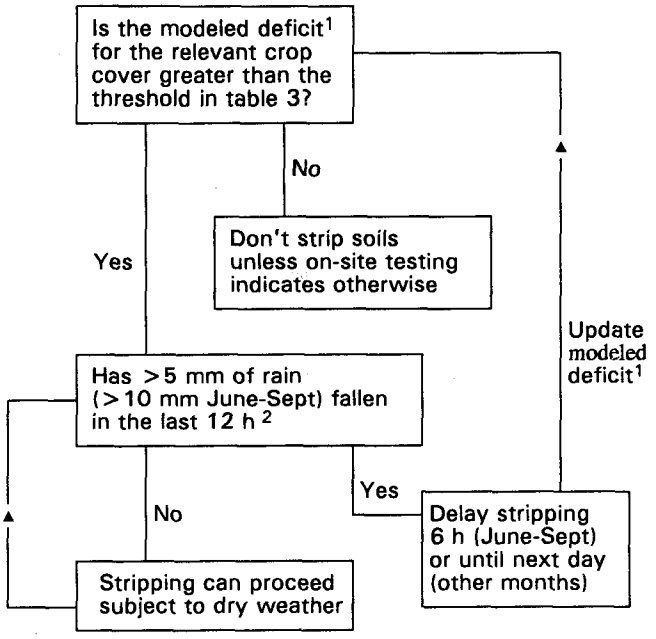
suitable. Information on dry or average years is of little relevance in this situation and it is necessary either to directly check soil consistency with a plastic limit test or to run a moisture-deficit model interactively with daily information on rainfall in the same way as irrigation might be scheduled. Figure 5 gives an example of the decision processes that might be incorporated in such a model.

The approach in figure 5 brings rainfall into the equation. As a general rule soil stripping and handling should not be started during wet weather and should be suspended if rain starts during operations. However, there is the need for some flexibility as steady rain on a dry soil is likely to have less immediate effect on soil strength and plasticity than light rain on a very moist soil. After rainfall ceases there will always be surplus water in the surface layers of soil. If earthmoving continues immediately, the surface layer will become compacted, ruts will be formed, and any further rain will tend to lie on the surface and drain away far more slowly than previously. Conditions will then continue to deteriorate during earthmoving. Suspending earthmoving for a period of hours after rain stops enables surplus water to drain away and there is a consequent lessening of the risk to soil structure when earthmoving recommences. The amount of rain that has fallen is important. Less than 5 mm over 12 hours will increase the moisture content of the surface 100 mm of an average soil by <4g/100g. Amounts of >5 mm in spring and autumn are likely to return the upper 100 mm of soil (which may have a moisture content only a few percent below plastic limit) to plasticity but, in summer, larger amounts of rainfall (up to 10 mm) are likely to be accommodated by the soil without this occurring.

Influence of Vegetation

While the foregoing has emphasised the importance of building up a significant soil moisture deficit before soil stripping commences, the part played by a good vegetative cover has not been addressed. It may be obvious to soil scientists that in a temperate climate subsoil layers will not dry significantly without the moisture demand of a plant root system, but it has been common practice not to crop agricultural land in the months preceding the start of mineral working and soil stripping.

The disadvantages of this are seen in figure 6, which compares the data for some of the bare soil and wheat plots with that for the grass plots in our study. While



[1] weekly modeled deficits can be updated daily by adding 2 mm and subtracting the daily rainfall.
[2] rain-gauge readings at 07.00 or 17.00 h.

Figure 5. Flow chart for scheduling soil stripping using moisture deficit and rainfall data.

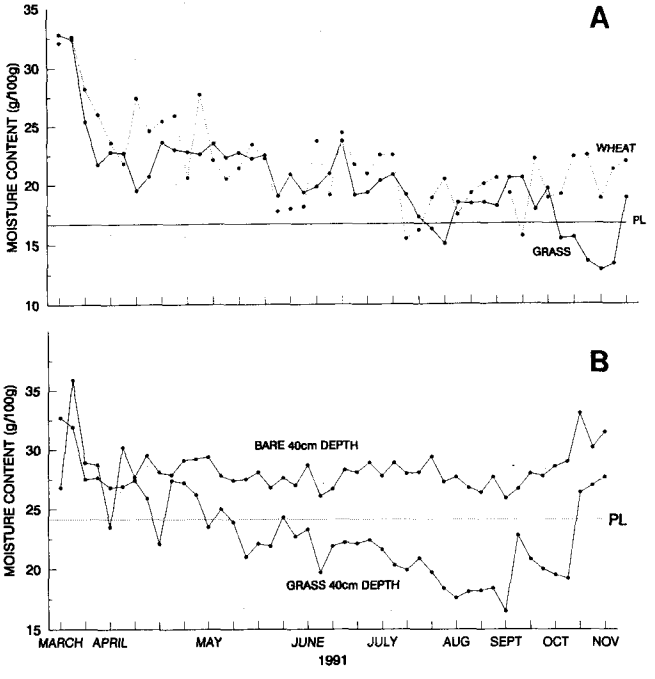


Figure 6. Extent of soil drying in relation to crop:
(A) light loam subsoil at 100 cm depth;
(B) heavy loam soil at 40 cm depth.
PL = plastic limit moisture content.

the wheat crop (fig. 6A) extracts moisture at depth as efficiently as grass up to mid summer, once it senesces prior to harvest it ceases to use soil moisture. The grass crop thus has a distinct advantage in continuing to dry deep soil layers, particularly when the autumn is relatively dry but there is a flush of autumn grass growth. Without any vegetation, evaporative drying seldom extends below a depth of 30 cm, so while the topsoil may dry to a friable state, the major part of the subsoil will remain plastic throughout the year, especially when heavy-textured (Fig. 6C).

Conclusions

The study has shown that the season for stripping soil resources from surface mine and quarry sites can be defined in relation to the attainment and duration of moisture contents below the plastic limit. Defining the soil-stripping season using such criteria is acceptable in the drier parts of England and Wales but less so in the wetter parts, where heavy-textured subsoils may seldom dry to the plastic limit. One option for testing for suitable soil moisture conditions is to regularly monitor soil moisture contents and plasticity at key depths in the soil profile to be stripped. However, soil moisture deficit and agrometeorological data could be used to identify no-go periods and best periods for stripping soils and could also be used for day-to-day scheduling of soil stripping. The assumption that critical soil deficits increase in relation to the clay content of a soil layer (as a result of an increasing difference between the moisture contents at field capacity and plastic limit) only holds true for soils of comparable mineralogy. There is evidence that the plastic limits of ferruginous and calcareous soils are closer to field capacity moisture content than in other similarly textured soils. Use of the critical soil moisture deficits presented in table 3 for such soils would thus overestimate the amount of soil drying necessary prior to stripping, but at least the error would be on the safe side.

Code of Practice

The results of this research are to be integrated into a Code of Practice for Soil Handling to be prepared by the Ministry of Agriculture, Fisheries and Food for public consultation in 1994. Soil stripping is, of course, only one element of soil handling and the Code must consider re-spreading of soils from stockpiles. Although the old adage "soil which enters a stockpile in good condition is more likely to be removed from it in good condition" is largely true, it is important to ensure that stockpiles are kept vegetated, that a soil moisture deficit exists during removal of soils from them and that the respreading season allows enough time for a ground cover crop to be established before the end of the growing season. For this reason the season for respreading soils is likely to be shorter than the season for stripping them. While the Code will seek to minimize the damage to soils that can occur during conventional earthmoving systems, there is considerable opportunity for mineral extraction companies to further reduce the chance of damaging soil through adopting systems that avoid the trafficking of soils (e.g. Bransden 1991).

Acknowledgements

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USE OF CONE PENETROMETER DATA TO EVALUATE PRIME FARMLAND ROOTING MEDIA[1]

Robert E. Dunker, Scott L. Vance, Charles L. Hooks, and Robert G. Darmody[2]

Abstract: Reclamation of prime farmlands after mining includes various methods of excavation, transportation, and placement affecting the physical properties of the reconstructed soil. Poor soil physical condition has proved to be the most severe and difficult limiting factor in the reclamation of prime soils. Deep tillage has become a practice accepted by the mining industry as a final step in the reclamation process for row-crop acreage. The penetrometer can be an important management tool for the mine operator to assess levels of soil compaction resulting from soil reconstruction and to evaluate the effectiveness and depth of deep tillage operations. Soil strength measurement with the deep profile penetrometer is shown to be a viable method in assessing long term yield potential of mined land when chemical and plant nutritional variables are not yield limiting.

Additional Key Words: soil strength, mine reclamation.

Introduction

Soil horizon replacement required by Illinois law (Illinois Dep. of Mines and Minerals 62 Ill. Adm. Code Part 1823) mandates the excavation of the natural topsoil and subsoil or suitable rooting material prior to the mining operations. The soil horizons are successively replaced to a total depth of 122 cm (48 in) over the graded mine spoil. The physical condition of a reconstructed soil will vary due to both the pre-mine soils used for construction and to the methods of excavation and placement (Vance et al. 1987). Poor soil physical condition has been identified as the major limiting factor affecting crop performance on mined land in Illinois (Dunker et al. 1992a, Hooks et al. 1992). The Illinois Department of Mines & Minerals (IDMM) allows the use of deep tillage to ameliorate subsoil compaction created in mine soil reconstruction and has become an accepted practice by the industry in the Illinois coal basin. IDMMM classifies deep tillage as any tillage to a depth below 45 cm (18 in) and since it is a non-standard agricultural tillage practice, it can be applied only once before the mandatory productivity testing period. If deep tillage is to be a viable method for achieving productivity restoration standards for mined land, effects must be immediately measurable and permanent. Yield effects of tillage depth and persistence of tillage effects has been studied (Dunker et al. 1992b, Dunker et. al. 1993) using soil strength measured with a deep profile cone penetrometer as the parameter by which tillage depth, effectiveness, and longevity is evaluated. Soil strength averaged over 23 to 112 cm (9 to 44 in) soil depth was significantly correlated with six year mean corn yields (-0.97**) and four year mean soybean yields (-0.92**).

Parameters commonly used to relate the mechanical impedance of soil to root growth are bulk density and penetrometer resistance. Cone penetrometers have been used to measure soil strength in agricultural and engineering applications for many years. Soil penetrability is a measure of the ease with which an object can be pushed or driven into the soil. Soil factors influencing penetration resistance are matric potential (water content), bulk density, soil compressibility, soil strength parameters, soil structure, and soil texture (Bradford 1986).

Soil bulk density, the ratio of the mass of dry solids to the bulk volume of soil, is commonly used to characterize soil compaction and soil structure. Since bulk density relates to the combined volumes of the solids and pore spaces, any factor that influences soil pore space (such as compaction) will affect bulk density. Fine-textured soils such as silt loams, clays, and clay loams generally have lower bulk densities than sandy soils. The fine-textured

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[2] Robert E. Dunker, Agronomist, Scott L. Vance, Sr. Research Specialist, Charles L. Hooks, Agronomist, and Robert G. Darmody, Associate Professor of Pedology, University of Illinois, Urbana, IL, USA.

soils tend to be organized in porous structural units (peds), resulting in high total pore space and a low bulk density. Sandy soil particles may pack closely together, which results in higher bulk densities.

Some studies have supported the concept of a critical bulk density beyond which roots cannot penetrate as the physical parameter that will best characterize root growth (Veihmeyer and Hendrickson 1948, Zimmerman and Kardos 1961). Other research (Taylor and Gardner 1963, Taylor and Burnett 1964) has suggested that penetrometer resistance, not bulk density, may be a better predictor of root system performance. This may be especially true for mine soils because during the reclamation process, large blocks of naturally compacted soil may remain undisturbed when replaced. High traffic reclamation systems will cause additional compaction effects on these soil materials, resulting in a high strength, low porosity soil. In ameliorating this compaction with deep tillage, shearing action creates large void spaces within the compacted blocks. Bulk density, which gives a very good estimate of porosity within a measured soil unit, may tend to underestimate the effects of large voids on root system performance. Uhland cores, a method that extracts small soil units to estimate bulk density, may not contain the large voids in proportion to the total soil volume. Soil strength, as measured by a penetrometer, may more accurately reflect resistance encountered by roots.

Numerous studies have described the effects of mechanical impedance on root growth and crop yield (Taylor 1971, Russell 1977, Kahnt et al. 1986, Boone and Veen 1982, Shierlaw and Alston 1984). Taylor and Gardner (1963) and Taylor and Burnett (1964) have reported that penetrometer resistance values greater than 2 MPa (290 psi) may result in severe root impedance, and at 2.6 MPa (380 psi) root elongation will be severely restricted.

Development of Cone Penetrometer for Mined Land

The cone penetrometer was originally developed by the U.S. Army Corps of Engineers and was designed for hand operation (Bradford 1986). Hand-operated penetrometers will work well for measuring resistance at near surface depths. However, accuracy is limited by the ability of the operator to maintain constant pressure, and the applied force is limited by the strength of the operator. Most hand-operated penetrometers utilize a proving ring and dial gauge requiring a second person to read the dial at the operators vocal command. Electronic load-measuring recording devices have been developed to replace the proving ring with either a linear displacement transducer or a strain-gauge load cell (Prather et al. 1970, Anderson et al. 1980). Vehicle-mounted systems were developed and used by Williford et al. (1972), Cassel et al. (1978), and Smith and Dumas (1978). These vehicle systems were designed to measure near-surface tillage effects on natural soils.

A constant rate deep profile recording penetrometer was developed to serve the need to quantify physical properties of reconstructed soils (Hooks and Jansen 1986). This device, capable of recording soil strength to a depth of 112 cm (44 in), utilizes a modified tractor-mounted hydraulic (Giddings) coring machine. This system uses a 645 mm^2 (1 in^2) 0.525 rad (30 degree) right circular cone and operates at a constant penetration rate of 3 cm sec^{-1} (1.2 in sec^{-1}) (ASAE Std 313.2).

A modification of the original device is currently under development and testing. It incorporates two penetrometer masts center mounted on a small low ground pressure articulated tractor (Hooks et al. 1993). The electrohydraulic operating system is remotely controlled by a laptop computer in the tractor cab. Data from two samples taken to a depth of 122 cm (48 in) can be simultaneously collected and displayed to the operator. This system has reduced manpower requirements while increasing productivity of data collection over the previous system.

Cone penetrometer data of minesoils is taken in the spring, when soils are uniformly moist and near field capacity. This minimizes the effects of variable soil moisture on soil penetration resistance. Sampling at each location consists of paired samples taken to a depth of 112 cm (44 in). The data are typically recorded as average

penetrometer resistance of a given depth segment. In University of Illinois studies on soil strength effects on crop yields, five, 23 cm (9 in) depth segments are used for data analysis (Vance et al. 1992).The first segment (0 to 23 cm) is not used in the data analysis. This segment is easily altered by tillage and its moisture content may be highly variable due to differences in surface topography.

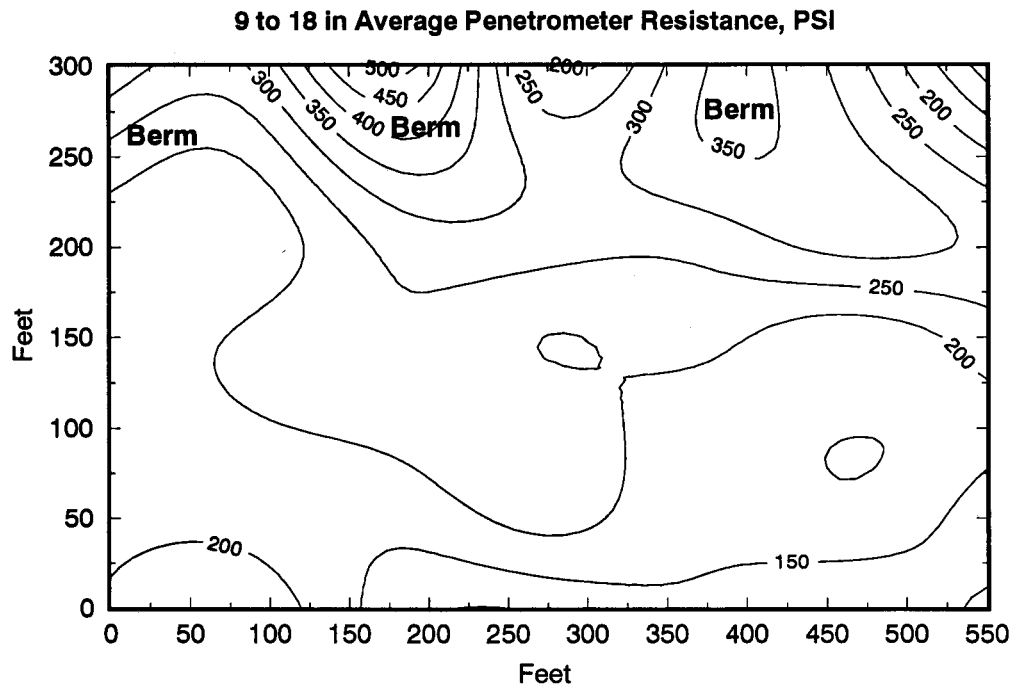
From an engineering or physical approach, soil strength is a real and true value that should be predictable within given values of moisture content, texture, density, etc. In evaluating reclamation treatments, soil strength may be approached as a relative value that is a composite of the effects of moisture content, texture, density, etc. Moisture content, particularly when below field capacity, is a major factor in soil strength. However, when penetrometer data is collected in the spring when soils are most uniformly moist, we consider minor differences in soil moisture between adjacent reclamation treatments to be a reflection of the soil/environment interaction and a valid part of the composite value "soil strength."

Application as a Management Tool

Soil strength data are an excellent parameter for use in reclamation planning and evaluation. Use of a deep profile recording penetrometer is a relatively fast and nondestructive method of assessing compaction effects of reclamation operations. For example, the reclamation plan at one mine in Illinois specifies moving topsoil with a cross-pit wheel to create topsoil storage berms on the graded rooting media. This mine had previously used a scraper haul system to transport and replace topsoil, but unacceptable compaction resulted from excessive wheel traffic. The topsoil berm system eliminates or reduces wheel traffic by use of low ground pressure dozers to push topsoil from the berms. The operator was still concerned about the effects of placing large topsoil berms on the graded rooting media. To assess compaction beneath the berms, soil strength levels were measured with a cone penetrometer to a depth of 112 cm (44 in) on a 15 by 15 m (50 by 50 ft) grid where berms had been located before topsoil was replaced. Profile data were segmented by depth and contour plotted to provide a series of maps of the area (figs 1-2). Investigation of aerial photography revealed that part of three berms had been located along the north (top) area of the tract, and a small one may have extended into the southeast (bottom left) of the area. The effects of the large berms are evident on every depth segment map. However, the effect of the smaller berm is detected only in the 45 to 68 cm(18 to 27 in) and 68 to 91 cm (27 to 36 in) depth segments. Penetrometer data in this format allows the operator to locate compaction problems and to make decisions on the number and size of berms placed in an area.

One of the most significant advances in reclamation of prime farmland soils in the Midwest has been the development of deep tillage technology. The degree and depth of compaction in mine soils vary with the reclamation practices used in soil reconstruction (Vance et al. 1987). Because of the need for deep soils in the corn belt, compaction is a serious problem when rooting is limiting above 122 cm (48 in). Deep-tillage equipment now exists that will ameliorate compaction to a depth of 122 cm (48 in). Intermediate-depth tillage equipment that can effectively disrupt soil to a depth of 91 to 96 cm (36 to 38 in) has been refined and is commonly used where severe compaction is limited to that depth. Dunker et al. (1992b) evaluated the effects of six deep-tillage treatments ranging in effective depth from 23 to 122 cm (9 to 48 in) on a severely compacted scraper-placed soil in southern Illinois. Corn yield increased with increasing tillage depth within and across the 4 yr studied. Post tillage penetrometer data indicated that amelioration effects of tillage remain at least 4 yr after application of tillage treatments. Subsequent study (Dunker et al. 1993) shows these effects persist into the sixth year and suggests that unless the soil is recompacted by mismanagement, effects of deep tillage to depths of 91 to 122 cm (36 to 48 in) should be permanent.

Application of deep tillage on mined land in Illinois is an expensive operation costing between $140 and $180 per hectare ($350 to $450 per acre). Penetrometer measurement can be a valuable method the operator can use to verify tillage effectiveness and depth since many operators are specifying tillage depth as part of the agreement with



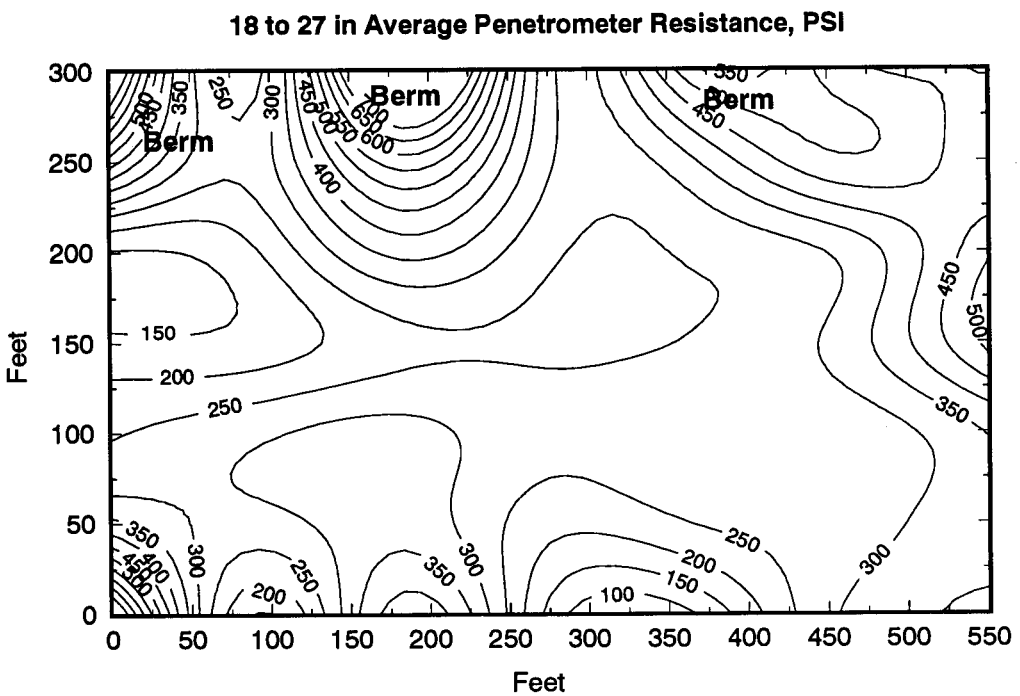
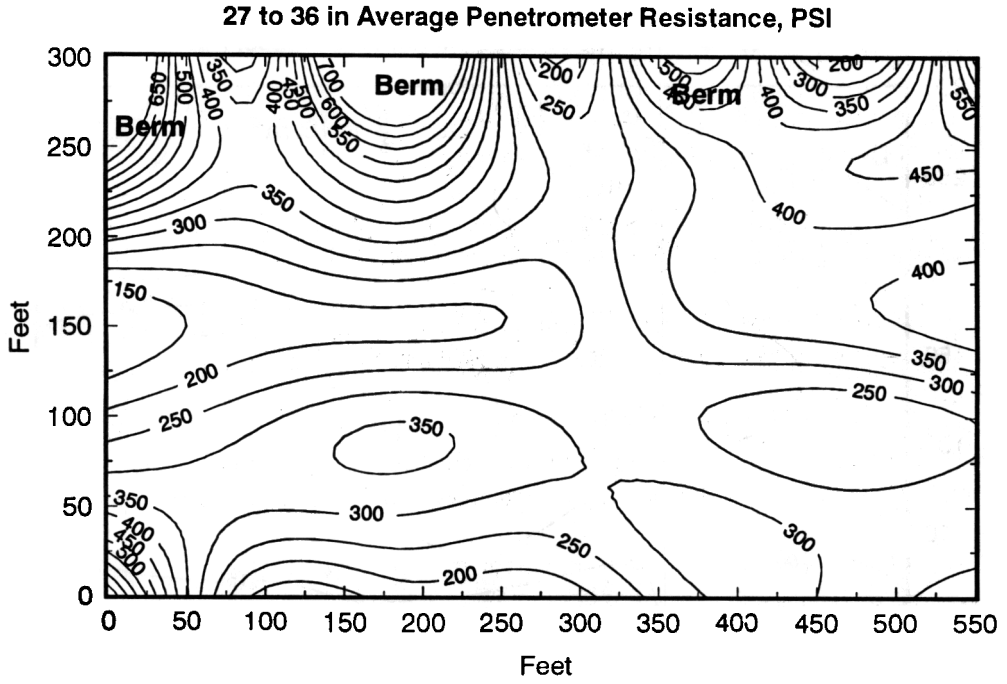


Figure 1. Soil strength contour maps of 9 to 18 in and 18 to 27 in depth segments in topsoil berm area.



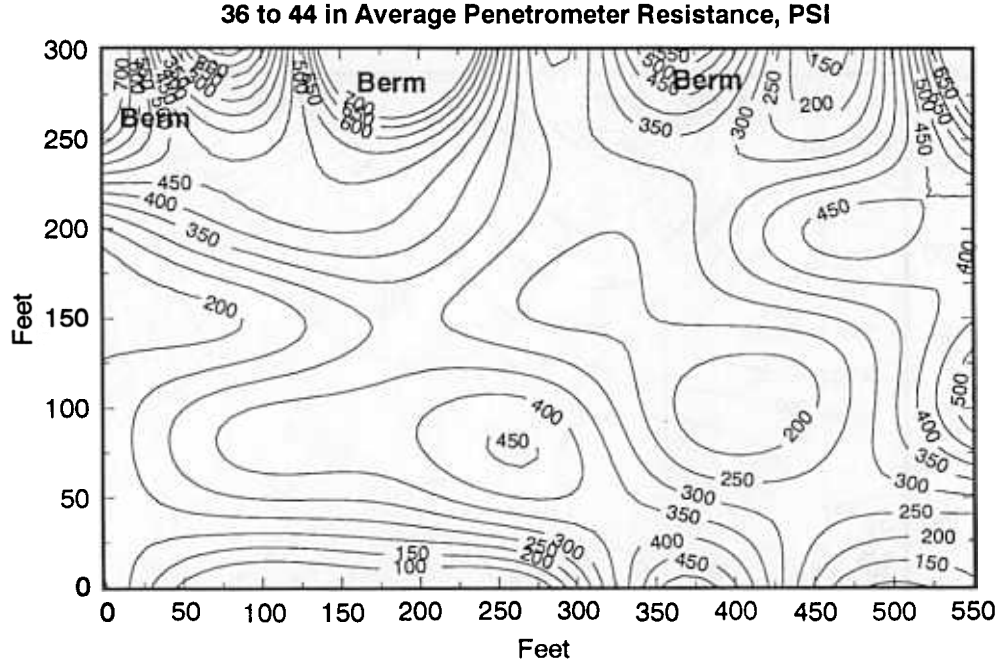


Figure 2. Soil strength contour maps of 27 to 36 in and 36 to 44 in depth segments in topsoil berm area.

the deep tillage contractor. Figure 3 shows two methods by which soil strength data can be used for this purpose. The contour map presents data from an area where the north half was deep tilled to 122 cm (48 in). The mine operator can look at these data and be assured that the tillage was effective in ameliorating compaction and is uniformly applied. The lower graph (fig. 3) presents the average profiles of the treatments and gives the mine operator an indication of treatment differences. If needed by the mine operator, profiles for specific sample points can be generated to analyze problem areas.

To evaluate persistence of tillage treatment effects and repeatability of penetrometer readings, soil strength measurements were taken in 1988, 1989, 1991, and 1993 on a deep-tilled scraper placed mine soil in southern Illinois (Dunker et al. 1993). Figure 4 presents profile data of two of these tillage treatments. The TLG-12 treatment (Kaeble Gmeinder TLG-12) utilizes a cut-lift operation to shatter the soil to a depth of approximately 80 to 91 cm (32 to 36 in). The DMI treatment (DMI, Inc. prototype deep plow) utilizes a two-lift solid winged shank ripper and has an effective tillage depth of 122 cm (48 in). Penetrometer profile data presented in figure 4 show that effects of these treatments remain unchanged over the 6 yr period (treatments were applied in 1987) and that soil strength data measured with the penetrometer is repeatable over time.

Application for Regulatory Use

Penetrometer measurements on reclaimed mined land vary widely. These are associated with wide ranging values in crop yield. Prime farmland reclamation areas with high soil strength levels have had the lowest crop yields, and low soil strength soils have had the highest yields. Vance et al. (1992) reported significant correlation of 23 to 112 cm (9 to 44 in) average penetrometer resistance with 6 year mean corn yield (-0.98**) and six year mean soybean yields (-.0.99**) in a study evaluating several methods of reclamation.

Illinois performance standards for prime farmland require measuring yields of reclaimed areas and comparing them to a predetermined target yield. Target yields are based on productivity indexes of premine soils with a weighted productivity index calculated for each permit. The Illinois Department of Agriculture has developed the Agricultural Lands Productivity Formula (a crop yield based system) to provide a calculated standard yield to be used for mined land comparison to determine if productivity has been restored (Giordano 1992). This formula includes a countywide-based weather adjustment factor to account for weather effects. It is based on USDA Agricultural Statistics Service information and adjusted for harvest loss. If the mine operator can produce crops equal (based on a 90% confidence interval) to the target yield for 3 yr within a 10 yr period, productivity is deemed restored.

Figure 5 presents the relationship of mean corn yields expressed as a percent of the prime farmland target yield for each treatment to 23 to 112 cm (9 to 44 in) average penetrometer resistance. This database represents corn yields collected at numerous locations with various soil reconstruction methods in Illinois over a 15 yr period. This relationship of soil strength and yield is curvilinear. This would be expected because as soil strength decreases, yield can only increase to a certain potential level. In contrast, as soil strength increases to a level that prohibits root penetration by mechanical impedance, any additional increase in soil strength would have little or no effect.

Section 515 (b) (7) of the Surface Mining Control and Reclamation Act (Public Law 95-87) specifies regulations for soil removal, storage, replacement, and reconstruction. It states that the root zone material of prime farmland must be replaced with proper compaction and uniform depth over the regraded spoil material. This compaction requirement is yet to be quantified by the Office of Surface Mining in rule making. There has been considerable interest in recent years in the development of a soil-based model to evaluate mine soil productivity. A soil-based productivity index based on the premise that root growth is a function of the sufficiency of the soil could be a suitable method to evaluate mine soils if it is based on long-term mined land yields. Neill (1979) identified five soil properties (potential available water storage capacity, bulk density, soil pH, electrical conductivity, and aeration)

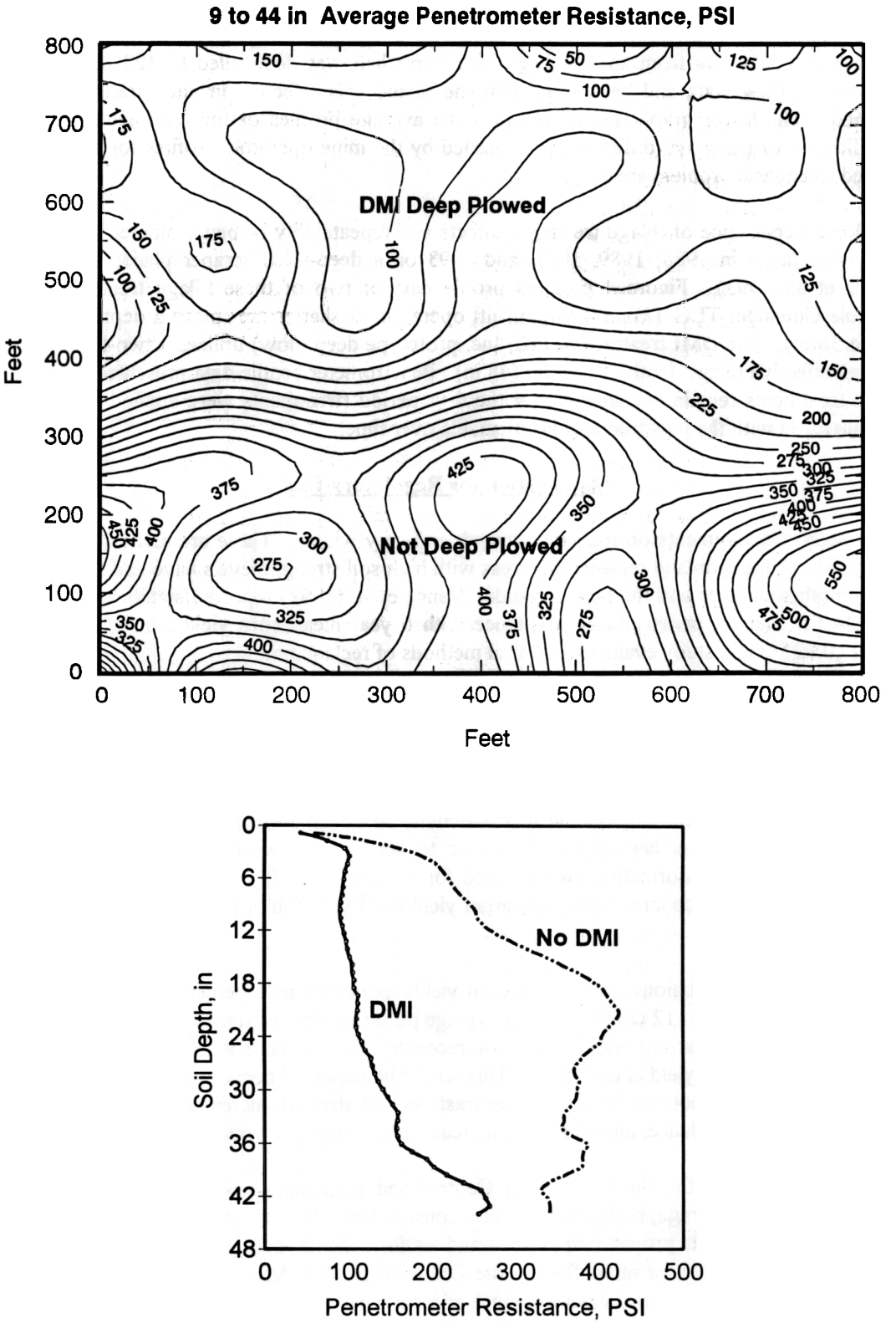


Figure 3. Soil strength contour map (top) of DMI deep-tillage (48 in) area and mean soil strength profiles (bottom) for the tilled and nontilled areas.

as being easily measured soil parameters that predict root growth. Cone penetrometer readings could be introduced to the productivity index model as a potentially efficient alternative to bulk density as a measure of soil compaction (Barnhisel et al. 1992). Soil strength measurement with the deep profile penetrometer is a viable method for assessing long term yield potential of mined land when chemical and plant nutritional variables are not yield limiting factors. While yield variation among years is associated more closely to weather variables than to soil factors, soil strength is closely correlated to mean yields averaged over multiple years.

Development of a compaction standard for prime farmland rooting media will enhance the success of a crop-based performance system. Requiring that rooting media meet a sufficiency level of compaction as a measure of quality control will prevent severely compacted mine soils from entering the crop performance testing stage. This will assure that only high quality mine soils will be allowed to pass through the bond release system.

Summary

The penetrometer can be an important management tool for the mine operator to assess levels of soil compaction created during soil reconstruction and to evaluate the effectiveness and depth of deep-tillage operations. Soil strength measurements with the deep profile penetrometer appear to be a viable method of assessing long term yield potential of mined prime farmland of the midwestern corn belt when chemical and plant nutritional variables are not yield limiting. The inclusion of a penetrometer resistance measurement as a surrogate for a compaction standard will improve the success of surface mine reclamation.

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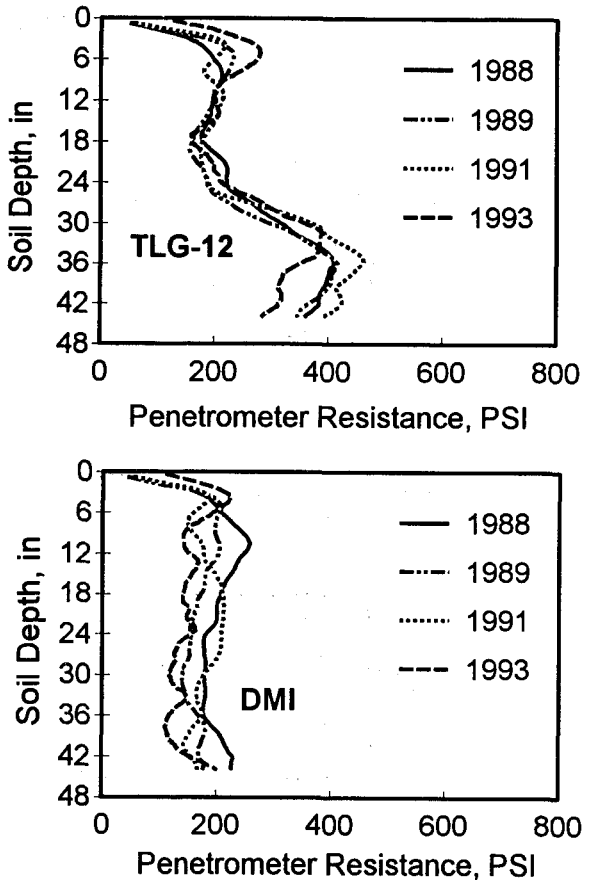
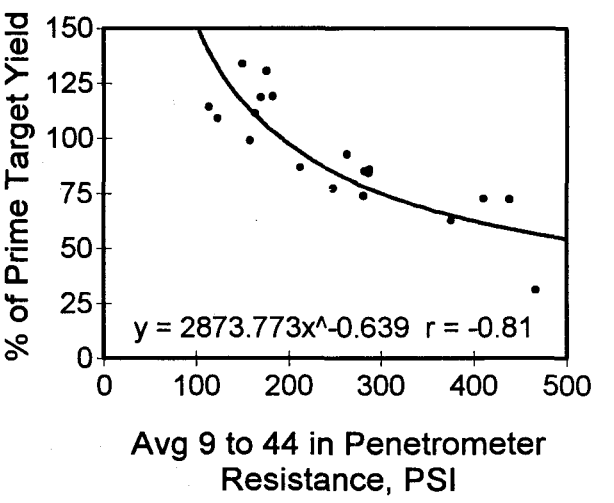


Figure 4. Mean penetrometer profiles for deep-tillage treatments over a 6 yr period, 1988-93.



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MEASURING MINE SOIL PRODUCTIVITY FOR FORESTS[1]

J. A. Burger, J. E. Johnson, J. A. Andrews and J. L. Torbert[2]

Abstract: The Surface Mining Control and Reclamation Act (SMCRA) of 1977 requires that mine operators reclaim mined land to achieve crop productivity levels equal to or greater than premined conditions. The standard for evaluating reforestation success is based solely on tree-seedling survival; however, survival is an estimator of stand density and not an indication of site productivity. The purpose of this study was to develop a model for estimating mine soil productivity based on the capability of growing merchantable timber. Seventy-eight fixed-area plots in Virginia and West Virginia were selected to evaluate the effects of mine soil/site properties on tree growth. These plots ranged from 0.02 to 0.04 ha and consisted of 5- to 9-year-old eastern white pine (*Pinus strobus*). A 1-m deep pit was dug at the base of one representative tree on each plot to determine rooting depth and physical and chemical properties of the subsurface horizons. The properties most correlated with tree growth were rooting volume, defined as the depth to a restrictive layer, slope percent, soil extractable phosphorus and manganese content, and electrical conductivity. A productivity model was developed based on the premise that limiting soil and site characteristics affect a species' optimum root distribution and thus its aboveground productivity. The developed model was verified using a white pine data set ranging in age from 11 to 15 years that was obtained from 14 reclaimed mines in Virginia. The model was significantly correlated with aboveground biomass and site index. Similar results were observed when predicted productivity estimates and site index values were added separately into established biomass equations.

Additional Key Words: reforestation, forest productivity, productivity models, forest site quality.

An Argument for Restoring Forests on Mined Land

Forest land as a designated postmining land use is gaining in popularity with coal operators in Virginia, Kentucky, and West Virginia. In Virginia, the Division of Mine Land Reclamation estimates that 80% to 85% of all reclaimed land is in a forest land use. Although hayland/pasture has been the predominant postmining land use in Kentucky, the Division of Field Services of the Kentucky Department of Surface Mining reports that reclamation efforts have been steadily shifting toward the use of trees. Hayland/pasture (like industrial and residential uses) is considered a higher-order land use than forestry; thus, conversion from premining forested conditions to these higher-order land uses is allowable under the provisions of Public Law 95-87 (U.S. Department of the Interior, Office of Surface Mining Reclamation and Enforcement 1988), and is desirable to a degree, in order to diversify the economic base of localities (Zipper 1986).

However, most of the hayland/pastureland created as a result of mining is going unused for its intended purpose because it is often in remote areas, is unfenced, and is often too steep to traverse with agricultural equipment. Areas 5 years of age and older are succeeding to mostly low-value pioneering woody vegetation, while other areas are experiencing vegetation dieback due to inadequate maintenance of mine soil fertility levels required by these domesticated grasses and legumes. Consequently, coal mine operators are realizing that maintenance and repair costs of hayland/pastureland offset the initially lower cost of establishing pastures instead of trees. There is also an increasing concern in the coal industry about the difficulty of demonstrating the productivity of

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[2]J.A. Burger, Professor of Forest Soil Science, J.E. Johnson, Associate Professor of Forestry Extension, J.A. Andrews, Graduate Research Assistant, and J.L. Torbert, Research Associate, Virginia Tech, Blacksburg, VA, USA.

hayland/pastureland in order to obtain bond release.

Another reason for this shift toward tree crops is landowner interest. Economic analyses are showing that investments in wood production have greater potential than those in forage production (Zipper 1986, Torbert and Burger 1993). The States of the Appalachian coal belt (Pennsylvania, West Virginia, Virginia, Kentucky, Tennessee, Alabama), particularly the portions that are mined for coal, are already heavily forested. Forestry and forest products are major contributors to the economic base of these States. Added-value analyses for Virginia show that the forest industry is the single most important industry in the State, employing at least 150,000 workers (Hall et al. 1987).

Importance of Productive Mine Soils for Forests

Forestry-based economies are dependent upon production and utilization procedures that are very much analogous to those used in agriculture. Forest trees are a cash crop like corn and soybeans; the only difference is the length of the production period. And like agricultural crops, the production of tree crops varies greatly with climate, soil conditions, and cultural practices. Foresters, like agronomists, are continually striving to improve genotypes and planting stock, and to improve soil conditions and fertility to maximize crop production. And landowners, whether they are growing tree crops or agricultural crops, measure their success (and livelihood) based on the tons of saleable plant products (wood fiber or grain) produced per unit land area.

Although reclamation to original land productivity levels is the intent of the law (Public Law 95-87, the Surface Mining Control and Reclamation Act), this intent is not reflected in current regulations. To reclaim surface mines to "unmanaged forest," regulations require only that a diverse assortment of tree species exceeding 30 cm in height at densities greater than 1,000 stems per hectare be established 5 years after initial planting. No productivity standard is required. A productivity standard for wood production that is based on tree survival and a 30-cm size standard, with no evidence of the ability of the reclaimed site to produce a saleable product, is not reasonable. As is the case for other crops, reclamation standards for forests should be based on the site's potential to produce, and not based only on tree survival.

Different reclamation approaches can result in a wide range of forest site productivity. Across 36 reclaimed sites in Virginia, all of which had 10-year-old white pines, Torbert et al. (1988a) found site index (a measure of forest site quality based on predicted heights of trees at age 50) ranging from 35 to 110 ft. Assuming these stands are harvested at age 35, a white pine stand growing on an area with a site index of 110 will yield over 10 times the volume of wood as a stand growing on an area with a site index of 35. Furthermore, the timber quality on the better site will be greater, since a larger proportion of the harvested wood will be in the high-value sawtimber category, as opposed to the low-value pulpwood or intermediate-value small sawlog categories. Thus, a harvest from the site-index-110 stand may yield 20 to 30 times more money than a harvest from the site-index-35 forest. These drastically different levels of productivity that result on reclaimed mine soils demonstrate the need to be able to estimate productivity and to determine which mine soil properties must be controlled during reclamation to produce high-quality forest land.

Although numerous studies have evaluated tree growth on mine soils (Brenner et al. 1984, Ashby et al. 1980, 1982), very few studies have attempted to "predict" tree growth or SI from mine soil or topographic variables. Considering that one of the intentions of Public Law 95-87 is to ensure that mined land is returned to its premining level of productivity, it is necessary that the relationships between mine soil properties and forest productivity be developed and understood by coal operators, regulatory personnel, and landowners.

Research Approach

To develop an estimator of mine soil productivity, a data base for young white pine (*Pinus strobus*) was used that contained both soil/site and growth information. Seventy-eight plots (ranging from 0.02 to 0.04 ha in

size) from 14 reclaimed surface mines in Virginia and West Virginia were used in this study.

White pine growth information was obtained at the end of the 1990 and 1991 growing seasons. Internode distances, total height, and ground line diameter were measured on all white pine. This information was used to derive four growth measures: total height at age 5, 2- and 3-year terminal growth, and a free-to-grow point. Two- and three-year terminal growth refers to the final 2 and 3 years of growth up to age 5. The free-to-grow point was defined as the average annual growth rate that occurred after the year that height exceeded the average of the two previous years by 100%.

Percent slope was measured on all plots. A soil pit (1 m^3) was located in the center of each plot at the base of a representative tree that exhibited no signs of suppression or damage. Subsurface soil samples were collected to determine effective rooting depth. Soil depth was measured as the depth to solid rock or to a traffic pan. Mineral soil samples were collected from each of the delineated horizons down to a depth of 1 m. Bulk density was determined for each of the subsurface horizons using the clod method (Blake 1965).

Two soil sampling points were randomly located on the plot to obtain surface soil samples (0 to 10 cm). At each of these locations surface bulk density was measured using the excavation method described by Blake (1965).

Soil samples were returned to the laboratory, air-dried to a constant moisture content, sieved using a 2-mm screen to separate coarse fragments, and analyzed for percent sand, silt, and clay using the hydrometer method. Soil pH was determined using a 1:1 soil-water mixture with a glass electrode (McLean 1982) and soluble salts were determined by measuring the electrical conductivity (EC) of a 1:5 soil-water extract (Bower 1965). Phosphorus was extracted with $NaHCO_3$ (Olsen and Dean 1982) and determined using a Jarrell-Ash plasma emission spectrometer.

An independent data set that included 14 sites on 7 reclaimed mines located in southwestern Virginia planted with white pine between the ages of 11 and 15 was used to validate the developed mine soil productivity estimator called productivity index (PI). Correlation analysis was used to determine the linear relationships between PI, SI, and aboveground biomass. Site index for individual trees was calculated using Beck's equation (1971a) for polymorphic curves, and biomass was calculated using Kinerson and Bartholomew's biomass equation (1977). Beck's intercept equation is useful to determine SI in stands younger than 20 years old (Balmer and Williston, 1983). Regression analysis was used to determine the significance between PI, SI, and aboveground biomass (Gale et al. 1991). The performance of PI was then evaluated using multiple-regression techniques, testing the predictive ability of SI versus PI in determining changes in aboveground biomass. Because of the importance of age in influencing changes in biomass, a Schumacher-type equation (Schumacher, 1939) of the form

$$\text{LN } Y = a + b/\text{age} + c(\text{index})$$

was used to compare predictions of aboveground biomass using the two site quality indices with age (Gale et al. 1991).

Model Development

The original productivity index (PI) model was developed by Neill (1979) having the following form:

$$PI = \sum_{i=1}^{r} (A \times B \times C \times D \times E \times RI)_i. \qquad (1)$$

In equation 1, A is the sufficiency of potential available water storage capacity, B the sufficiency of aeration,

C the sufficiency of bulk density, D the sufficiency of pH, and E the sufficiency of electrical conductivity. The product of these is summed over r, which refers to the number of 10-cm increments within the rooting depth. The RI term is the optimum fraction of roots in a soil horizon, based on Horn's water depletion curve (1971). Each model component is normalized to range from 0.0 to 1.0. A value of 0.0 indicates a totally limiting level of a soil property and a value of 1.0 indicates an optimum level. Operationally, for each depth increment the levels of each of the measurable soil properties are assigned a sufficiency value. These values are multiplied together with the RI for the ideal soil for that depth increment. Finally, the resulting values for each depth increment are summed to give the PI for the entire rooting depth.

Gale (1987) modified this equation, using the geometric mean, which gives equal weight to proportional differences in factor ratings and not to absolute differences. She also modified the equation to include percent slope and climate. Following is the form of her equation:

$$PI = \sum_{i=1}^{r} ([A \times B \times C \times D]_i^{1/4} \times WF) \times [SI \times CI]^{1/2}, \qquad (2)$$

where A was the sufficiency of potential available water, B the sufficiency of aeration, C the sufficiency of bulk density, D the sufficiency of pH, SI the sufficiency of percent slope, and CI the sufficiency of climate. The weighting factor (WF) was based on a curve developed by Gale and Grigal (1987) for midtolerant species.

We modified Gale's equation (1987), retaining the basic form but eliminating her sufficiencies for climate, aeration, and percent available water content, while adding sufficiencies for electrical conductivity and phosphorus. Our equation follows:

$$PI = \sum_{i=1} ([pH \times EC \times P \times BD]^{1/4} \times WF)_i \times [SI], \qquad (3)$$

where pH is the sufficiency of pH, EC is the sufficiency of electrical conductivity, P the sufficiency of phosphorus, BD the sufficiency of bulk density, and SI the sufficiency of percent slope.

Sufficiency Curve Development

Sufficiency curves, normalized between 0 and 1, were developed for each soil and site variable using information from either this study or previous sufficiency curves developed by Neill (1979), Pierce et al. (1983), and Gale (1987). The productivity index (PI) was calculated by determining the sufficiency rating for each soil horizon characteristic in addition to the WF and site characteristics on each plot for white pine. Because topography is not horizon-specific, its sufficiency was computed separately and multiplied by the PI. Specifics on the derivation of these curves can be found in the work by Andrews (1992).

Model Evaluation

A significant proportion of the variation in site index (SI) was explained by PI using simple linear correlation (R^2 = 0.72) (fig. 1). Both the PI (fig. 2a, R^2 = 0.71) and SI were significantly correlated with aboveground biomass (fig. 2b, R^2 = 0.90).

The addition of PI to age was compared with the addition of SI to age when used in a Schumacher-type equation (Schumacher 1939). In this study, equations with SI explained more of the variation in stand production than did equations with PI (table 1). A problem with multicollinearity in the equation was detected due to the significant correlation between PI and age. Because of the problem of multicollinearity, PI was not as good a predictor of site quality as SI used in the Schumacher-type equation. Another potential problem with PI is neglecting to sample each separate soil horizon. Since each soil horizon is important to root growth, precision of

the PI model may be lost when soil horizons are not sampled separately but are composited (Gale 1987). Within the range of SI sampled (50 to 122 ft), SI's of 60, 80, 100, and 120 produced similar results for aboveground biomass, as did PI's of 0.4, 0.6, 0.8 and 1.0 (fig, 3; PI's ranged from 0.30 to 0.97).

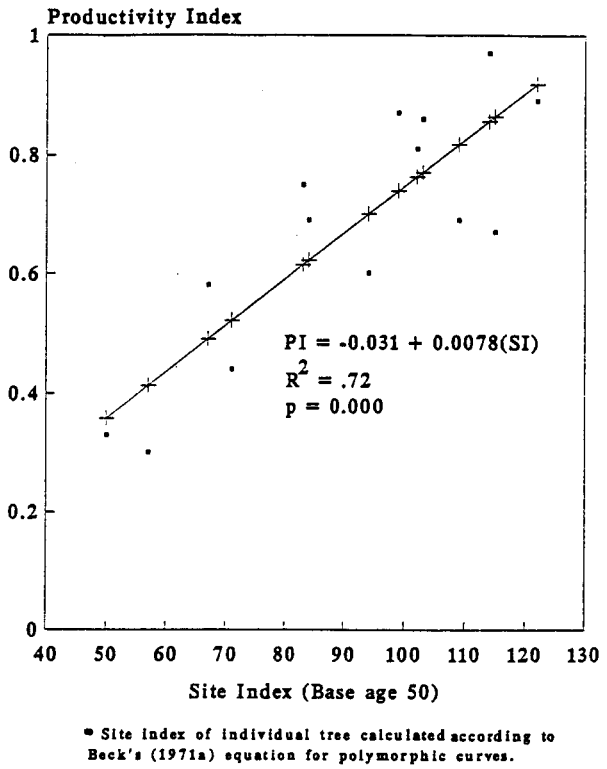


Figure 1. Relationship between site index (SI) and productivity index (PI) for 14 sites located in southwest Virginia.

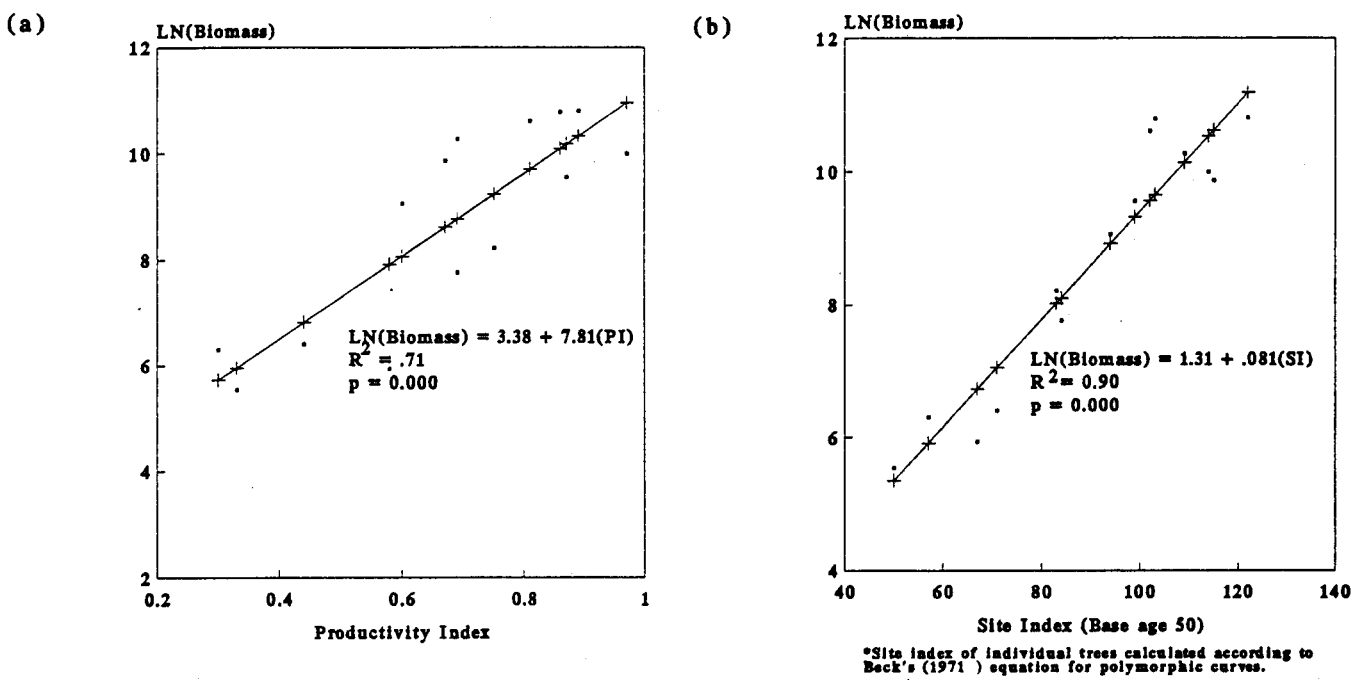
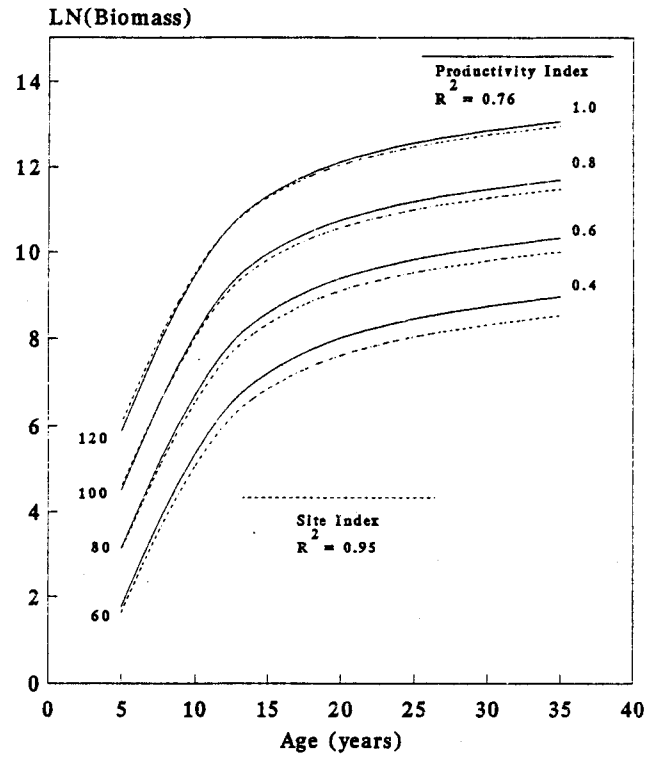


Figure 2. (a) Comparison of productivity index (PI) and biomass prediction; (b) comparison of site index (SI) and biomass predictions.

Table 1. Comparison of regression equations to predict tree biomass using productivity index (PI) versus site index (SI).

Equation	Adjusted R^2	P > F
LNBio = 7.42 - 42.03 (1/Age) + 6.86 (PI)	0.71	0.001
LNBio = 5.24 - 40.26 (1/Age) + 0.07 (SI)	0.94	0.001



*Site index of individual trees calculated according to Beck's (1971) equation for polymorphic curves.

Figure 3. Comparison of tree biomass predictions with age, using either site index (SI) or productivity index (PI).

Assessing the accuracy of PI using SI lends uncertainty to the results. Problems with SI have frequently been reported in the literature (Monserud et al. 1990). However, the results of this model assessment indicate that PI can predict site quality with an acceptable accuracy.

Previous research has shown that as a measure of site quality the PI performs better in younger stands than in older (Gale et al. 1991). It is a better index of site quality in younger stands because it uses the effects of soil characteristics on vertical root distribution. The PI model is based on mineral soil characteristics and is more applicable in younger stands that have not accumulated thick organic layers. As a stand matures, a litter layer accumulates, shifting the rooting habits of the tree from extracting nutrients from the mineral soil to an increased emphasis on nutrients in the forest floor (Wells and Jorgensen 1975). This was attributed to the depletion of mineral soil nutrients with increasing stand age and the increase in available nutrients in the forest floor (Wells and Jorgensen 1975).

A tree's vertical root distribution will change with age and with soil properties. Coile (1935) determined

that a tree's root system will reach its maximum depth by age 10. Our developed PI model incorporates the interactions among soil properties in addition to interactions between soil properties and roots. The PI model has the potential of being applied to a wide range of forested and nonforested sites and in sites that are too young or old to use SI. It also allows the user to predict the influences of soil modification on root growth and subsequent productivity (Larson et al. 1983). By applying values to individual sufficiency factors that depict modified soil conditions, a different PI can be calculated and compared with that from an unmodified soil.

Conclusions

Tree vertical rooting patterns affect the aboveground productivity. The PI model integrates the genetic rooting potential of a tree and its response to varying soil and topographic conditions. Sufficiency curves are used in the PI model to describe a tree's root response to individual soil/site properties, relating root growth responses to aboveground productivity.

The PI can be used on forested and nonforested sites, and in stands that are too young or too old for traditional SI estimations. The profiles of sufficiencies can be used to identify layers that are limiting to root growth and productivity, as well as the magnitude of those limitations.

The developed PI model appears to have the potential of being used as an estimator of site quality for white pine on reclaimed strip mines. PI was significantly related to SI and biomass. Using a Schumacher equation to test the effect of the index added to age, SI predicted better than PI. This was a result of multicollinearity between age and PI. Although the sample size for this analysis was small, these reclaimed minesites represent a wide range of site qualities and soil characteristics. The high degree of accuracy of prediction of white pine SI and tree biomass indicates that there is a practical application for applying this model on reclaimed minesites.

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THE RESTORATION OF SPECIES-RICH GRASSLAND ON RECLAIMED LAND[1]

Anthony D. Kendle[2]

Abstract: Restoration projects in the United Kingdom are increasingly targeted toward the reestablishment of land of nature conservation value, as loss of natural habitat is now a major concern when granting planning permission. Many of the techniques used for habitat creation are of limited value as they focus unduly on the introduction of plants in patterns which simulate the original countryside. However unless the processes that support the development and maintenance of biodiversity are also restored, there can be no confidence in the long term value of the mitigation. This paper explores some processes that contribute to the development of diverse plant and animal communities and the ways in which these can be addressed in the practice of land reclamation. In the case of species-rich grassland this involves creating a diverse environmental framework, managing nutrient levels, and devising appropriate long term management. Allowing natural colonisation also helps to restore the abstract conservation value of naturalness.

Additional Key Words: habitat creation, wlldflowers, environmental impact assessment, conservation planning.

Introduction

The implementation of legislation requiring environmental impact assessment in the United Kingdom has coincided with a growth in the technology of habitat and ecosystem restoration. One of the specific requirement of an environmental assessment is the identification of appropriate mitigation measures that will minimize the harmful effects of a development. Developers are also finding that nature conservation issues frequently present major barriers to the process of obtaining planning permission. This has inevitably given a boost to the practice (or attempted practice) of habitat restoration. In some instances developers have obtained planning permission as a consequence of promises to restore wildlife habitats.

Before an appropriate habitat restoration methodology can be developed, it is essential that clear goals are recognized from the start. Possibilities include -
Species-targeted programs e.g. aiming to reestablish one, or a few, identified flowering plants with little concern for other components of the system.
Community-targeted programs e.g. restoring specific vegetation complexes identified by predisturbance surveys.
Maximum diversity i.e. encouraging the widest possible range of plants and animals, but with little concern for the detailed make-up of these communities.
Maximum popular appeal i.e. working with easy, common and showy species for public enjoyment. Aesthetics, location, and access may be more important than actual plant community makeup. A relatively low species diversity can be acceptable if the visual impact is satisfactory.

Much early research into habitat *creation* in the United Kingdom was specifically concerned with the latter goal, creating attractive species-rich vegetation but using common, resilient plants in high profile situations such as urban parks (Wells 1983). The work was of indirect value to nature conservation in that it could alleviate pressure on existing seminatural habitats, bring more people into contact with nature, and encourage support for wider conservation objectives. However, the use of rare species in such initiatives was positively discouraged.

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[2]Anthony D. Kendle, Lecturer in Landscape Management, University of Reading Department of Horticulture and Landscape, Reading, Berkshire, United Kingdom.

In contrast, *mitigation* or *recreation* work often aims to rescue, or compensate for the loss of, species or communities of local or national importance. **The methodology appropriate to these two situations will fundamentally differ.** However, landscape architects, ecologists, and restorationists working in mitigation have often been misdirected by the, widely publicised, technology developed for amenity vegetation development and have failed to address deeper issues associated with the development of habitats of conservation value.

Research onto the creation of wildflower-rich communities has been carried out at Reading University and on reclaimed industrial land for over ten years (Kendle 1992; Bisgrove 1988). This work has confirmed the ease with which it is possible to create simulations of a wide range of plant communities on sites which have previously been much less diverse. The technology employed has obvious uses, but the critical issue concerns the reasons why these communities were previously depauperate given that the environmental conditions lie within the range of physiological tolerence of the plants introduced. Were they limited by the arrival of propagules, by lack of opportunities to colonise, by factors which limit reproduction, or by factors which will damage even established plants but which recur unpredictably? It can be argued therefore that these trials, by succeeding, point towards the weaknesses rather than strengths of current habitat creation methodology. Rather than reproduce here any of the data which have been collected from trials using wildlflower seeds and plants, the intention is to look towards the need for a change in research philosophy and in restoration practice. What is needed is a more assessment of how natural habitats develop conservation importance, and how these principles can be converted into techniques which can be employed in the habitat creation process.

When Is Habitat Restoration Successful?

Given the complexity of the species interactions we can never *prove* to have succeeded in a habitat restoration project; i.e. that everything has survived and will continue to function as desired. Simberloff (1992) suggests that as long as it is impossible to tell that a restored system differs structurally or functionally from an undisturbed system, we can, conditionally, accept that successful restoration has been achieved. Most projects will not achieve even this goal, and some loss of quality is likely. Although this does not remove the justification for the attempt, it makes it hard to assess what is reasonable for the mitigation program to achieve. However, the objective should be to produce a habitat of lasting value and importance.

Assessing the success of a mitigation scheme implies that we could fully characterize the system before disturbance but grasslands are dynamic systems constantly open to change. All of the species exist in an intimate web of competition and mutual support. The more precisely defined or sophisticated the target community is, the more difficult it will be to replicate. Some restoration projects rely upon simple measures such as species composition before and after disturbance. However to match an original grassland these species should also have the desired population numbers and also be found in the "correct" mosaic and pattern of distribution. There should also be an absence of any detrimental species that threaten the integrity of the system, e.g. invasive weeds. Of course, as well as the higher plants all other essential components of the ecosystem need to be present (invertebrates, fungi, etc.) and many of these groups are ignored, or are almost impossible to study, in site surveys.

Assuming a community could be restored which has the desired patterns and abundances of componant species, is this proof of success? Again, there has to be consideration of the likely processes and changes that will occur in the ecosystem over time. Under the prevailing environmental conditions and the management planned, will these communities continue to thrive, and if not how quickly will deficiencies in the restoration become apparent? Some species have a short lifespan and rely upon new generations. If there is a lack of correct regeneration opportunities these will show problems quickly, but others may survive vegetatively for over 100yr before recruiting a new generation. The planning system requires rapid assessment methods, but we also need long-term monitoring programs to determine how stable created grasslands are. Ideally, experience will allow researchers to identify those species that occur early and that are good indicators of long term success.

What Is Required To Restore Conservation "Importance"?

It does not follow that a restoration program that fails to match the pre-disturbance conditions will necessarily lack any nature conservation value, indeed it may at times be possible to make significant gains for local wildlife by a fundamental change in landscape and land use. However, it is often less well understood that it may be possible to exactly match the conditions and communities previously found on a site rich in wildlife, and yet there could still be a decline in the importance of the habitat in the eyes of conservation planners.

In any attempt to create or restore habitats of conservation importance, it seems sensible to take as a reference the criteria that conservationists would use in assessing the importance of existing habitats. In the United Kingdom the guiding criteria used by the main government organization charged with nature conservation were outlined by Ratcliffe (1977) and have been used as a basic framework by conservation bodies ever since. Most of these criteria are pragmatic and obvious. Many can be applied as easily to the evaluation of a manmade habitat as to an existing seminatural one.

Ratcliffe's criteria for evaluating conservation significance are listed below:

Size	Typicalness
Diversity	Recorded History
Naturalness	Position in an ecological or geographical unit
Rarity	Potential value
Fragility	Intrinsic appeal

A key concept is that of naturalness. A habitat that has been heavily modified or influenced by man is regarded as of inherently less value than one that has developed in response to purely natural processes. This idea embodies many concepts, such as irreplaceable information value. An unmodified environment is scientifically worthy of study, and although it is always possible to introduce a human influence, it is impossible to ever erase it. There is also a cultural importance; in many people's minds a habitat that is exposed to human influence loses some spiritual quality, a touch of 'wilderness' perhaps, and is hence degraded. Not surprisingly therefore, the greatest limitation to habitat creation in the eyes of traditional conservationists is that the new or restored community, no matter how superficially successful and viable, is devoid of "naturalness".

These attitudes may change. Global environment change and nutrient pollution present a great challenge to a noninterventionist stance. Although species migrations in response to climate have occurred many times in history, the rapidity of the changes sometimes predictted, and the intensity of United Kingdom land use, which fragments and isolates seminatural communities, means that positive intervention may be necessary. It is likely that restorationists may have to influence plant and animal distribution in a much more active way than ever before, but it is also vital to find ways of doing this in a way that maintains or repairs the emotional and cultural values that people attribute to the land.

How To Restore 'Naturalness'

In a small, intensively exploited country like the United Kingdom there are no areas of land that have had no human influence, and evaluating "naturalness" becomes a question of assessing the degree and severity of interference. Most communities of high conservation value are rated as seminatural, which is usually taken to mean that although mankind has intervened in some way with the management or functioning of the ecosystem, no direct control has been exerted over the species which are found there. It is interesting in this context to note that nature reserve managers feel able to use different maintenance operations to manipulate and artificially diversify plant communities in their care, yet they would feel horrified at the suggestion of deliberately introducing plants. The psychology of conservation is such that humans are allowed to modify the environmental and management framework that allows certain patterns of species diversification to occur, but nature must be allowed to do the work of colonizing this framework.

In restoration work the option therefore exists to concentrate upon the development of the environmental framework rather than the introduction of seed or plants. In Holland Londo (1994) has developed a system whereby habitat creation is achieved entirely by site manipulation, importing different soil types, spreading these in varying depths and aspects and with diverse topography, and with diverse management overlaid. Much of the interest comes from waiting to see what comes up naturally. The value of this approach is that it -

1. focuses attention on the importance of modifying the environment.
2. utilizes natural processes to make the end result more "valid" and acceptable to many traditional conservationists.

Does A Restoration Project Need Plants?

Natural colonization is often overlooked as an option in restoration and the professionals involved rely upon transplanting or seed sowing instead, but it may have technical as well as cultural justification. The planning process tends to assume that landscapes are instantly creatable. However, there is a danger that setting the wrong standards and demanding early performance will encourage the use of exactly the wrong techniques for habitat creation. Certainly Londo identifies an inverse relationship between the speed with which a habitat revegetates and the potential development of botanical diversity in the long term.

It is well understood that stress in habitats can allow the establishment of unusual species with specific requirements and may also limit the growth of competitive plants more severely than it does that of many poor competitors (Grime 1979). Transplanting and seed sowing are horticultural techniques, and not surprisingly initial establishment success is greater when conditions are improved in line with horticultural practice, i.e. when a uniformly fertile and moisture-retentive growing medium is provided. However, the creation of this uniform and favorable habitat, while increasing early plant establishment success and hence high ratings by planning authorities, may in the long term undermine the heterogeneity and stresses that allow the sward to maintain species-richness.

Of course the fragmentation of habitats may make natural colonization too slow to be acceptable. A compromise may be to establish some areas quickly using plants, but to allow opportunities for continued invasion. This approach again creates the risk that communities could be created that have no long-term integrity, but that exist long enough to pass initial evaluation.

There is inevitably an interaction with the management intensity. Domestic and botanic gardens testify to the fact that plant species taken from a variety of habitats around the world and placed in a range of combinations of species and vegetation structure can, once established, thrive and grow for as long, or sometimes longer, than they do in the wild. They are therefore tolerant of wide variation in edaphic and climatic factors. This contrasts sharply with the ecologists' perception of the highly subtle variations that can naturally control plant distribution. The most obvious factor that differs in an intensively managed landscape is competition which is controlled by weeding. To avoid providing false results it is therefore vital that the aftercare period following restoration does not involve prolonged reliance upon a management type or intensity that could not be realistically maintained in the long-term.

Competition is in fact most serious at certain vulnerable stages in the life cycle. Weins (1977) proposes the concept of the ecological "crunch". This suggests that plant distribution is controlled by occasional very severe periods of environmental hostility rather than by prolonged low-level adversity. The horticultural manager's role is often to nurture desired plants through such critical stages. The most likely time for a "crunch" is when a plant is young; the seedling stage is when plants are most vulnerable and show their most specialized ecological requirements. Therefore many problems will only become obvious when a second plant generation is being established.

If natural colonization is allowed to take place, it is at least an initial test that environmental conditions support the juvenile as well as adult stages. Of course, even then there are other reasons to doubt whether the community that establishes early on will reflect that which will survive in the long term. For example, in a recreated grassland, soil disturbance could affect mineral release and fertility, at least temporarily promoting the growth of nutrient demanding species. Weed germination will increase, and unfilled niches in the sward will allow these weeds to become established.

The population of associated animals and microflora will also be badly affected. Some plant species may depend upon mycorrhiza, pollinators, or seed dispersal agents that may not initially be there and pests may be present in higher numbers until controlling species also invade.

Building An Ecosystem That Has The Potential For Diversity

Given the problems that exist with a pattern-oriented approach to habitat creation, despite the obvious appeal appeal for planners, it seems preferable to address the processes and underlying environmental factors which support and maintain the functioning of the desired ecosystem. People working on agricultural restoration recognise that the most important task is to improve soil fertility and drainage before attention can be given to the crops. The same principles must hold true for habitat restoration. So what are the key factors controlling diversity in plant communities?

A fundamental tenet of plant ecology is the Competitive Exclusion Principle, which argues that under any given set of environmental conditions there will be one species that is physiologically and ecologically favored and that would eventually come to dominate the community to the exclusion of all else. Diversity exists where there are factors that mitigate against the trend to monoculture. Not surprisingly several theories have been developed to attempt to identify these. Crawley (1986) identified the following main models, which he emphasizes are not necessarily mutually exclusive; indeed many are closely linked processes. The task of the restoration ecologist is to determine which of these mechanisms have practical implications for the design, construction, and management of restored environments. Many of these models invoke mechanisms that prevent the plant community from reaching equilibrium, i.e. they assume that perfect competition is not possible in a community that is in some way disturbed. Environmental fluctuations allow species to grab transient opportunities before their competitors can reinvade or exert their full dominance. Other models attempt to identify processes or factors that can maintain species-richness within more stable ecosystems.

The Lottery Model assumes that establishment microsites appear randomly, and that colonization is opportunistic and random, reflecting the timing and method of propagation relative to gap formation. Once established, the individuals may remain for a long time.

Spatial Heterogeneity models suggest that apparently uniform areas do in fact contain many subtly different microsites that favor competition by different species. The idea is embodied in the Regeneration Niche ideas of Grubb (1977). In a classic ecological experiment Harper et al (1969) showed that very minor alterations in soil surface texture were enough to control the germination of different plant species.

The Musical Chairs Model argues that if the overall environment is patchy such that different species live close to each other, migration between patches may occur at a rate that balances the rate of extinction of the less competitive types in each location.

Niche Separation models argue that in many cases plants avoid or reduce the severity of competition by exploiting sufficiently different resources or sources of resources in the habitat. As with many such models, arguments that can easily be followed for one dimensional resources become very complex when attempting to understand interactions across all possible resources, as presumably direct competition for just one resource may be enough to lead to exclusion if that resource is fundamental to survival.

Selective Herbivory may be important in some settings. Although in some cases herbivory may select against diversity, by eradicating palatable species, it may also be expected to increase diversity if the dominant competitors are edible. If nothing else, frequency dependent feeding may influence the strongest competitors most seriously and hence give an advantage to rare plants. "Keystone" predators may therefore be necessary to maintain a certain plant community (Paine 1966).

Refuge models propose that species-richness will be enhanced if poor competitors have some locations where they can survive free from competition and can act as a source of propagules to exploit opportunities. Obviously the refuge does not need to be very close if the species have very effective dispersal (e.g. orchids). The survival of poor competitors may be achieved by many mechanisms ranging from ability to tolerate extreme conditions that others cannot and herbivore resistance, through to the existence of long lived seed in the soil bank.

Disturbance models refer to the destruction of existing vegetation, often associated with soil disturbance that allows new species to invade. Obviously there are similarities with the lottery model, but the emphasis is on the effect of the rate of disturbance. At high disturbance only a few species will be adapted to survive, at low disturbance competitive exclusion occurs, but at intermediate disturbance levels species-richness can be high (Grime 1979).

Grime also proposes that similar principles apply with regard to stress, or environmental adversity that limits plant productivity. He suggests that species-richness is often associated with moderately hostile conditions. High fertility is of course recognized as a threat to the maintenance of diversity in many ecosystems (Marrs and Gough 1990). The development of diversity may therefore depend upon an understanding of the appropriate nutrient 'capital' for the biome in question (Roberts et al 1981) and of how to maintain low level nutrient cycling in the new ecosystem (Kendle 1992). Reclamation technology is often geared toward building the maximum fertility, which can be much easier than aiming at a specific level of nutrient cycling.

Of course the above models refer to situations where the diversity is great within a sample of a plant community, but different samples may be similar. Whittaker (1975) refers to this as alpha diversity. However, overall diversity may also be great in situations where environmental heterogeneity leads to the creation of a complex mosaic of community patches, each of which, individually, need not necessarily be species-rich. Samples may therefore be low in species, but differences between samples are great. This latter situation is referred to as beta diversity. Obviously in a restoration project the appropriate patterning of species distribution should be determined and emulated for each target community, but it is worth recognising that designing and engineering beta diversity can be much simpler than the task of creating alpha diversity .

In amenity situations high alpha diversity is not necessarily the best objective to set. In trials carried out at Reading University, wildflower seed mixtures were sown onto a range of different soil types. High ratings for visual preference were often most closely associated with the swards of lowest biodiversity. Aesthetically people often prefer situations where individual species have a strong visual effect as a result of forming large vigorous free-flowering clumps. These conditions are often characteristic of environments that show strong beta diversity and poor intermixtures of species. Similarly some of the characteristics that may help to maintain high alpha diversity, such as grazing or disturbance, may not result in a sward that is visually pleasing.

When considering the zoological components of the habitat, it must also be recognized that diversity is not always linked to the same criteria as is the case with plants (Kirby 1992). For example, some ground nesting bees require bare soil and are in particular danger of their habitat needs being ignored in any predisturbance survey, whilst effective restoration from the bee's perspective may be rated as poor establishment by the monitoring authority!

Some animals are specialist feeders and will only survive if their host plants are present. High plant diversity will increase diversity in these animals. However, insect diversity is often associated with diversity of physical structure as well. We also find that some insects may not be very effective at dispersal and may have less successful methods of surviving prolonged adversity compared with plants. Not surprisingly these vulnerable species are often the ones that are under greatest threat and hence rate highest in conservation terms. Because of their specialized life cycles the concept of refuges becomes of acute importance for their survival. To encourage insect diversity, a management framework (e.g. rotational burning) needs to be imposed that allows different plant communities to express a range of different morphologies and also allows the animals to avoid overall destruction.

For mobile species the concept of environmental heterogeneity takes on additional complexity, and the habitat mosaic within the entire ecosystem to be restored must be considered. If two habitats, e.g. a grassland and a woodland, are created adjacent to each other, then diversity within the overall area may be increased by several means. Most obviously the site will contain species that need either woodland or grassland conditions. There will also be species that require intermediate conditions provided within the ecotone, and those that depend on both of these habitats in order to survive. In the latter case this may include, for example, animals that shelter or nest in woodland and feed in grassland and insects that require different habitat conditions at adult and juvenile stages in their life. Habitat mosaic will also influence the patterns and rates of dispersal of different species.

Habitat mosaic is a complex subject that conservation ecologists are only just beginning to explore in depth, and the species that depend upon the existence of certain *patterns* within the countryside rather than any specific habitat are certainly the cinderellas of natural science. We are a long way from being able to develop consistent rules that can be applied with confidence in restoration projects. However if this goal could be accomplished, the construction of a diverse mosaic of landscape types is clearly open to relatively easy manipulation by landscape designers and engineers during land reclamation programs. Of course it must also be remembered that if an area of land is diversified into several different biotypes, there may be species that have extensive area requirements for feeding and that may disappear because of the fragmentation of the extensive landscape they need.

Applying These Principles To The Restoration Of Species-Rich Neutral Grassland

In the light of the above discussions, it is possible to highlight some technical solutions suitable for specific goals, although in many cases there is still substantial research to be done to convert concepts into practice. The restoration of neutral grassland presents an interesting case study because it challenges our abilities to the extreme and yet is politically highly desirable. In the United Kingdom species-rich neutral grassland communities are often associated with alluvial soils in river valleys. They were usually managed as hay meadows and were prone to periodic winter flooding. They were frequently productive because of the high fertility of the alluvial deposits but had a short season of usefulness. Harvesting of the sward to make hay was typically carried out from midsummer. This may have been followed by aftermath grazing or by further hay cuts. These habitats are under various threats, including improvements to land drainage technology, greater control of river water levels, aggravated by abstraction of drinking water and herbiciding or ploughing and reseeding in order to improve the grassland mixtures.

These areas are also under pressure for extraction of sand and gravel, primarily for general construction and development. Because the economic transport distances for aggregates are so low, extraction is often undertaken in areas that are under pressure from development so that the likelihood is even greater that the remaining areas of grassland are of high local conservation importance. Wherever aggregate extraction directly damages a surviving meadow, it would now be almost impossible to conceive of a situation where planning obligations did not involve habitat restoration. Even where the gravel extraction takes place on areas where the habitat has already been degraded, it is often seen as desirable to attempt to recreate more meadowland.

Understanding and repairing the dynamic processes that preserves species diversity on such sites is of course essential. Neutral meadows are interesting because they are fertile and yet diverse; this is not a common situation in the United Kingdom flora (Grime 1979). In neutral meadows, species-richness is probably maintained by a combination of the stresses imposed by periodic flooding, and the disturbance caused by harvesting, grazing and alluvial deposition that creates recurrent opportunities for recruitment. Patterns of nutrient availability will change dramatically through the year, and this too may limit the growth of demanding perennials.

Techniques Of Meadow Creation

Currently proposals are under discussion with mineral developers to undertake a new phase of neutral grassland restoration research which will be focused on process rather than pattern. The key elements of the restoration will include:-

Habitat and Environment Design. Although the image of hay meadow communities is often one with moderate to high alpha diversity, a riverine ecosystem will often exhibit high beta diversity as well, with many different zones of vegetation reflecting mean water depth through the year, as well as different patches that reflect different frequencies, duration, and timing of seasonal flooding and the fate of the sediment load in the water. The composition of the soil on which these patterns are imposed will also influence the community that develops. Although we are only just beginning to understand the relationships between these variables (Rodwell 1993), if the right principles can be developed, flooded grasslands are open to an engineered approach where landforming, water pumping, or use of sluices can control community composition.

Where possible, therefore, a variety of soil types should be introduced into the site, even if these differ only slightly in composition. The opportunities and need for complex landforming is already recognised by the gravel industry when wetland restoration is attempted and these principles can fairly easily be adapted to terrestrial habitats. Inevitably a highly diverse topography may not be tolerable on all meadow sites, but it may still be possible to consider producing complex subsurface landforms which are then covered to a more even grade by the top layer of the soil profile. This will be particularly effective where a subsoil is used with extreme drainage characteristics, such as a clay or a sand. Site design must also recognize the potential value of a mosaic of different biotypes; the biodiversity of the grassland will probably be made richer by having features such as ditches, ponds or scrub nearby.

Species Introduction. Although plant introduction must not dominate the restoration process, there are still planning pressures that will require a reduced colonisation period. The main options available for introducing plants are -
Seed using either a commercial formulation or a 'haymeadow' mix taken from existing swards of conservation value. Seeding can be onto bare soil or into an existing sward using a slot seeder. Haymeadow mixes may be dominated by whichever species successfully set seed in the sward that season.
Transplants plugs or even pot grown plants. This is expensive and best reserved for those species that establish poorly because of slow or delayed germination. Again transplanting is often carried out into existing swards, although it is possible to plant into bare ground and to sow a low rate grassland mix over the top.
Plant Litter e.g. hay crop remains used as a substitute for a commercial seed mix.
Turves taken from a damaged site for inoculum, dotted into a new site, or as a complete cover.
Soil e.g. spreading topsoil salvaged from a damaged or destroyed habitat.

The latter two are favoured because they will act as carriers of many organisms, ranging from seeds, vegetative propagules, soil fauna, fungi, and other microorganisms. These may be invaluable in establishing a diverse and functioning habitat. They may also help to introduce plants that are impossible to buy through the trade and to preserve local genotypes. However, the proportions and composition of the mix are unpredictable. Low density plug planting is often useful to give a good visual aspect and to introduce those plants with limited seed bank survival without pre-empting all colonisation niches.

Management Of The New Grassland Habitats

Duffey et al (1974) state that "management is often the major factor affecting the floristic composition of a lowland grassland". Management is clearly one source of those environmental factors that encourage diversity by influencing whether refuges exist, the degree of disturbance, nutrient cycling and gap formation that affect alpha diversity. Zonation in management can also reinforce beta diversity and control structural diversity in the vegetation,.

In the majority of restoration programs management, while accepted as important, has been seen as a largely routine and one-dimensional operation by which a target plant community is preserved once created. However, it is also possible to consider management as a primary tool for habitat creation. It is often the sole factor which has dictated whether different biotypes such as grassland or woodland cover have developed over most of the United Kingdom countryside. Some studies have looked at the way different cutting regimes can lead to profound changes in grassland vegetation (Parr and Way 1984). Such maintenance processes should be used in a less ad hoc way so that the habitats that we create are reinforced over time rather than possibly degraded.

Together with environmental design, ensuring adequate and correct management is therefore an essential factor in habitat creation, especially when the target community is a plagioclimax. However, many existing conservation areas in the United Kingdom are in a management crisis following revolutionary changes in farming technology that have undermined traditional practices. It is clearly pointless to create new habitats that also cannot be looked after. Perhaps a worse scenario would be if new habitats dissipated maintenance efforts and increased the threat on relict areas. This is of such fundamental importance that management considerations must be fed back into planning assessment and evaluation of proposals. An existing species-rich habitat cannot be rated as any less valuable whether or not there is a management plan for its long-term survival; a planned habitat could be judged on just such pragmatic criteria.

The techniques available for the grassland manager are of course to mow, to graze or to do both. The timing and the frequency of these operations will also influence the sward composition. The long-term reduction of fertility, through removal of cuttings or folding livestock, is often recommended as a key part of grassland habitat creation (Marrs and Gough 1990). However, in many lowland neutral grasslands it is impossible to significantly reduce soil nutrient stores by this means, particularly since annual flooding may restore fertility. The most important aspect of the removal of cuttings is to reduce the direct physical damage caused by smothering.

The most specific management task associated with new swards is weed control, selectively eradicating those species encouraged by disturbance such as *Cirsium*. In a traditional nature conservation area, the management program will often be oriented towards maintaining a plagioclimax. The major problem weeds often arise from later successional stages, e.g. scrub invading grass. In new habitats the manager may be trying to encourage the movement towards a plagioclimax that has not yet been reached and there may be a need to deal with species that belong to earlier successional stages or that are relicts from a previous land use. The standard solution for weed control is to cut with high frequency in the first year. Some meadow species may not tolerate this and must be introduced as transplants after year one.

The reinforcement of populations of desirable species may also be necessary, requiring reseeding or further planting. Habitat creation schemes are usually based on a single sowing or planting, but research has shown that random or unquantifiable factors can influence colonization, and that several attempts may be necessary before any given species "takes" (Simberloff 1992). Interestingly, many traditional grasslands were oversown with hay sweepings and this may have played a role in maintaining species-richness (Rodwell 1993).

Above all, the development must provide funding for long term management support. Many restoration schemes in the United Kingdom are based on a 5yr aftercare period, and this is completely inadequate. Some developers rely on the naive hope that community or non-governmental groups such as wildlife trusts will take on the maintenance, but these organizations are overstretched looking after their existing responsibilities. Bonds or the establishment of management trusts are essential.

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RAPTOR USE OF REVEGETATED COAL STRIP MINE SPOILS IN INTERIOR ALASKA[1]

Charles L. Elliott[2]

Abstract: Raptors associated with reclaimed coal strip mine spoils on the Usibelli Coal Mine in southcentral Alaska were observed in 1981 and 1982. Of the 10 raptor species identified on the mine, 6 (red-tailed hawk, golden eagle, Northern harrier, American kestrel, hawk owl, and short-eared owl) were observed hunting on the reclaimed areas.

Additional Key Words: Alaska, raptors, strip mine reclamation.

Introduction

The value of strip mined lands as wildlife habitat in the contiguous United States has been known for many years (Yeager 1942, Riley 1957, and review by Elliott 1989). If wildlife options involving surface mine reclamation are to be promoted, their value and feasibility must be established. It must be demonstrated that creation of quality wildlife habitat is an attainable product of reclamation and is a viable land use alternative (Suchecki and Evans 1978). These attributes, as they pertain to raptors, have been demonstrated for surface mined lands in the coal areas of the Eastern and Western United States [e.g., raptors have been reported to use strip mines as nesting sites and hunting areas (Birkenholz 1958, Allaire et al. 1982)]. However, information concerning the relationship between coal strip mine reclamation and raptors in Alaska is lacking.

In 1980, the University of Alaska's Agricultural and Forestry Experiment Station began a study of revegetated strip mine spoils in southcentral Alaska. One objective of the study was to evaluate postmining land use of reclaimed areas by native wildlife (Elliott and McKendrick 1988, 1990). In this study I report the observed use made of revegetated coal strip mine spoils by interior Alaskan raptors.

Methods

The study was conducted during 1981 and 1982 on the Usibelli Coal Mine (UCM). The UCM is located in interior Alaska, approximately 185 km south of the city of Fairbanks and 13 km east of the community of Healy (63° 53'N, 149° 01'W). The Healy area is situated within the northern foothills of the Alaska Mountain Range and is physiographically diverse. Elevations in the region range from 396

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[2]Charles L. Elliott, Professor of Biology, Eastern Kentucky University, Richmond, KY, USA. 40475-3124.

to 914 m ASL. Mean annual precipitation is 43 cm, and mean annual air temperature is -3° C. Vegetation in the study area conformed to four general cover types: conifer forest, tall shrub, shrub tundra, and disturbed/reclaimed.

The conifer forest cover type was a combination of woodland and open and closed spruce forest. The woodland and open spruce forests were located on upland terraces and consisted of sparsely distributed low-growing spruce and low-to-prostrate-growing shrubs. Principal trees and shrubs included black and white spruce (Picea mariana and P. glauca), quaking aspen (Populus tremuloides), mountain cranberry (Vaccinium vitis-idaea), willow (Salix spp.), birch (Betula spp.), blueberry (Vaccinium spp.), and alpine bearberry (Arctostaphylos alpina). The closed spruce forest was characterized by dense stands of mature spruce occurring along drainages, ridges, and terraces.

The tall shrub cover type consisted of a green alder (Alnus crispa)-feltleaf willow (Salix alaxensis) association. This habitat occurred most frequently at the base of north-facing terraces, along creek drainages, and on floodplains.

The shrub tundra cover type was typified by a glandular birch (Betula glandulosa) and ericaceous shrub-sedge association. The disturbed/reclaimed cover type represented sites on the mine that had been disturbed by mining and reclaimed with graminoids and forbs.

Coal mining had greatly altered the soil profile on the disturbed and reclaimed areas. After the coal was extracted from an area, the combined topsoil and excavated subsoil (the overburden) were redeposited, and the area was graded back to approximate original contour, scarified, furrowed, and aerially seeded and fertilized. As a result of seed mixtures composed mainly of graminoids (Elliott et al. 1987), the resulting disturbed and revegetated areas resembled expansive meadows dominated by introduced grasses (mainly Alopecurus pratensis, Festuca rubra, and Calamagrostis canadensis). Although the disturbed and reclaimed areas represented sites reclaimed between 1972 and 1982, natural reinvasion of the areas was practically nonexistent (Elliott et al. 1987). At the time of this study there were approximately 1,000 ha of disturbed and reclaimed land present on the mine area.

Raptor species observed on the mine area in the course of daily year-round field work were recorded. The vegetation cover type in and/or over which each bird was observed was recorded and the activity of the raptor during the observational period was noted. If a bird was observed to capture a potential food item, the bird was followed (if possible) and a variable-power spotting scope was used to identify the prey item. Regurgitated "castings" or "pellets" (Moon 1940) collected beneath the perches of two species of raptors were analyzed following the procedure of Glading et al. (1943).

Results and Discussion

Ten raptor species were observed on the disturbed and reclaimed areas of the Usibelli Mine: red-tailed hawk (Buteo jamaicensis), golden eagle (Aquila chrysaetos), goshawk (Accipiter gentilis), sharp-shinned hawk (Accipiter striatus), Northern harrier (Circus cyaneus), merlin (Falco columbarius), American kestrel (Falco sparverius), hawk owl (Surnia ulula), boreal owl (Aegolius funereus), short-eared owl (Asio flammeus). No birds nested on

reclaimed areas, but six species were noted using revegetated regions for hunting.

A single short-eared owl spent the 1981 summer season on the Usibelli Mine. By observing the owl when it was perched, I was able to locate eight regurgitated pellets; 75% of which contained tundra voles [Microtus oeconomus; the two remaining pellets contained red-backed voles (Clethrionomys rutilus)]. The predominance of tundra voles in the diet was consistent with the dietary observations reported for the species elsewhere in Alaska (Pruitt 1959). The majority (80%, N=21) of short-eared owl sightings (and all observations of hunting, N=10) were on or over reclaimed areas. The revegetated areas and their short-statured vegetation were similar to the undisturbed habitats reported to be utilized by short-eared owls in Alaska (Gabrielson and Lincoln 1959:542-545).

Red-tailed hawks were observed over all cover types on the study area. Single birds were observed a total of 17 times during the summers of 1981 and 1982. Two birds together were sighted on three occasions (once in 1981, twice in 1982). All sightings of multiple birds occurred when an arctic ground squirrel (Spermophilus parryii) had been captured by one hawk and an in-flight prey exchange, or theft, took place between the birds. Individual red-tailed hawks were observed with prey items on four occasions: three instances when hawks had captured arctic ground squirrels and one observation of a hawk in flight with a snowshoe hare (Lepus americanus) in its talons. Based on these limited observations, red-tailed hawks in the Usibelli area seemed to prey on ground squirrels to a greater extent than hawks observed by Lowe (1978) elsewhere in interior Alaska.

Hawk owls were observed associated with the disturbed and reclaimed mine areas five times. Two sightings were of birds capturing food items (voles) on a reclaimed area; on three separate occasions birds were observed perched on top of dead snags along an infrequently used haul road.

During the summers of 1981-82, golden eagles were observed associated with the study area on 31 occasions. All sightings were of solitary birds. Eagles were located in all cover types, but observed hunting activity was concentrated at an arctic ground squirrel colony located in a revegetated area. The apparent dietary dependence of golden eagles in interior Alaska on arctic ground squirrels has been reported by Murie (1944). Eagles were observed capturing ground squirrels from the colony three times; in addition I twice saw single birds flying over the disturbed and reclaimed areas carrying other types of prey. One bird had a snowshoe hare; the other eagle was holding a gallinaceous-type bird, which I assumed to be a willow ptarmigan (Lagopus lagopus).

One pair of kestrels were observed on the reclaimed areas during each of the summers of 1981 and 1982. The birds were sighted over all cover types, but the vast majority of sightings (78%, N=87) were over open habitats (i.e., revegetated areas and shrub tundra).

Pellets collected from a male kestrel's feeding perch located adjacent to a reclaimed area revealed that kestrels in the mine area consumed mainly arthropods (frequency of occurrence = 86%), small mammals (75%), and birds (36%) (Elliott and Cowan 1983). Young and Blome (1975) reported the food habits of kestrels in northern Ontario, Canada, and noted that (based on frequency of occurrence) invertebrates comprised 76% of the diet, birds 20%, and mammals 2%.

It was postulated that the greater incidence of mammals in the diet of kestrels associated with the Usibelli Mine was due to the greater availability of small mammals on the revegetated areas. Tundra vole densities on the revegetated sites were twice the densities found on nearby undisturbed tundra (Elliott 1984).

Northern harrier preference for meadow-inhabiting birds and small mammals (Beebe 1974) and the birds' propensity to nest in grassy habitat were thought to account for the species' high use (85% of observations, N=64) of revegetated sites in 1981 and 1982. On three occasions a female was observed in a revegetated area as she dove to the ground in what I interpreted as attempts to capture prey. However, no diet information was collected for harriers.

Two species of accipiters, the goshawk and sharp-shinned hawk, were sighted on the mine area. The goshawk frequented the revegetated areas adjacent to a natural lake. A pair of sharp-shinned hawks nested in a creek drainage adjacent to a revegetated area during the summers of 1981 and 1982. Sharp-shinned hawks were mainly sighted flying along the drainage, but in 1981 individuals were observed on four occasions over revegetated areas. In every instance, the hawk was chasing or harassing a short-eared owl. Merlins and boreal owls were never observed hunting on the reclaimed areas, but were sighted both years (11 and 3 total sightings, respectively) flying over the revegetated sites.

The effect of current coal strip mine reclamation techniques on interior Alaskan raptors will depend on the size and shape of the disturbed area and the amount of undisturbed forest present. Short-eared owls and marsh hawks, because of their preference for open terrain and a microtine diet, derive the greatest benefit from current reclamation methods.

The major problem associated with mine reclamation for most interior Alaskan raptors is the loss of nesting habitat. Coal extraction methods presently employed by the Usibelli Mine result in the disturbance of small areas of land at any one time. There were always areas of forest adjacent to, or in close proximity to, a reclaimed area. These stands of variously aged trees provided the nesting habitat required by most interior Alaskan raptors. Boreal owls in Alaska nest in cavities in closed-canopy deciduous forest, while hawk owls nest in open-canopy forests or on the edge of extensive cleared areas--the nests being in the tops of tree stubs (Meehan and Ritchie 1982); sharp-shinned hawks nest in mixed stands of deciduous trees with conifers scattered throughout (Clarke 1982); red-tailed hawks tend to nest in mixed stands of deciduous trees (Lowe 1978).

Although the data presented in this article are derived from a limited time period and small sample size, it is my hope that the information (and evident lack of information) concerning reclaimed strip-mined lands and raptors in interior Alaska will serve as a stimulus for further studies. Surface mining activities that remove a majority of the forest in an area, and subsequent reclamation activities that create large expanses of treeless grasslands, could be very detrimental for most Alaskan raptors. The mining of what has been estimated to be 8.7 billion st of coal located within the interior region of Alaska (National Research Council 1980) could result in the disturbance of thousands of hectares of tundra and forest. Avoidance of the potential negative impact such large scale mining could have on raptors will require a better understanding of the role reclaimed strip mines can play in meeting the habitat needs of raptors in the northern boreal forest.

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ECOLOGY OF A BRAZILIAN BAUXITE MINE ABANDONED FOR FIFTY YEARS[1]

Jackeline S. Lorenzo[2], James J. Griffith, Ivo Jucksch, Agostinho L. de Souza, Maria G. F. Reis, and Antônio B. do Vale[3]

Abstract: This study analyzed natural regeneration on bauxite-mined land to learn how mining affects plant communities and to identify appropriate woody species for site reclamation. The site chosen in Poços de Caldas, Minas Gerais State, had last been mined 50 yr ago, by hand. Such a long-standing site, abandoned but free of further ecological disturbance, is rare in the tropics and subtropics. Three site conditions were studied: heavily mined, lightly mined, and adjacent undisturbed terrain. Mined sites were found to have soils of sandy clay loam texture; undisturbed soils have clayey texture. Chemical characteristics also differ, especially aluminum (Al^{3+}), which is much lower on mined sites. Woody species show more exuberance on heavily mined sites with family Melastomataceae showing the highest index of importance value; family Compositae was highest on undisturbed sites. Considering that mining usually degrades soil, it might seem surprising that, for the three sites, the most copious vegetation occurs on the most heavily mined site. It is suspected that bauxite mining, in some cases, may improve soil chemistry by removing aluminum. The hand mining done 50 yr ago also created pits that accumulate water and sediment during rains, which may be useful to woody species.

Introduction

A bauxite mine called Campo do Saco, located in the municipality of Poços de Caldas in southern Minas Gerais State, provided a rare opportunity to study regeneration of woody species on long-standing degraded lands. This mine started production at the outset of the 1930's. Excavation by pick and shovel continued during the 1940's until the mine was finally abandoned with no attempt at reclamation. Despite this abandonment, a considerable quantity of the area's bauxite remained untouched. Under new ownership, the mining company periodically opened small excavations in the area until 1971, as part of its legal obligation to maintain the mineral concession.

Intending to reopen the mine in 1991, the company decided to do a model job by preplanning reclamation. It commissioned, among others, a phytosociological study of existing vegetation to predict environmental impacts from reopening the mine. The objective of this paper is to analyze how vegetation at Campo do Saco has changed in relation to soil characteristics modified by mining. It was done by the Department of Forest Resources, Federal University of Viçosa, in cooperation with the company that operates the mine.

Various authors have described mining's environmental impact in temperate regions. Changes in naturally established plant communities have been reported by researchers such as Johnson et al. (1982), Manner et al. (1984), and Smith et al. (1988). Palaniappan's (1974) work on Malaysian cassiterite mines is one of the few regeneration studies on abandoned sites in the tropics.

In general, these and other studies point to the following hypotheses for the Campo do Saco situation:

1. Degradation caused by mining usually includes substantial alteration of the original, native vegetation and soil structure. Mining in the Poços de Caldas region may not be so harmful as undisturbed vegetation overlying bauxite deposits in the region tends to be low and sparse.

2. Disturbed soil characteristics after mining should largely explain characteristics of plants found on excavated sites. For this study, this relationship will probably have to be understood within the context of soil conditions typical of the "cerrado" phytogeographic region, which marginally includes the Campo do Saco study area.

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[2]Jackeline S. Lorenzo, Biologist, Department of Environment, Oficina Nacional de Planificación, Santo Domingo, Dominican Republic.

[3] James J. Griffith, Professor of Forestry, Ivo Jucksch, Assistant Professor of Soils, Agostinho L. de Souza, Professor of Forestry, Maria G. F. Reis, Professor of Forestry, and Antônio B. do Vale, Professor of Forestry, Universidade Federal de Viçosa, Viçosa, MG, Brazil.

3. If undisturbed after cessation of mining, a natural succession of plants will probably occur on the abandoned sites. However, it is not known if 50 yr at Campo do Saco has been enough time for the reestablishment of its original ecological communities. Neither is it known if a reclamation project could achieve, even in the long term, perfect biological restoration, given the degree of degradation. An analysis of naturally occurring regeneration could, however, reveal promising candidate species for the area's future reclamation.

Material and Methods

Campo do Saco is located in southern Minas Gerais State, Brazil, between 21° 15' 20" latitude south and 46° 33' 55" longitude west, in a region characterized by mild subhumid mesothermal climate with two dry months. Average annual precipitation is 1,722 mm, January being the hottest month (20.3° C average) and the rainiest (297.21 mm average); July is the coldest (13.6° C) and the driest (25.7 mm). Frosts may occur in winter.

Semideciduous forest ("floresta subcaducifólia") and grasslands ("savana gramineo-lenhosa") represent the vegetation cover of the region (Brasil 1993). On more fertile sites, forest strata may reach 20 to 30 m in height. The grasslands occur above 1,000 m and are covered by herbaceous plants; however, isolated shrubs also occur. The Campo do Saco area is a hilltop covered with native vegetation. Altitude is generally 1,300 m.

Bauxite in Poços de Caldas occurs on ridge tops and is concentrated on the northern part of the region's plateau. The mineral occurs near the surface, covered by bauxite-rich soil about 50 cm deep (Dias 1982).

To determine the area's existing phytosociology and predict impacts likely to affect vegetation when the mine reopens, three study strategies were necessary:

1. Verify the situation of flora at the Campo do Saco area where no old excavations exist and no new mining has yet occurred, as a point of reference for the possible future reclamation objective of reestablishing original plant communities.

2. Compare the plants' situation on previously mined sites with the adjacent undisturbed area, as an indication of the biological conditions that future excavations will leave behind with the reopening of the mine.

3. Investigate how mining intensity affects regeneration conditions for plants. To achieve this, it would be necessary to compare the area lightly mined (small, shallow excavations up to 95 m long and 2.0 m deep) with heavily mined areas (larger and deeper pits up to 150 m long and 3.35 m deep).

To carry out these studies, the Campo do Saco area was classified into three sites: I - unmined (control plots); II - lightly mined; and III - heavily mined terrain. Of the total area, site I occupies 2 ha; II, 2.03 ha; and III, 4.3 ha. Each site was divided into four quadrants, in which were randomly distributed various 10 - by 8 - m plots which represent approximately 2.5% of the total area of each site.

Soil samples were taken at four different points for each site, at six depths: 0 to 5 cm, 5 to 15 cm, 15 to 40 cm, 40 to 60 cm, 60 to 80 cm and 80 to 100 cm. Texture was analyzed for percentages of sand, silt, and clay. Also analyzed were pH in H_2O (1:2.5), concentrations of available phosphorus (P) and potassium (K) using a Mehlich-1 extractor, and exchangeable calcium (Ca^{2+}), magnesium (Mg^{2+}), and aluminum (Al^{3+}) using the 1*N* KCl extractor to determine sum of bases (SB) and effective cation exchange capacity (CEC).

Vegetation sampling consisted of identifying and counting woody species for each plot and then measuring the height and coverage of each individual. Coverage (C_i) for each species was measured using the expression

$$C_i = \left(\frac{L_1 + L_2}{4}\right)^2 \times \pi \qquad (1)$$

in which L_1 and L_2 are the perpendicular lines of crown length and width.

Relative coverage (CR_i) was estimated by the expression

$$CR_i = \frac{C_i}{CT} \times 100, \qquad (2)$$

in which C_i is the coverage provided by the "i"th species and CT is the sum of coverages for all the species in the area.

The parameter estimates for species density and frequency were obtained by using the following expressions:

$$\text{Absolute density } (DA_i) = \frac{N_i}{A}, \tag{3}$$

$$\text{Relative density } (DR_i) = \frac{DA_i}{\sum_{i=1}^{p} DA_i} \times 100, \tag{4}$$

$$\text{Absolute frequency } (FA_i) = \frac{U_i}{U_t} \times 100, \tag{5}$$

$$\text{Relative frequency } (FR_i) = \frac{FA_i}{\sum_{i=1}^{p} FA_i} \times 100, \tag{6}$$

where N_i = number of individuals sampled of the "i"th species,

A = area sampled, ha,

DA_i = absolute density of the "i"th species,

DR_i = relative density of the "i"th species,

U_i = number of sample units in which the "i"th species is present,

U_t = total number of sampling units,

FA_i = absolute frequency of the "i"th species,

FR_i = relative frequency of the "i"th species,

and p = total number of species sampled in the community.

Using these estimates, the ecological importance of each species was calculated, using the Cottam and Curtis index of value importance (IVI) as described by Matteucci and Colma (1982):

$$IVI = FR_i + DR_i + CR_i \,. \tag{7}$$

These measures should help explain the observed dynamics of ecological succession at Campo do Saco and suggest which pioneer species should be planted.

Results

Field data collection was done from September to December 1989, followed by analysis of each site's soil and vegetation.

Physical and Chemical Soil Characteristics

Analysis showed marked differences between soils on undisturbed site I and mined sites II and III, but little difference between soils on the two types of mined sites (table 1).

Table 1. Values for physical characteristics of soil at different depths in sites I (unmined), II (lightly mined), and III (heavily mined).

Area and soil depth, cm	Physical characteristic			
	Sand, %	Silt, %	Clay, %	Texture class
Site I (unmined):				
0-5	19.30	28.00	52.70	Clayey.
5-15	19.70	21.50	58.80	Clayey.
15-40	22.80	17.30	59.90	Clayey.
40-60	12.60	18.30	69.10	Very clayey.
60-80	14.80	14.20	71.00	Very clayey.
80-100	22.80	16.80	60.40	Very clayey.
Site II (lightly mined):				
0-5	60.80	15.80	23.50	Sandy clay loam.
5-15	54.80	14.50	30.80	Sandy clay loam.
15-40	39.80	13.30	46.90	Sandy clay.
40-60	56.25	8.25	35.50	Sandy clay.
60-80	52.80	11.50	35.70	Sandy clay.
80-100	63.00	10.70	26.30	Sandy clay.
Site III (heavily mined):				
0-5	52.75	24.00	23.25	Sandy clay loam.
5-15	59.50	13.50	27.00	Sandy clay loam.
15-40	44.75	15.75	39.50	Sandy clay.
40-60	50.75	10.25	39.00	Sandy clay.
60-80	59.00	11.00	30.00	Sandy clay loam.
80-100	61.00	12.00	26.75	Sandy clay loam.

Mined plots were found to generally have soils of sandy clay loam texture; undisturbed soils have clayey texture. According to Almeida (1977), drainage confined by topography contributes to creation of a clay layer (clayey soil) in ore deposits of this type. This material gets mined out along with the ore.

In terms of acidity (figure 1*A*), the undisturbed soils (site I) present more adverse conditions, with pH varying from 4.8 to 5.3, whereas mined-out soil (sites II and III) pH values range from 5.0 to 5.3 in lightly mined plots and from 4.9 to 5.6 in heavily mined plots. In the three types of sites, pH increased slightly with sample depth.

Results of the CEC analysis also indicated that the undisturbed site is more adverse (figure 1*B*). Values ranged from 0.08 to 2.63 on that site, from 0.05 to 1.75 for the lightly mined site, and from 0.02 to 1.24 for the heavily mined site. The CEC value almost always decreased as samples got deeper; the largest contrasts between the three types occurred at a depth of 5 to 15 cm.

The relation between sum of bases (SB) for the three sites also showed the same tendencies (figure 1*C*): the unmined terrain had highest values (0.03 to 1.79), followed by lightly mined terrain (0.05 to 1.52) and then by heavily mined terrain (0.02 to 1.24).

Although scarce, available P in the soil varied among the three situations (figure 1*D*): 0.85 to 11.0 for undisturbed sites, 2.45 to 16.4 for lightly mined sites, and 1.65 to 15.8 for heavily mined sites. Analysis of aluminum (Al^{3+}) revealed the greatest difference between unmined and mined sites (figure 1*E*). The undisturbed terrain contains more aluminum,

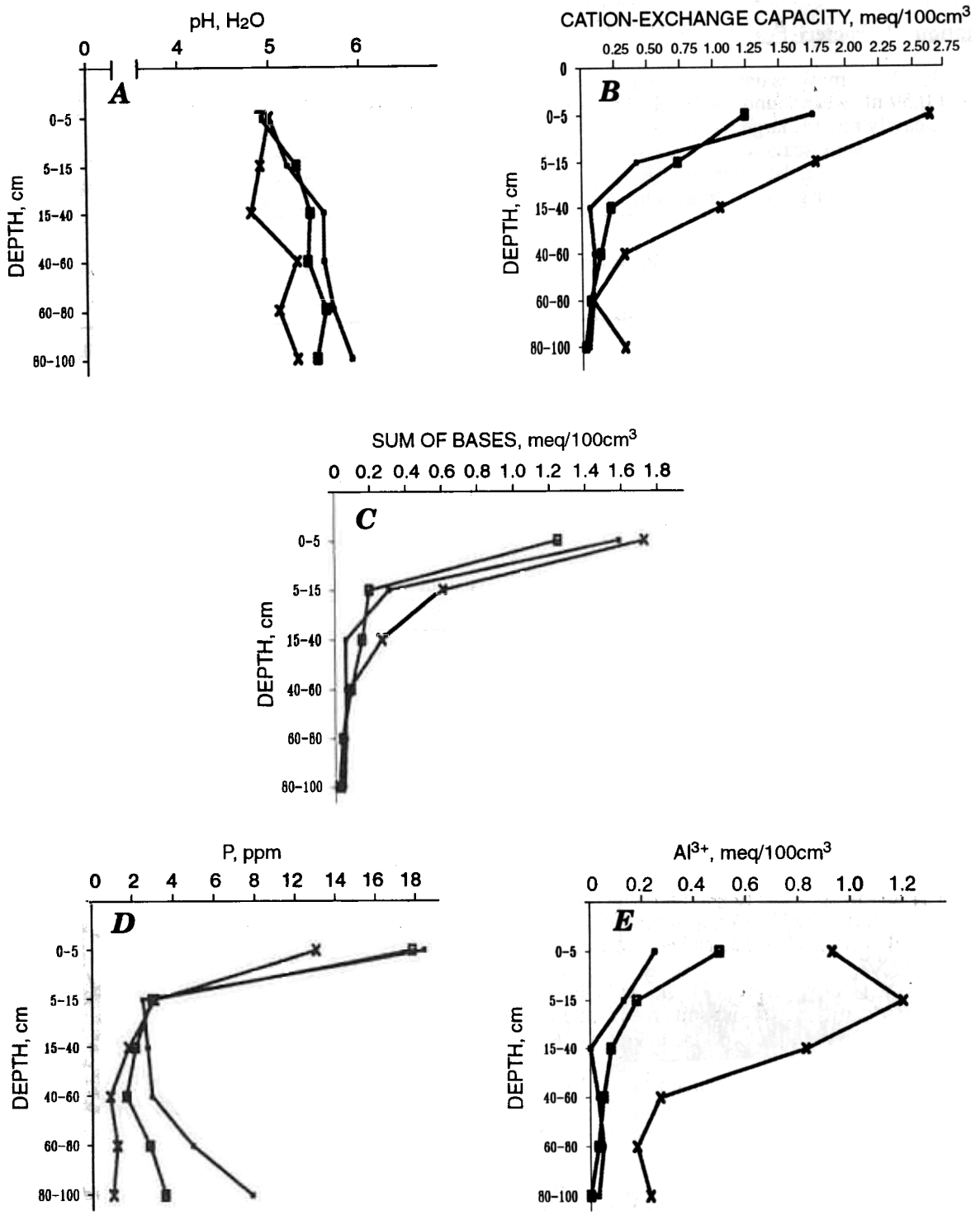


Figure 1. Variation in chemical characteristics of soil according to depth in sites I (unmined) (x), II (lightly mined) (◆), and III (heavily mined) (■) .

ranging from 0.23 to 1.20 meq/100 cm^3. Aluminum concentration reached 0.25 meq/100 cm^3 in shallow excavation pits and 0.50 meq/100 cm^3 in deep pits.

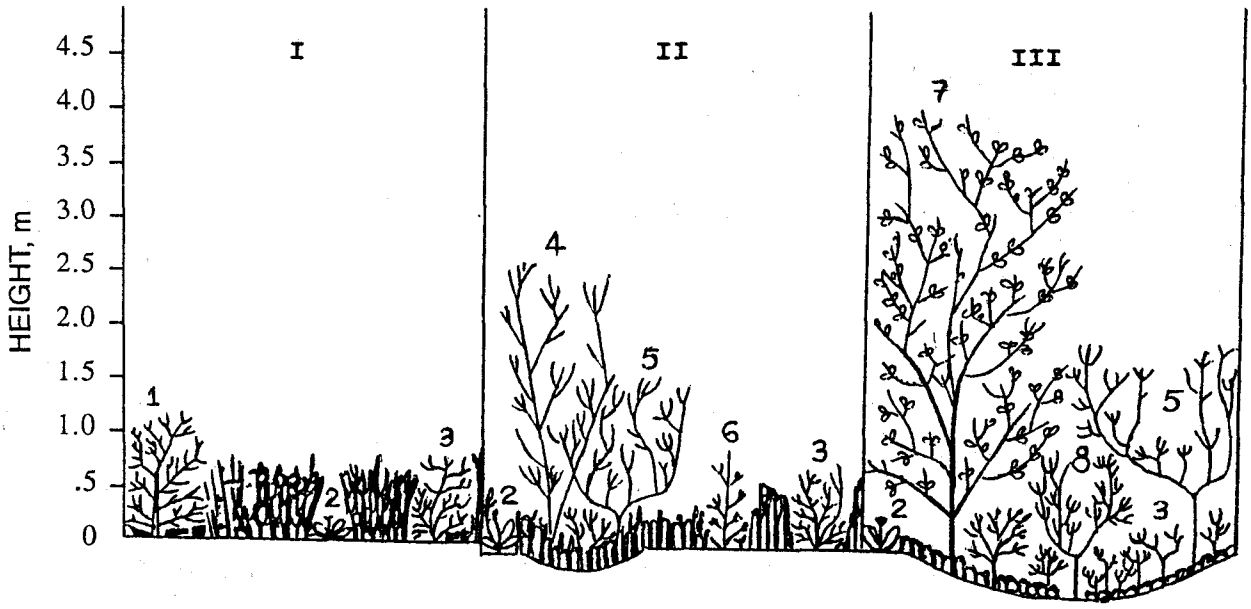
Vegetation Characteristics

Table 2 summarizes data on vegetation. On site I, the undisturbed terrain, low woody species (maximum average height of 0.59 m) were found dispersed among dense grass cover. On the other hand, woody species found on site III, the most heavily mined (large excavations), revealed greater density and a maximum average height of 1.25 m. On site III, individuals were scattered among sparse, low grasses. Vegetation covering small excavations (site II) showed intermediate parameters between undisturbed and heavily mined sites. It is possible that mining's removal of grasses - strong competitors for growing space - has aided woody species.

Table 2. General data for vegetation (woody species) found on sites I (unmined), II (lightly mined), and III (heavily mined).

Site	Total surface area sampled, m^2	Maximum average height, m	Total number of individuals	Total number of species	Total coverage, m^2	Coverage by m^2	Total density, individuals/ m^2
I	1,040	0.59	31.9	30	45.37	4.36	0.31
II	480	1.00	46.1	23	77.56	16.00	.96
III	1,040	1.25	124.0	40	479.74	46.13	1.19

Figure 2 illustrates the general profile of vegetation at the Campo do Saco study area, for each type of site. *Miconia ligustroides* Naud. (Melastomataceae) reached greatest height (1.25 m) on the undisturbed site, *Eupatorium vautherianum* DC. (Compositae) showed greatest height (3.0 m) on the shallow excavation site, and *Tibouchina* aff. *moricandiana* (Melastomataceae) was the tallest (4.2 m) on the large-excavations site.



1 - *Miconia ligustroides* Naud.
2 - *Palicourea rigida* H. B. & K.
3 - *Baccharia brevifolia* DC.
4 - *Eupatorium vauthierianum* DC.
5 - *Leandra lacunosa* Cogn.
6 - *Eupatorium squalidum* DC.
7 - *Tibouchina* aff. *moricandiana* Baill.
8 - *Leandra sylvatica* Cogn.

Figure 2. Profile representing vegetation found on sites I (unmined), II (lightly mined), and III (heavily mined).

Besides height, other important differences were found among woody species, principally between mined and unmined sites. Figure 3 shows the distribution, by order of index of value importance (IVI), for the botanical families found at each type of site. Differences in population parameters for the principal species common to all three types of sites were also observed (table 3).

Species with largest index of value importance were

- On unmined terrain, *Baccharis brevifolia* DC. (74.76), *Sapium marginatum* Muell. Arg. (26.57) and *Aegiphila tomentosa* Cham. (21.81).
- On lightly mined sites, *Miconia ligustroides* Naud. (67.85), *Leandra lacunosa* Cogn. (49.73), and *Eupatorium squalidum* DC. (35.00).
- On heavily mined sites, *Tibouchina stenocarpa* Cogn. (78.06), *Leandra lacunosa* Cogn. (68.54) and *Miconia ligustroides* Naud. (46.45).

The high IVI value for *Tibouchina stenocarpa* Cogn. on mined sites and its absence from the undisturbed site (figure 2) probably indicate that the occurrence of this species depends on drastic site alteration. However, it also occurs in other types of adjacent plant communities outside the study area, such as alongside stream courses.

Relatively few species are found in both mined and undisturbed sites. Those that do occur in all three sites always showed greater phytosociological values for the mined terrain. Only two woody species showed much higher values for undisturbed terrain: *Baccharis brevifolia* DC. and *Aegiphila tomentosa* Cham. It may be concluded that these two probably naturally dominate the latter stages of the original "cerrado" grassland vegetation.

Vegetation in Relation to Soil

The undisturbed site is characterized by very clayey soils of moderate acidity, moderate levels of exchangeable Al^{3+}, and P deficiency. In these soils, a plant formation has developed consisting of savanna almost free of woody species. Dense grasses reaching a height of 0.8 m dominate this community. A few woody species, scattered and with little coverage, reaching 1.25 m in height, were also found in these soils.

Excavated sites feature soils with sandy clay loam texture, moderate acidity, low levels of exchangeable Al^{3+}, and P deficiency. These levels of P, however, are greater than those found on undisturbed terrain. Grasses on the heavily mined site show levels of density, coverage, and height ranging from low to medium. On the other hand, woody species found on these large excavations showed high density, medium coverage, and a height of 4.20 m. Many more species were found on heavily mined terrain than on the undisturbed site.

Dias (1982) reported that chemical analyses of soils from the Poços de Caldas region generally show high acidity and high levels of exchangeable Al^{3+}. For the case of the Campo do Saco study area, the excavations and the 50-yr rest following abandonment have probably altered soil conditions at the mined sites, causing changes in the normal development of vegetation. In addition, excavation lowered the soil's surface closer to the water table, and the remaining pits accumulated rainwater and sediments, factors that modify vegetation's behavior.

If mining had not occurred, it would be difficult to say if original vegetation would have evolved into what we see today covering the study area because some soil characteristics, such as pH, show similar values for both mined and undisturbed sites. The presence of aluminum does not necessarily affect plant growth. However, depending on nutrient richness and H^+ ion activity (pH), it may lead to greater Al^{3+} ion activity, provoking plant toxicity. Because mining removes aluminum present in bauxite, excavating ore might improve soil conditions.

Conclusions and Recommendations

As of 1991, fifty yr after the cessation of bauxite mining, vegetation covering Campo do Saco presented three situations:

1. Nonexcavated sites were covered by normal "campo-cerrado" vegetation. If there had not been any mining, this savannalike vegetation would probably be covering the entire study area today.

2. Heavily excavated sites featured woody species, which were visually more copious than for the rest of the study area. The natural regeneration of these woody species is assumed to have largely disseminated from other nearby plant communities, including riparian forests, brushy secondary growth, and other adjacent "cerrado" formations.

3. Lightly excavated sites developed vegetation with intermediate characteristics between undisturbed and heavily mined sites.

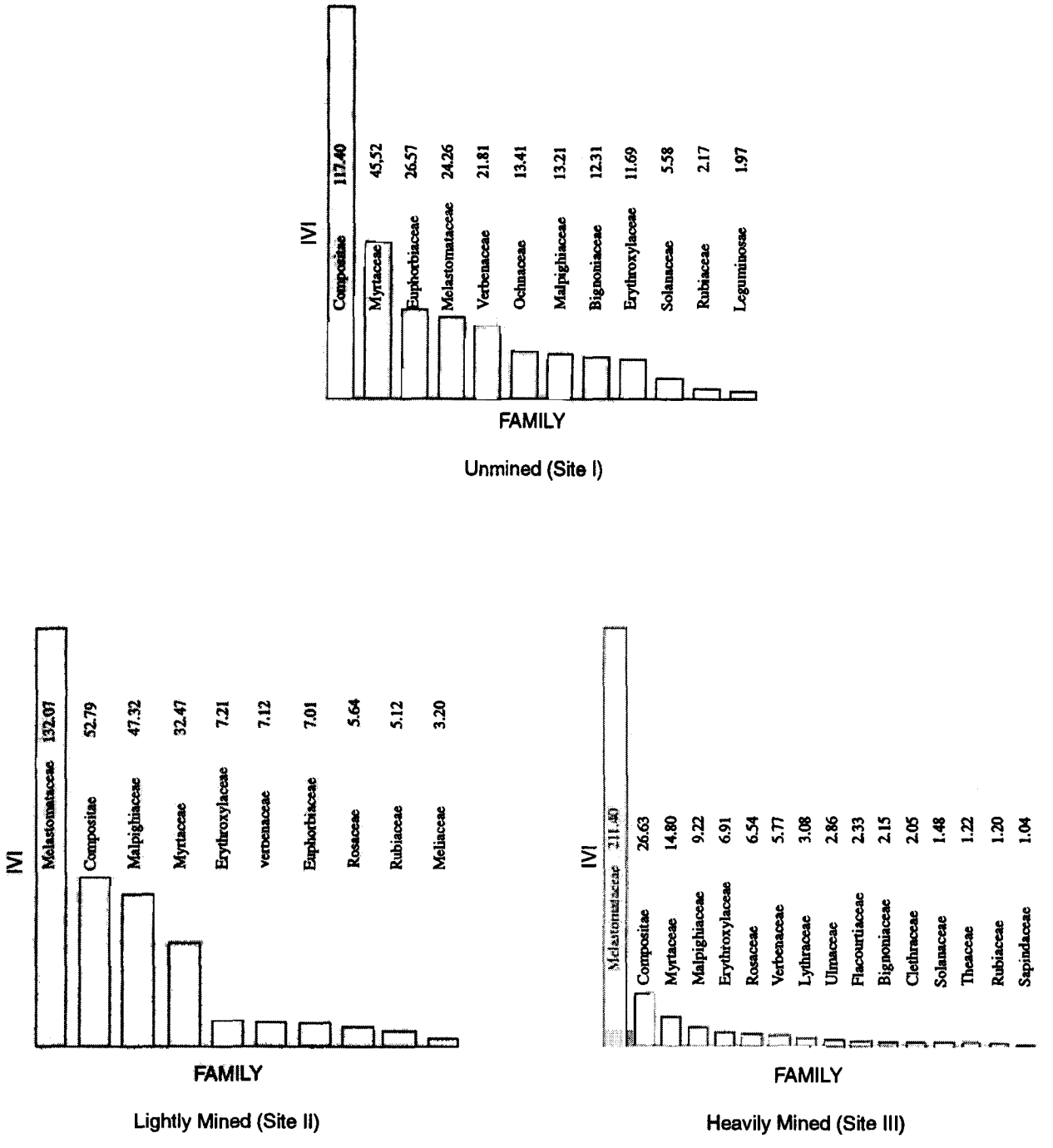


Figure 3 Distribution by order of index of value importance (IVI) of botanical families found at sites I (unmined), II (lightly mined), and III (heavily mined).

Table 3. Population characteristics of principal species found on sites I (unmined), II (lightly mined), and III (heavily mined).

Family and species	H_{max}, m			CR_i, %			FR_i, %			DR_i, %			IVI		
	I	II	III	I	II	III	I	II	III	I	II	III	I	II	III
Bignoniaceae															
Jacaranda caroba DC.	.40	ND	.40	2.11	ND	.01	3.66	ND	.94	1.57	ND	.08	7.24	ND	1.03
Compositae															
Baccharis brevifolia DC.	.76	.80	.84	23.73	.14	2.06	13.41	1.75	4.72	37.62	.43	4.72	74.76	2.32	11.50
Kanimia oblongifolia Baker	.64	.60	ND	.80	.11	ND	3.66	1.75	ND	2.82	2.17	ND	7.28	4.03	ND
Eupatorium squalidum DC.	ND	.90	1.35	ND	4.98	.09	ND	7.02	2.83	ND	23.00	1.30	ND	35.00	4.21
Baccharis dracunculifolia DC. ...	.50	ND	1.40	.65	ND	.42	1.22	ND	.94	.94	ND	2.10	2.81	ND	3.46
Eupatorium vauthierianum DC.	ND	3.00	ND	ND	.96	ND	ND	1.75	ND	ND	.65	ND	ND	3.36	ND
Erythroxylaceae															
Erythroxylon suberosum A. St. Hil.	.60	2.10	2.75	1.19	1.33	2.08	2.44	1.75	3.77	.63	1.09	1.05	4.26	4.17	6.91
Euphorbiaceae															
Sapium marginatum Muell. Arg. ...	1.10	1.70	ND	13.64	.88	ND	8.54	5.26	ND	4.39	.87	ND	26.57	7.01	ND
Malpighiaceae															
Byrsonima intermedia A.	.64	.82	.80	7.60	4.07	1.24	3.66	8.77	6.60	2.19	2.82	1.37	13.45	15.66	9.22
Melastomataceae															
Leandra lacunosa Cogn.	.68	1.40	2.00	.84	11.66	29.00	2.44	8.77	11.32	1.88	29.30	28.22	5.16	49.73	68.54
Miconia ligustroides Naud.	1.25	1.70	1.50	4.57	46.71	18.38	1.22	8.77	12.26	.31	12.37	15.81	6.10	67.85	46.45
Tibouchina frigidula Cogn.	.57	.90	.90	3.42	3.56	.38	4.88	7.02	.94	4.70	3.91	3.23	13.00	14.49	4.54
Leandra sylvatica Cogn.	ND	ND	.70	ND	ND	.31	ND	ND	.94	ND	ND	1.21	ND	ND	2.46
Tibouchini aff. **moricandiana** Baill.	ND	ND	4.20	ND	ND	.62	ND	ND	3.77	ND	ND	.89	ND	ND	5.28
Tibouchina stenocarpa Cogn.	ND	ND	3.00	ND	ND	38.94	ND	ND	12.26	ND	ND	26.85	ND	ND	78.06
Myrtaceae															
Campomanesia Ruiz & Pav. sp.	.30	.59	1.65	2.01	3.34	.42	3.66	8.77	1.90	1.57	1.74	.24	7.24	13.85	2.54
Eugenia Mich. ex Linn. sp.	ND	.40	.75	ND	.18	.49	ND	1.75	2.83	ND	.22	1.13	ND	2.15	4.45
Psidium cinereum Mart. ex DC. ...	.92	ND	.54	1.76	ND	.02	4.88	ND	.94	1.25	ND	.08	7.89	ND	1.04
Psidium Linn. sp.	.30	ND	1.20	.24	ND	.12	1.22	ND	1.90	.31	ND	.89	1.78	ND	2.90
Campomanesia obversa Berg	.41	.25	ND	.39	.28	ND	1.22	8.77	ND	1.25	1.08	ND	2.86	1 .13	ND
Pseudocaryophyllus acuminatus (Link) Burret	.25	.20	ND	1.00	.20	ND	9.76	5.26	ND	5.64	.87	ND	16.40	6.33	ND
Rosaceae															
Bubus brasiliensis Mart.	ND	1.10	1.25	ND	.62	.23	ND	3.51	2.83	ND	1.52	1.37	ND	5.64	4.43
Rubiaceae															
Palicourea rigida H. B. & K.	.16	.33	.30	.32	.37	.01	1.22	3.51	.94	.63	1.52	.24	2.17	5.19	1.20
Solanaceae															
Solanum lycocarpum A. St. Hil. ..	.62	ND	.60	.35	ND	.30	3.66	ND	.94	1.57	ND	.24	5.58	ND	1.48
Verbenscese															
Aegiphila tomentosa Cham.	.76	.50	.80	8.71	1.03	.70	2.44	1.75	.94	1 .66	4.34	1.21	21.81	7.12	2.86

ND = not detected.

Bauxite mining also affected natural plant regeneration at Campo do Saco because it modified these soil characteristics:

1. Substrate texture - soils that were originally very clayey before mining gave way to sandy clay loam.

2. Available humidity - excavations lowered the soil surface closer to the water table; these pits also accumulate water and sediment when it rains.

In general, the two exploited but later abandoned sites at Campo do Saco revegetated into more copious landscapes than did the undisturbed site. Mining, however, usually does not improve landscape conditions. The effect observed at Campo do Saco is probably exceptional, reflecting increased water availability and bauxite removal.

As for reopening the mine, this study supports these recommendations:

1. Set site rehabilitation, and not restoration, as the revegetation program's goal. Edaphic conditions altered by mining seem to prohibit a return to original plant communities at Campo do Saco. It would be better to improve these conditions so that mined-out lands can be returned to productive and self-sustaining use.

2. Try cultivating *Tibouchina stenocarpa* Cogn., *Leandra lacunosa* Cogn., and other species of Melastomataceae. This study suggests that planting these pioneer species could accelerate the natural process of regeneration at Campo do Saco.

3. To disseminate best adapted species, use several different seed sources such as

- Plants that regenerate naturally in the area.
- Stands of riparian forests and other plant communities adjacent to the mine, protected during mining as reserves for this use.
- Topsoil removed during the initial phases of the mine reopening, especially from previously unmined parcels.
- Wildlife (especially birds and bats), which carry seeds to the degraded sites.

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RELOCATION AND RESTORATION IN LIMESTONE QUARRIES: IMPLICATIONS FOR INVERTEBRATE COMMUNITIES FOLLOWING TWO EXTREME FORMS OF MANAGEMENT[1]

W. Rod Cullen and C. Philip Wheater[2]

Abstract: Limestone quarrying often results in the need to protect existing ecosystems and restore abandoned quarries, and hence the development of appropriate management techniques. This study discusses the effects of two such, relocation and restoration blasting, on invertebrate communities. Invertebrates are important components of ecosystems that are frequently ignored during, or following, management. Following the relocation of part of a magnesian limestone grassland in County Durham, UK, the invertebrate fauna was examined in turfs relocated at different dates. Elements of these communities declined from the most recently relocated turfs, rising in the oldest plots. This may be due to an initial shock caused by disturbance, followed by recovery as the turfs establish and settle. Invertebrate recovery may be slower than for plants, due to direct or indirect (via changes in plant communities) influences. Restoration blasting in postworked quarries aims to produce landforms that can support ecosystems similar to those on natural dalesides. Comparisons between natural, abandoned and managed sites in the White Peak area of Derbyshire, UK, indicate that natural dalesides have more invertebrate orders than disused quarries, which have higher values than restoration blasted sites. The time communities have been developing may be important. The restoration blast piles have been effected by severe rabbit grazing, and the effects on the vegetation has repercussions on the associated invertebrate communities. These extreme forms of management may influence invertebrates to a greater extent than plants. It is, therefore, important to encourage an integrated approach to management, to take account of all the communities involved.

Additional Key Words: reclamation, restoration blasting, colonization, establishment.

Introduction

Limestone is an important raw material, used in the manufacture of a wide range of products. In Britain figures for 1988 estimated that 13,920 ha of land was affected by limestone quarries (Brown 1992). By 1987 annual production had risen to 124 million mt (Gunn and Bailey 1991) with a projected rise well into the 1990's. Often the occurrence of limestone coincides with some of Britain's most valuable ecological and scenic areas. Resulting conflicts have led to the development of several techniques for the protection and restoration of threatened or damaged areas, while enabling extraction to continue. In Britain planning permission for a new quarry or extension to an existing quarry requires the company to submit a reclamation strategy (Gunn and Bailey 1991). There is, therefore, a need for effective management both before and after quarrying, and several techniques are currently being used. We shall discuss some of the implications of two of the more extreme methods, one before extraction (relocation) and one following abandonment (restoration blasting).

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[2]W. Rod Cullen, Researcher, and C. Philip Wheater, Principal Lecturer in Ecology, Department of Environmental and Geographical Sciences, The Manchester Metropolitan University, Manchester, M1 5GD, UK.

Relocation is an extreme form of management, potentially retaining a whole ecological unit. In England Byrne (1990) recorded 92 registered, existing or proposed, relocations, 48 of which were grasslands. To retain a magnesian limestone grassland threatened by quarrying, part of Thrislington Plantation, County Durham, UK, was relocated over 8 yrs. Such grasslands in northeast England are important ecological links between northern montane and southern lowland grasslands and are now limited in size and distribution (Park 1988). Thrislington Plantation SSSI (Site of Special Scientific Interest) is the most diverse, representative, and extensive area (22 ha) of the remnant magnesian limestone grasslands in the region (Richardson 1984). The relocation is fully described by Park (1988, 1989). Briefly, following removal of the topsoil from the receiver site, turfs 4.75 m by 1.75 m by 0.4 to 0.5 m deep were moved in the winter to prevent drying and splitting. Interturf gaps were infilled with overburden and soil from the donor site. For 2 yrs these gaps were leveled, and emergent weeds were controlled. After 2 yrs, the plots were flail mown annually to check scrub invasion. Relocations are not always successful (Byrne 1990), making it important to examine the apparent success at Thrislington. This study examines some of the effects of the relocation on the invertebrate fauna of one plant community (Festuca-Helianthemum) in plots relocated during the first 3 and last 2 yrs of the relocation program (1982-90). This enables comparisons to be made between the temporal extremes of the relocations.

Limestone quarries are often difficult to restore, being of coarse substrate, deficient in minerals, and often excessively drained (Bradshaw and Chadwick 1980). Such factors may provide habitats for early successional, species-rich plant communities similar to seminatural calcareous grasslands (Davis et al. 1985). Natural colonization and succession of quarries is often slow (Davis et al. 1985) and is no longer considered an acceptable reclamation technique (Bailey et al. 1991). Modern extraction methods produce harsher, more engineered landforms which evolve slower following abandonment than faces worked by the previous methods of manual and black powder blasting (Gagen and Gunn 1987). Most options for quarry after-use require treatment of the faces to reduce rock fall and improve the visual appearance (Bailey et al. 1991). Recently, the Limestone Research Group has developed a method of limestone quarry restoration, aiming to construct "daleside landform sequences" capable of supporting self-sustaining ecosystems (Bailey et al. 1991). Restoration blasting utilizes high explosives to produce landforms that replicate a natural daleside model (Gagen and Gunn 1987), Greatrocks Dale, which is classified as Festuca ovina-Avenula pratensis grassland, Dicranum scoparium subcommunity using the National Vegetation Classification (Ash 1981). Cover material, mainly limestone waste, is applied to the scree slopes prior to hydroseeding and tree planting to facilitate the development of vegetation (Bailey et al. 1991). The current study compares the invertebrate communities developing on recently created restoration blast piles with those on a natural daleside model and several naturally colonized abandoned quarries in the same locality.

Methods

Five sites at the temporal extremes of the Thrislington relocations were surveyed from June 29, 1990 to August 3, 1990 for vegetation using a 20-pin pinframe and for invertebrates using a 3 m by 3 m grid of 9 pitfall traps. The sites were relocated during the winters of 1982-83 (site A), 1983-84 (site B), 1984-85 (site C), 1988-89 (site G), and 1989-90 (site H). Diversities were calculated using the Shannon Index (H') using log base e (Magurran 1988).

During May 1992, five pitfall traps were placed on identifiable features in each of three restoration blasts, the natural daleside model, and seven disused quarries. Random vegetation surveys were carried out in April using two

0.25 m by 0.25 m quadrats on each feature. All sites were located within the White Peak area of Derbyshire and in close proximity to each other. The three restoration blasts were created in November 1988 (RB5), February 1989 (RB7), and April 1989 (RB8/9). Seven disused quarries, differing in age since abandonment (37 to 100 yrs), face length, face height, area, and method of extraction, were examined to compare restoration blasting with natural colonization. Comparisons among sites were made using a nonspecific, nonparametric analysis of variance (Meddis 1984).

Results-Relocation

Surveys recorded 11 species of grasses and sedges and 29 species of herbaceous and woody plants. Plant species number and diversity decreased slightly from the most recently relocated plots to the intermediate ones, rising again for the more established plots (fig. 1). The number of individual hits increased initially (between plots H and C), falling to plot A. Hits on bare ground were found only in plots G and H.

Table 1. Invertebrate orders found in pitfall traps per relocated plot.

Invertebrate orders	Plot				
	A	B	C	D	E
Opiliones	152	88	76	203	346
Acarina	185	114	216	66	136
Araneae	279	175	192	177	197
Isopoda	91	40	55	41	55
Julidae	15	22	12	2	4
Polydesmidae	0	0	0	1	2
Lithobiomorpha	5	4	14	15	3
Geophilomorpha	0	0	1	0	0
Pulmonata	19	54	9	9	1
Collembola	186	178	99	79	256
Orthoptera	16	11	19	6	10
Dermaptera	2	3	0	0	3
Hemiptera	79	74	21	66	80
Lepidoptera	10	13	7	4	7
Hymenoptera	282	138	100	106	215
Diptera	18	15	26	48	119
Coleoptera	158	69	175	200	634

[1]The figures indicate the numbers of animals found per site from a total of 9 pitfall traps per site and 5 weekly sample periods. For transplant dates of sites, see figures 2 or 3 or text.

All the plots contained similar invertebrate orders, with plot H having

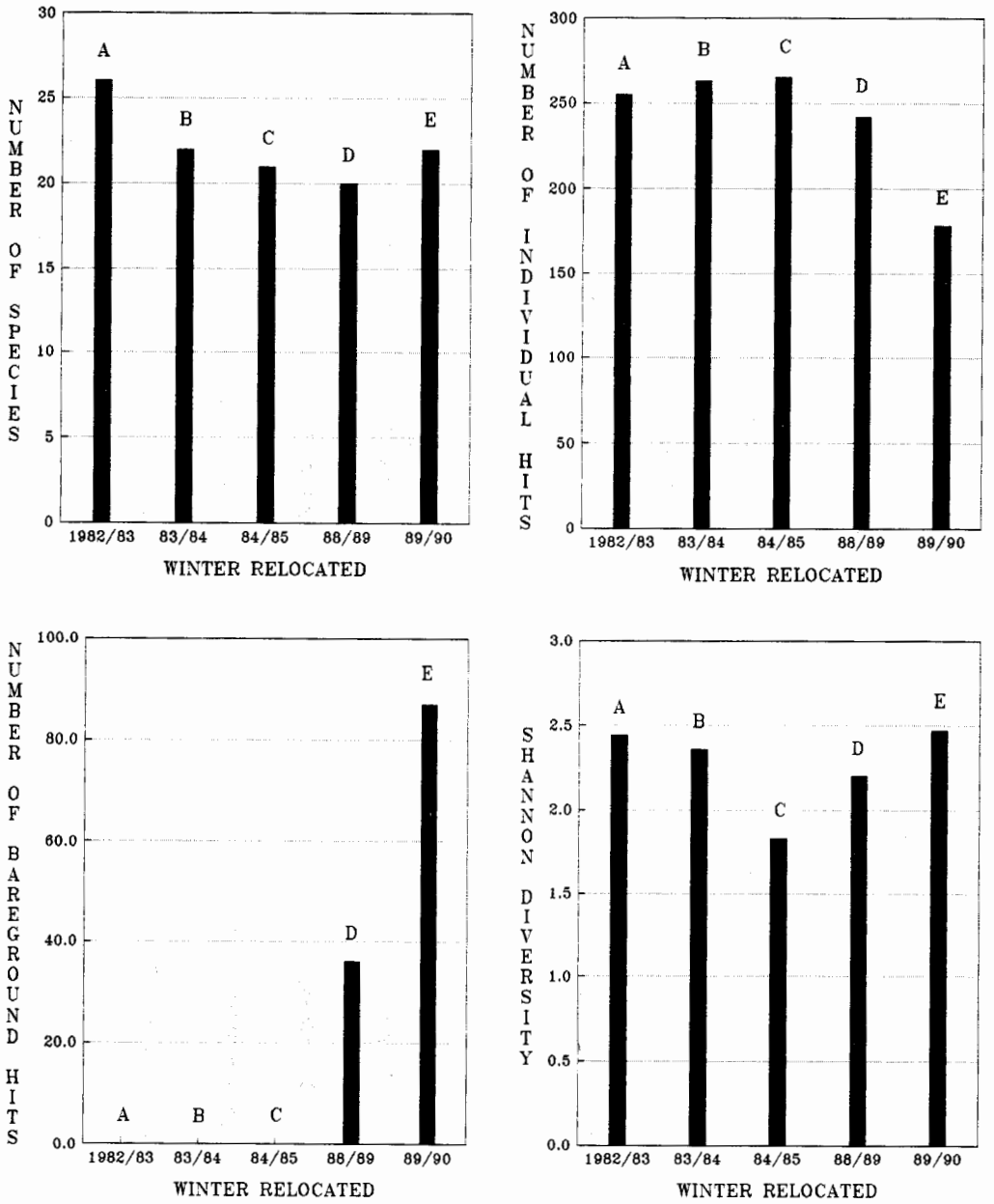

Figure 1. Floral changes with time since relocation.
Letters indicate sites, see text for details.

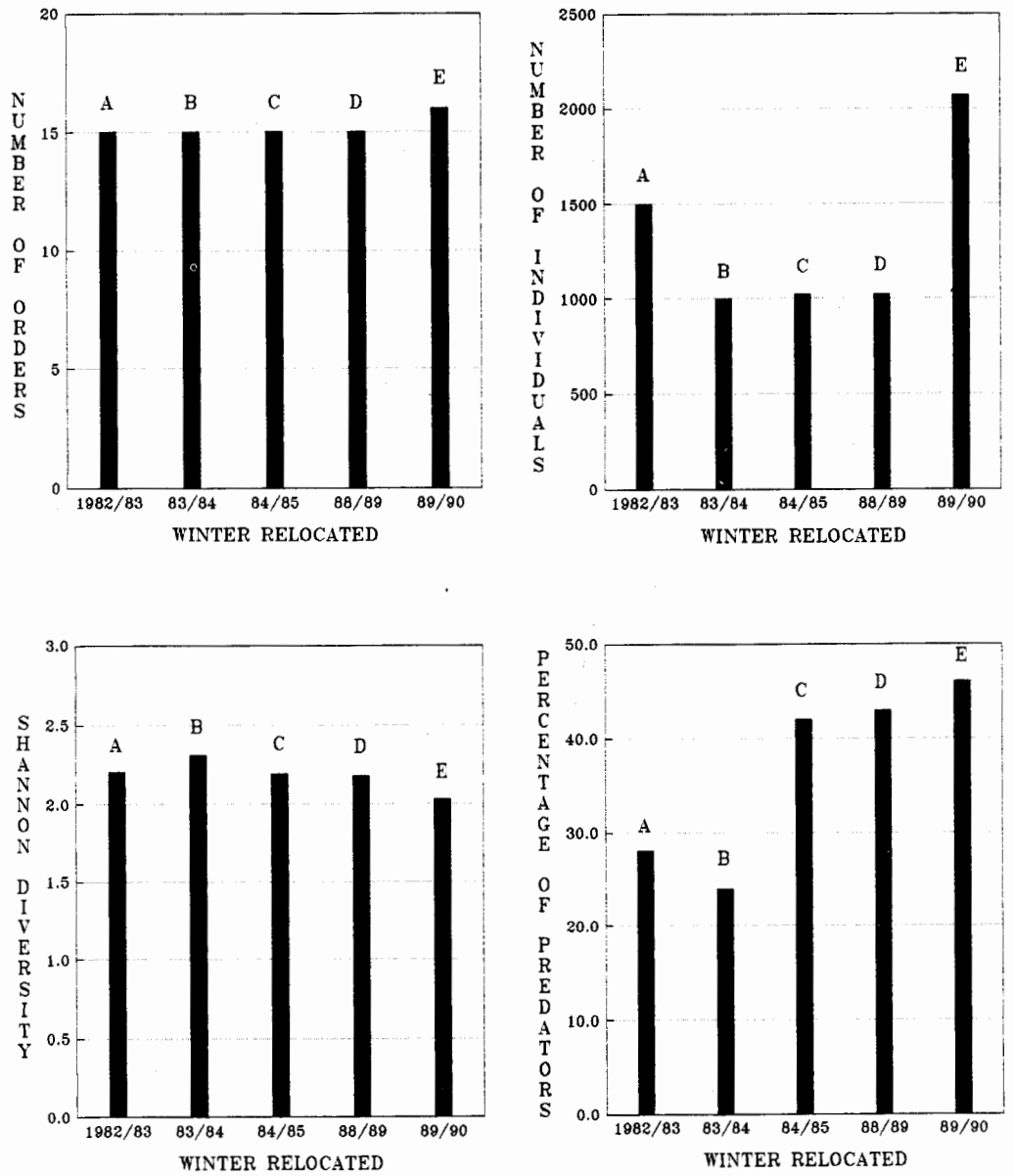

Figure 2. Invertebrate changes with time since relocation. Letters indicate sites, see text for details.

16 compared with the other plots' 15. The numbers of animals were greatest at plot H, followed by a decline to plot B and a rise to plot A (fig. 2). Of the eight insect orders recorded, all were found at the extremes (plots A, B, and H), and seven were found at the intermediate plots (table 1). Again an initial decline in the number of individuals from plot H occurred (to a minimum at plot C), followed by a rise to plot A. The diversities of invertebrate orders (fig. 2) were positively correlated with the time since relocation (Rs = 0.8999, df = 3, p = 0.037).

Discussion-Relocation

Most of the 40 vascular plant species found were typical of calcareous soils (Grime et al. 1990) and more species were found at the oldest relocated plot (fig. 1). Variation between plots may be due to different tolerances to disturbance or the exploitation of new areas (turf edges and gaps). Variation in the individual hits between plots (fig. 1) indicates, in plots A to C, repeat hits on the same plant by adjacent pins (which were not recorded), and in plots G and H repeat hits and hits on bare ground. Bare ground between turfs has been colonized in plots A, B, and C. The lower diversity at plot C may be due to dominant species such as Carex flacca (Glaucous Sedge), Dactylis glomerata (Cocksfoot) and Helianthemum nummularium (Common Rockrose).

The number of invertebrate orders varies little between plots (fig. 2). The number of individuals initially declines with age of turf, from plot H, rising to plot A. Sheppard (1990), in an earlier survey of the site, recorded a decline in species number during the first few years following relocation. The increase in diversity with time since relocation may reflect community stabilization following constancy of management and colonization of bare ground. The proportions of the major predator groups (Opiliones, Araneae, Chilopoda, Carabidae, and Staphylinidae) tend to decline with time since relocation (fig. 2). Pitfall catches increase with abundance and activity; the mobility of some species (Uetz and Unzicker 1976) and the vegetation around the traps (Greenslade 1964) influencing catches. The predators in plots G and H are highly active species; here gaps between the turfs may increase their activity and, thus, capture. Managing grasslands by cutting encourages new plant growth and maintains the sward, benefiting phytophages (Morris and Rispin 1988), which being less mobile than many predators are less likely to be caught in pitfall traps. However, Orthoptera, Lepidoptera larvae, and Pulmonata are more numerous in plots A-C (table 1). The fall and subsequent rise of some invertebrates may be due to initial changes caused by disturbance, followed by recovery with time. Such changes may result from the direct influence of the relocation, or as a consequence of changes in vegetation type and cover (e.g. the changes in the numbers of Hemiptera, which may reflect the availability of food plants).

Both the vegetation and invertebrates show changes in community structure with time since relocation. This may indicate an initial "shock" following relocation. For invertebrates the initial effects may be more traumatic than for plants. Such effects and the availability of new microhabitats (such as bare ground) may alter the community dominance (e.g. by enhancing predator activity). Changes in the vegetation may influence invertebrate herbivores and hence predators, leading to a longer time lag in recovery for invertebrates than for plants. Many such relocations show initial changes in vegetation structure and community dynamics, especially in infilled gaps between turfs (Byrne 1990). At Thrislington Plantation, where relocated turfs are large and limited disturbance is caused, recovery is possible. Some elements of the floral and invertebrate communities appear to have been initially effected by the relocation, although with possible subsequent recovery. At Thrislington Plantation these may be neither catastrophic nor persistent; with time the interturf gaps should be colonized and the area may develop similar communities

to those prior to relocation.

Results - Restoration Blasting

Significant differences were found between the areas examined (restoration blast piles, natural daleside, and disused quarries) for vascular plant species number, cover and diversity, total plant cover, and the percentage cover of bare ground (table 2). Significant differences were found between the restoration blast piles for vascular plant and total plant cover (table 2). In both cases RB8/9 was significantly higher than either RB5 or RB7.

Table 2. Analysis of vegetation.

	DIFFERENCES BETWEEN AREA TYPE		DIFFERENCES BETWEEN RESTORATION BLAST PILES	
	Non specific p	Order	Non specific p	Order
Numbers of plant species	0.000	ND > RB > DQ	0.140	No significant difference
Vascular plant cover	0.000	ND > DQ > RB	0.005	RB8/9 > RB5 / RB7
Total plant cover	0.000	ND > DQ > RB	0.006	RB8/9 > RB5 / RB7
Plant diversity	0.038	ND > RB > DQ	0.219	No significant difference
Bare ground cover	0.000	RB > ND / DQ	0.120	No significant difference

The invertebrate data showed significant differences between the types of sites for all aspects except Araneae per trap (table 3). Except for Mollusca per trap, greater numbers were found on the natural daleside, fewer in the disused quarries, and fewer still on the restoration blast piles. This pattern was repeated for the numbers of individuals and orders per trap. The number of Mollusca per trap was significantly higher for both the natural daleside and disused quarries than for the restoration blast piles. Significant differences were found between the restoration blast piles for Isopoda per trap and total individuals per trap (table 3). RB8/9 had significantly more Isopoda per trap than either RB5 or RB7, but this was reversed for total individuals per trap.

Discussion-Restoration Blasting

Since the natural daleside is the oldest site, it may be expected to have the most vascular plant species and highest diversity (table 2). The disused quarries are older than the restoration blast piles but have fewer vascular plant species and lower diversities, possibly due to the blast piles having recently been seeded with a daleside seed mix. The pattern for vascular plant and total plant cover is slightly different. The natural daleside is again the highest, probably due to its age and development. Disused quarries have higher values than the restoration blast piles, possibly reflecting a richer, but more patchy, cover on the latter sites. This is reinforced by the restoration blast piles having the highest percentage cover of bare ground. Davis et al. (1985) recorded that rabbit grazing affects vegetation establishment on bare quarry

floors and that levels of bare ground were greater where grazing was not prevented. The restoration blast piles have been subject to severe grazing by rabbits, which may account for the elevated levels of bare ground (table 2). However, the hydroseeding and the cover material used on the blast piles may also have had an influence. Significant differences were found between the restoration blast piles for vascular plant and total plant cover (table 2). RB8/9 have the highest values, probably reflecting the less severe grazing by rabbits here compared with RB5 and RB7.

Table 3. Analysis of invertebrates.

	DIFFERENCES BETWEEN AREA TYPE		DIFFERENCES BETWEEN RESTORATION BLAST PILES	
	Non specific p	Order	Non specific p	Order
Diplopoda/trap	0.002	ND > DQ > RB	0.051	No significant difference
Mollusca/trap	0.000	ND / DQ > RB	0.297	No significant difference
Isopoda/trap	0.000	ND > DQ > RB	0.010	RB8/9 > RB5 / RB7
Araneae/trap	0.161	No significant difference	0.628	No significant difference
Individuals/ trap	0.009	ND > DQ > RB	0.013	RB5 / RB7 > RB8/9
Total Orders	0.000	ND > DQ > RB	0.294	No significant difference

More orders per trap and more animals per trap (except Araneae) were found on natural dalesides than in disused quarries, with fewer still on restoration blast piles. This may be related to age or degree of establishment, the natural daleside being the oldest, most developed site. The communities on the disused quarries have been developing for 37 to 100 yrs, and the restoration blast piles are relatively new (4 to 5 yrs old). In contrast to the establishment of vegetation, which has been actively encouraged on the restoration blast piles, the invertebrates have colonized and established on their own. Time may, therefore, be important in the development of invertebrate communities on restoration blast piles. Several factors may determine the time scale for successful colonization, including proximity of suitable source areas, size of the available species pool, the mobility of animals, and the availability of suitable conditions and resources for the potential colonists. Araneae tend to be very mobile and active, and are often amongst the first predators attempting to colonize new sites (Gorman 1979). This mobility and the close proximity of the natural daleside and disused quarries to the restoration blast sites may account for the lack of significant differences in these animals between the areas. In comparisons of Isopoda between blast piles, RB8/9 had greater numbers per trap than RB5 and RB7, which have a reduced height of standing vegetation as a result of severe rabbit grazing. This has occurred to a lesser extent on RB8/9; the increased cover (table 2) and thus detritus may account for the higher levels of Isopoda here. RB8/9 was also significantly lower in total numbers of individuals per trap than either RB5 and RB7, possibly due to being the youngest site and hence having had less time to recruit animals.

General Conclusions

Grassland management for conservation is often geared to the requirements of plant species or vegetation types rather than those of invertebrates (Rushton 1988). This is certainly the case at Thrislington Plantation. Byrne (1990) stated that little attention was paid to the invertebrate communities at relocated sites. Thrislington Plantation is unusual in the level of monitoring that has taken place during and following relocation. Since the effects of relocations and subsequent management may be greater on the invertebrate communities than on the vegetation, we suggest that future programs monitor both components and consider implementing management strategies to enhance invertebrate conservation. It is also important to recognize that in all cases of extreme management, time considerations are important. This may be especially true for organisms, including invertebrates, that are usually neither introduced nor encouraged by the initial management process. Literature regarding the colonization of disused quarries and restored ecosystems by invertebrates is scant. This is in no small part due to the prevailing attitude that animals play little part in the reconstruction of a properly functioning ecosystem, considering that the first trophic level is the crucial element in any ecosystem. It is hoped that this, and similar, work will encourage a more integrated approach to ecosystem protection and restoration, catering for the necessary interactions between plants and animals during ecosystem development.

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ECOLOGICAL DYNAMICS OF A CREATED SEASONAL WETLAND IN NORTH DAKOTA[1]

Donald R. Kirby, Kelly D. Krabbenhoft, Mario E. Biondini and Greg P. Summers[2]

Abstract: The objective of the research was to compare created and natural seasonal wetlands in terms of (1) composition of vegetative communities, (2) net primary productivity, and (3) aboveground and belowground decomposition rates of plant tissues. The created seasonal wetland is located on the Falkirk Mine (North American Coal Corp.), which is on the western edge of the Prairie Pothole Region approximately 90 km north of Bismarck, ND. The natural seasonal wetland is located in the east-central portion of the Prairie Pothole Region approximately 150 km east of the Falkirk Mine. Permanent transects perpendicular to the wet meadow and shallow marsh vegetative zones were established between 1991 and 1992 for the created wetland and between 1990 and 1991 for the natural wetland. Aboveground live biomass was estimated by clipping random 0.1-m^2 or 0.25-m^2 quadrats in each vegetative zone. Decomposition rates were determined by incubating known amounts of fresh, dried plant tissue in mesh bags above and below ground over two growing seasons. Comparison of yield between the wetlands will be restricted to 1991. A total of 43 plant species were identified at the created wetland. Eighteen species are considered "weedy" pioneer types, while eight species are dominant wet meadow and shallow marsh plants identified with native seasonal wetlands. Aboveground live biomass in 1991 was 4,920 and 5,480 kg/ha for the wet meadow and shallow marsh zones of the created wetland. In comparison, the natural wetland in 1991 had aboveground live biomass of 11,886 and 8,521 kg/ha for the wet meadow and shallow marsh zones. Decomposition rates were slow in both created and natural seasonal wetlands. After 2 yr, 50% decomposition was never reached for aboveground and belowground zones in either wetland. Plant detritus remains in seasonal northern prairie wetland systems for extended time periods.

Additional Key Words: surface coal mining, prairie pothole region, net primary productivity, decomposition rate.

Introduction

The need for wetland protection and restoration is long overdue. Nearly 50% of the wetlands in the conterminous United States have been drained or altered (Office of Technology Assessment 1984). In North Dakota, approximately 60% of the original wetlands have been drained or altered with less than 1 million ha remaining (Tiner 1984, Mitsch and Gosselink 1986). Northern Great Plains wetlands are considered an important wetland region of the world, especially because of the numbers of waterfowl produced in the region (Ogaard et al. 1981, Weller 1981).

The concept of wetland creation has met various degrees of acceptance among environmental, regulatory, and development groups (Larson 1987). Wetland creation is accomplished by manipulating the hydrology, soil, and vegetation of a site. Previously, many restored wetlands considered successes were assessed solely on wetland plant establishment (Reimold and Cobler 1986, Blomberg 1987), while reports of the performance of other wetland functions in the literature are scant (Cooper 1987). Since Federal and State reclamation statutes regulate created wetlands as ecosystems, a better understanding of the basic functions in created wetlands is necessary to accurately assess the "success" of these restored wetlands.

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[2]Donald R. Kirby, Kelly D. Krabbenhoft, and Mario E. Biondini are professor, research specialist, and associate professor, respectively, Animal and Range Sciences Department, North Dakota State University, Fargo, ND 58105. Greg P. Summers is an environmental consultant, 317 SW. Alder, Suite 800, Portland, OR 97204.

The objective of this research was to compare created with natural seasonal wetlands in the Prairie Pothole Region of west-central North Dakota. Specific objectives were to (1) identify plant species and communities, (2) determine aboveground net primary productivity of each plant community, and (3) examine aboveground and belowground decomposition rates of plant tissues in each community.

Study Area

The natural seasonal wetland was located on the Cottonwood Lake Research Area approximately 50 km northwest of Jamestown, ND. The created seasonal wetland was located on the Falkirk Mine, Underwood, ND, operated by the North American Coal Corp. Both research sites are in the Missouri Coteau Physiographic Region, a large glacial drift complex (LaBaugh et al. 1987).

The semiarid, continental climate of the region is characterized by cold winters and hot summers. Temperatures range from -12° C in January to 22° C in July. Average annual precipitation is 45 cm, of which 80% falls between April and September. However, dry conditions persisted over the study period, and annual precipitation between 1989 and 1991 averaged 40 cm.

The natural wetland was classified as seasonal using the Stewart and Kantrud (1971) system. Seasonal wetlands have wet meadow and shallow marsh vegetative zones formed in concentric rings from increasing water permanency. Wet meadow zones are dominated by grasses, rushes, sedges, and forbs. Some woody vegetation may also be present. Shallow marsh zones are dominated by coarse emergent grasses, sedges, and some forbs.

The created wetland was reclaimed by engineering a catchment basin, topsoiling with previously excavated wetland soil, and seeding with a grass species mixture. The catchment basin was modeled using the U.S. Department of Agriculture's Soil Conservation Service TR20 Model. The model engineers restored wetlands for both surface acreage of catchment basin and acre foot of water storage. The study site, one of numerous restored wetlands on the mine, received national recognition in 1986 from the Office of Surface Mining (Excellence in Surface Coal Mining Award) and in 1989 from the Take Pride in America National Award Program for wetland reclamation.

Methods

Three transects were established in both wetlands. The transects extended from the center outward, bisecting vegetational zones. Subsequent sampling took place at permanent points along the transects, 1 for each zone and transect in the natural wetland (6 total) and 2 in the shallow marsh and 3 in the wet meadow zones for each transect in the created wetland (15 total).

Aboveground Live Biomass

In the natural wetland, aboveground live biomass (ALB) was estimated by harvesting twice a year. The first sampling coincided with the full growth stage of cattail (*Typha latifolia*), which occurred in mid-June. The second harvest occurred in mid-August at the beginning of cattail senescence. If live biomass increased between harvests, the increase was added to the initial sampling estimates. Four 0.1-m^2 quadrats were randomly located and clipped to species at each point. Clippings were dried to a constant weight at 60° C and weighed.

Created wetland ALB was estimated in early August at full growth of cattail. Five 0.25-m^2 quadrats were randomly harvested at each point on shallow marsh transects. Ten quadrats total were harvested (4, 3, and 3) at the three permanent transect points in the wet meadow zone. Clippings were separated to species, dried to a constant weight at 60° C, and weighed.

Biomass Decomposition

Aboveground and belowground decomposition was determined for the dominant emergent species using the litter bag technique (Brinson et al. 1981). Bags were constructed of 2-mm nylon mesh screen and were approximately 10 by 15 cm. To account for losses or accumulation of material in bags, control bags filled with Styrofoam[3] packaging peanuts were placed alongside every plant litter bag.

Nine aboveground and nine belowground litter bags were placed at each sampling point of the natural wetland. Aboveground bags were filled with a dominant species from each vegetative zone. Slough sedge (*Carex atherodes*) was used for the wet meadow zone and marsh smartweed (*Polygonum coccineum*) for the shallow marsh zone. Plant material from these species was collected in the spring of 1990, dried, weighed, and placed in bags. Belowground litter bags were filled with washed, dried, and weighed roots collected from their respective zone. All litter bags were placed in or on the ground on June 25, 1990. Bags were retrieved after 2 days, and after 2, 4, 8, 16, 54, 60, 108, and 114 weeks. Upon retrieval, bags were gently washed, dried, and weighed.

For the created wetland, seven aboveground and seven belowground litter bags of each species were placed at sampling points along transects. Wooly sedge (*Carex lanuginosa*) and Mexican dock (*Rumex mexicanus*) were selected for the wet meadow zone, while slough sedge, creeping spikerush (*Eleocharis palustris*), and marsh smartweed were selected in the shallow marsh zone. Belowground litter bags were filled with washed, dried, and weighed roots and rhizomes collected from their respective vegetative zone. Dried and weighed materials from these species were placed in or on the ground June 12, 1991. Bags were retrieved at 2 days, and at 4, 8, 16, 52, 68, and 104 weeks. Upon retrieval, bags were gently washed, dried, and weighed.

Rate of decomposition for both wetlands was derived using the following equation:

$$\% \text{ decomposition} = \frac{[OM_p x S_p - (S_r - (C_p - C_r)) x OM_r] x 100}{(OM_p x S_p)},$$

where OM_p = proportion of sample's organic matter when placed,
OM_r = proportion of sample's organic matter when retrieved,
C_p = weight of control when placed,
C_r = weight of control when retrieved,
S_p = weight of sample when placed, and
S_r = weight of sample when retrieved.

Statistical Analysis

Analyses were conducted using the Statistical Analysis System (SAS 1990). Treatments were the two wetlands, natural and created, while transects served as replication. Aboveground net primary production differences were tested using a t-test ($P<0.05$). Comparisons made were between years on each wetland and between wetlands in 1991. Least squares analysis ($P<0.05$) was conducted for aboveground and belowground decomposition in each vegetative zone. Differences were separated using Tukey's test ($P<0.05$). Comparisons between treatments and dates and within treatments across retrieval dates were made. Repeated-measures analysis ($P<0.05$) of the General Linear Models (SAS 1990) was conducted to test means and slopes of decomposition rates between wetlands for each zone.

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[3]Reference to specific products does not imply endorsement by the Bureau of Mines.

Results and Discussion

Species Composition

A total of 43 plant species were identified in the created wetland (table 1). Eighteen species are considered "weedy" pioneer types, while eight species are dominant wet meadow and shallow marsh plants identified with native wetlands. Owing to dry conditions, seven low prairie species were also found occupying the wet meadow zone.

Dominant species in the native wetland included quackgrass (*Agropyron repens*), smooth brome (*Bromus inermis*), northern reedgrass (*Calamagrostis inexpansa*), slough sedge (*Carex atherodes*), Canada thistle (*Cirsium arvense*), wild licorice (*Glycyrrhiza lepidota*), fowl bluegrass (*Poa palustris*), swamp knotweed (*Polygonum coccineum*), and western snowberry (*Symphoricarpos occidentalis*) in the wet meadow zone, and ovalleaf milkweed (*Asclepias ovalifolia*), slough sedge, Canada thistle, and swamp knotweed in the shallow marsh zone. Numerous "weedy" pioneer species and low prairie species also occupied the wet meadow and shallow marsh zones of the native seasonal wetland. A complete list of common dominant native wetland species by zone can be found in Stewart and Kantrud (1971), Fulton et al. (1986), and Kantrud et al. (1989).

Plant species in native, freshwater, and seasonal wetlands are distributed by the presence or absence of water and their tolerance to water permanence. Species and normal zonation patterns have been shown to be altered during extended wet or dry periods, which are common to northern prairie ecosystems (Smeins 1967, Stewart and Kantrud 1971, Fulton et al. 1986, Kantrud et al. 1989). Both wetlands in this study were subjected to an extended dry period from the fall of 1987 to 1992. The presence and relative abundance of low prairie and "weedy" pioneer species in both seasonal wetlands is indicative of these dry conditions. Despite the disturbances due to reclamation and extended drought, species composition of the created wetland appears promising for continued recolonization of the site.

Aboveground Live Biomass

Aboveground live biomass within zones was not different ($P<0.05$) between years for each wetland (table 2). Numerically, ALB was greater in 1991 for each zone of the natural seasonal wetland compared to the created wetland; however, owing to variability between transects and sampling sites, ALB within zones in 1991 was not different ($P<0.05$) between wetlands.

The range of ALB found in this study is similar to those of previous reports. Green (1987) and Mings et al. (1989) determined seasonal, freshwater wetland ALB for wet meadow zones ranging from 2,300 to 7,300 kg/ha. In a review, Fulton et al. (1986) reported typical wet meadow zone yields in freshwater wetlands to average approximately 4,000 kg/ha (2 st/acre). According to Fulton et al. (1986), aboveground production by shallow marsh species averages approximately 7,000 kg/ha (3.5 st/acre), or nearly twice that of wet meadow species. Rey et al. (1990) reported a range of 8,350 to 23,200 kg/ha for shallow marsh zones,
while Brinson et al. (1981) found net primary production in cattail-dominated (*Typha*) ecosystems ranging from 3,300 to 24,500 kg/ha.

Fluctuating moisture regimes greatly affect vegetation zonation and ALB in northern plains seasonal wetlands (Smeins 1967, Millar 1973, Kantrud et al. 1989). Kantrud et al. (1989) stated that high water levels can kill up to 25% of the emergent vegetation in prairie wetlands. Green (1987) reported annual ALB in a seasonal wetland increased from 3,750 to 7,200 kg/ha owing to an improved moisture regime. Dry conditions were experienced throughout the present study. This is most evident in the ALB estimated for the created wetland, where community evolution was still taking place. Despite the dry conditions, ALB in the natural wetland was favorable when compared to other reported values.

Table 1. Species identified in a created wetland on the Falkirk Mine, Underwood, ND, 1991-92.

Scientific Name[1]	Common Name	Zone[2]
Agropyron caninum	Slender wheatgrass	LP, WM
*Agropyron repens**	Quackgrass	LP, WM
Agropyron smithii	Western wheatgrass	LP, WM
Amaranthus albus	Tumble pigweed	SM, WM
Ambrosia artemisifolia	Ragweed	SM, WM
*Aster ericoides**	White prairie aster	WM
Aster puniceus	Swamp aster	SM
*Bromus inermis**	Smooth brome	LP, WM
*Calamagrostis inexpansa**	Northern reedgrass	SM
*Carex atherodes**	Slough sedge	SM
*Carex lanuginosa**	Wooly sedge	SM
Chenopodium album	Lambsquarter	WM
*Cirsium arvense**	Canada thistle	SM, WM
Descurainia sophia	Flixweed	SM, WM
*Eleocharis palustris**	Creeping spikerush	SM
Helianthus annuus	Common sunflower	WM
Kochia scoparia	Fireweed summercypress	SM, WM
Lactuca pulchella	Blue wild lettuce	SM, WM
Lactuca serriola	Prickly lettuce	SM, WM
Malva rotundifolia	Running mallow	SM, WM
Melilotus officinalis	Yellow sweetclover	LP, WM
Mentha arvensis	Mint	WM
*Phalaris arundinacea**	Reed canarygrass	SM
Poa pratensis	Kentucky bluegrass	LP, WM
*Polygonum coccineum**	Swamp knotweed	SM
Populus deltoides	Cottonwood	SM, WM
Rumex mexicanus	Mexican dock	SM, WM
Rumex pseudonatronatus	Dock	SM, WM
Salix amygdaloides	Peachleaf willow	SM, WM
Salix exigua	Sandbar willow	SM, WM
Salsola kali	Russian thistle	SM, WM
*Scirpus fluviatilis**	River bulrush	SM
*Scirpus validus**	Softstem bulrush	SM
Sisymbrium altissimum	Tumbling mustard	SM, WM
*Sonchus arvensis**	Sow thistle	WM
*Sparganium eurycarpum**	Bur-reed	SM
Stipa viridula	Green needlegrass	LP, WM
Taraxacum officinale	Common dandelion	WM
*Teucrium occidentale**	Germander	WM
Thlaspi arvense	Field pennycress	SM, WM
Tragopogon dubius	Goatsbeard	WM
*Typha glauca**	Blue cattail	SM

[1]Starred species are native to northern prairie wetlands.
[2]LP = low prairie; WM = wet meadow; SM = shallow marsh.

Biomass Decomposition

Decomposition rates for both zones and wetlands were slow (fig. 1). Decomposition of incubated plant materials never reached 50% even after 104 or 114 weeks. Plant detritus remains in northern prairie wetlands for extended time periods, which agrees with previous studies (Mason and Bryant 1975, Brinson et al. 1981, Polunin 1984, Murkin et al. 1989).

Aboveground decomposition rates for both vegetative zones were similar ($P<0.05$) between wetlands (fig. 1). However, belowground decomposition rates were greater ($P<0.05$) at and following 16 weeks incubation in both vegetative zones of the created wetland compared to the natural wetland.

Over time, aboveground and belowground decomposition rates increased ($P<0.05$) in both zones of the created wetland and in the aboveground environment of the natural wetland (fig. 1). Belowground decomposition rates of the natural wetland were similar ($P<0.05$) over the 114-week incubation period. Mean decomposition and rate of decomposition (slope) were greater ($P<0.05$) in the created seasonal wetland except aboveground in the shallow marsh zone, where mean and rate were similar.

The decomposition rates experienced can be explained by the continued dry conditions throughout the study. Absence of surface or subsurface water and warmer temperatures in the absence of water would lead to aerobic conditions, which would increase decomposition rates (Godshalk and Wetzel 1977, 1978, Polunin 1984, Ohlson 1987). Aerobic conditions existed throughout the study for aboveground and belowground incubated materials for the created wetland and for aboveground incubation bags for the natural wetland. Decomposition rates for these zones and wetlands followed similar patterns. Decomposition rates for belowground incubated materials in the natural wetland were lower and did not increase over the study period. This level and pattern of decomposition would be typical of anaerobic environments that may be exhibited in belowground shallow marsh vegetative zones.

Table 2. Aboveground net primary production (x + standard error) for a natural and a created seasonal wetland in central North Dakota kg/ha.

	Zone	
Wetland[1]	Wet meadow	Shallow marsh
Natural:		
1990	11,536 (792)	4,781 (3,028)
1991	11,886 (1,516)	8,521 (1,574)
Created:		
1991	4,920 (2,334)	5,480 (1,682)
1992	2,111 (875)	2,451 (1,311)

[1]No differences ($P<0.05$) were found between years within wetlands for each zone; No differences ($P<0.05$) were found in 1991 between wetlands for each zone.

Conclusions

The extended dry conditions experienced throughout the study significantly influenced the development of the created seasonal wetland. Numerous "weedy" pioneer species dominated the composition of vegetation zones in this wetland. However, common dominant native wetland species were present and hold promise for future recolonization of the site. Annual ALB was strongly influenced in the created wetland by the lack of moisture. Aboveground and belowground decomposition patterns in the created wetland were similar to aboveground decomposition in the natural wetland. This pattern would be indicative of dry, warm aerobic soil conditions in contrast to expected wet, cool anaerobic soil environments of natural seasonal wetlands. Overall, decomposition rates of plant materials are very slow in northern prairie wetlands, which is supported by the results of this study.

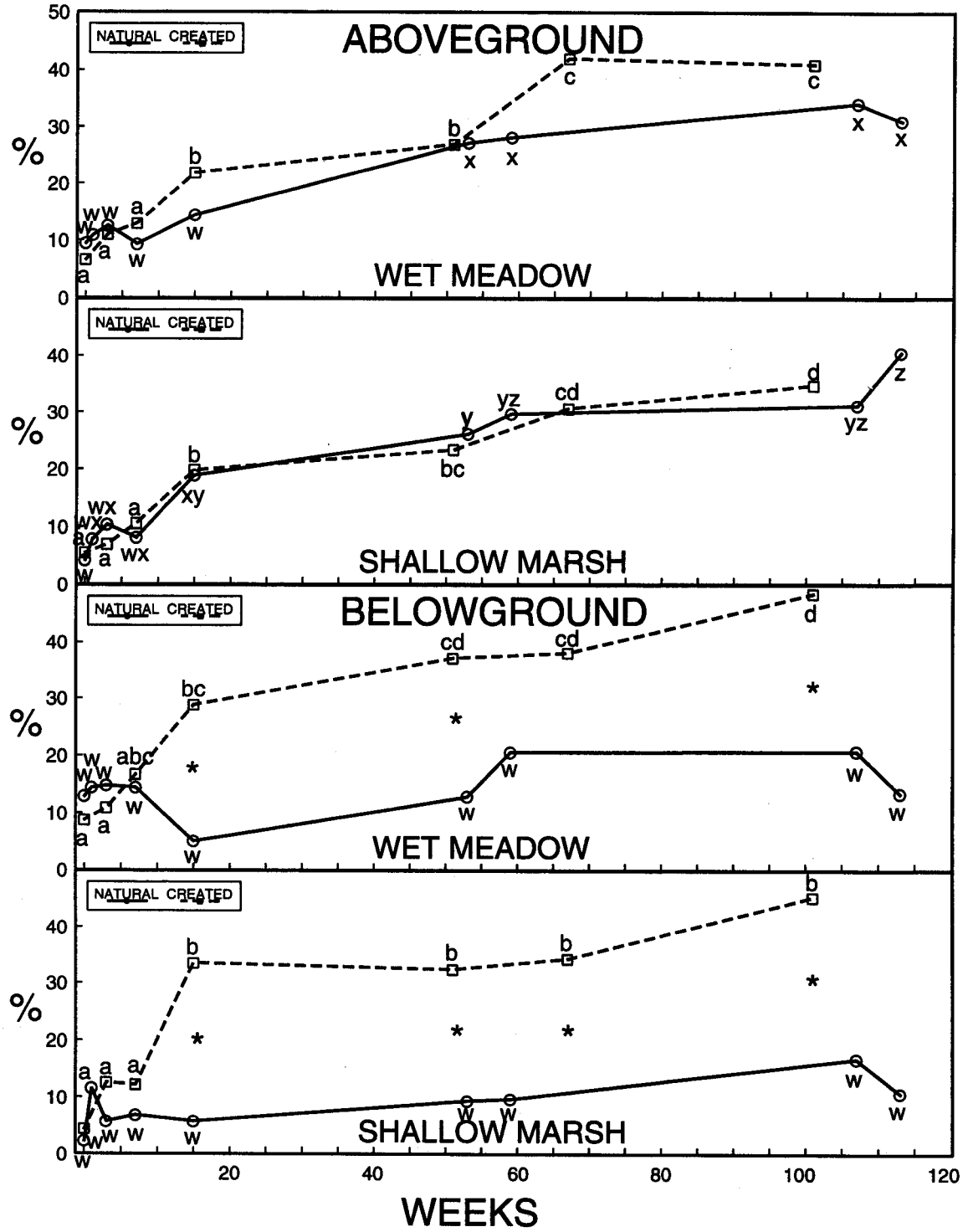

Figure 1. Decomposition rates for aboveground and belowground plant material incubated in wet meadow and shallow marsh vegetative zones of a natural and a created seasonal wetland.

* Difference (P<0.05) between natural and created wetland decomposition rate for an incubation stage.

abcd Difference (P<0.05) in decomposition rate over time for the created wetland.

wxyz Difference (P<0.05) in decomposition rate over time for the natural wetland.

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BIOTIC DEVELOPMENT COMPARISONS OF A WETLAND CONSTRUCTED TO TREAT MINE WATER DRAINAGE WITH A NATURAL WETLAND SYSTEM[1]

Harold J. Webster[2], Michael J. Lacki[3], and Joseph W. Hummer[2]

Abstract: Using 5-yr of baseline data from a constructed wetland, we compared the biotic changes in this wetland to conditions in a natural wetland to determine if biotic development patterns were similar. The constructed wetland was built in 1985 to treat a coal mine discharge and was planted with broadleaf cattail (*Typha latifolia*) within the three-cell, 0.26 ha wetland. The natural wetland was also dominated by broadleaf cattail in the emergent zone and was 0.17 ha in size, excluding the adjacent pond. The natural wetland was located immediately downstream from the constructed wetland and received comparable water flows. Species richness in permanent quadrats of the constructed wetland declined over the study period, while cattail coverage increased. Plant species composition diversified at the edges, with several species becoming established. The constructed wetland deepened and expanded slightly in area coverage during the study period. The constructed wetland supported herptofaunal communities that appeared more stable through time than those of the natural wetland and sustained a rudimentary food chain dependent upon autotrophic algal populations. Despite fundamental differences in substrate base, morphology, and water flow patterns, biotic trends for the constructed wetland coincided with succession-like patterns at the natural wetland. We suggest that further shifts in the biotic composition of the constructed wetland are likely, but the system should continue to persist if primary production meets or exceeds the microbial metabolic requirements necessary to treat mine drainage.

Additional Key Words: cattail, *Typha*, succession, herptofauna

Introduction

Construction of wetlands for mine drainage treatment is a widely used reclamation practice, despite gaps in the present understanding of how these systems function and uncertainty about how they will persist through time (Wieder 1989). Several authors have alluded to the importance of hydrologic patterns for the maintenance and stability of constructed wetlands (Odum 1987; Niering 1989, 1990; Kusler and Kentula 1990). Fluctuations in water levels are inherent to the integrity of any wetland, resulting in E. P. Odum's (1971) concept of wetlands as pulsing systems. Niering (1990) has indicated that predictable changes in the vegetation of constructed wetlands are probably not possible due to continual changes in hydrology, and that the objective of reclamation efforts should be the establishment of "self-perpetuating systems" and not specific vegetation types.

Cairns (1985) indicated that a thorough understanding of the successional processes in constructed wetlands is necessary for establishment of systems that perpetuate through time without periodic intervention or maintenance. However,

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[2]Harold J. Webster, Associate Professor of Biology, and Joseph W. Hummer, Instructor of Wildlife Technology, The Pennsylvania State University, DuBois Campus, DuBois, PA, USA.

[3]Michael J. Lacki, Assistant Professor, Department of Forestry, University of Kentucky, Lexington, KY, USA.

application of traditional concepts of succession to wetlands have failed to fully explain vegetational changes (Mitsch and Gosselink 1986). The use of the terms succession" and "climax community" are no longer recommended when addressing wetland plant communities. Niering (1987) has suggested that these terms be replaced by "biotic development" and "steady state," respectively, relative to cyclic changes in wetland vegetation and oscillations in hydrologic flow.

The interpretation of the vegetation mosaic in mine drainage wetlands is complicated because these sites are without a fully developed soil column (Kusler 1990) and are initially established with transplanted plant species (Hedin 1989). Broadleaf cattail (Typha latifolia) has been most frequently planted (Hedin 1989), and most sites start out with monotypic stands. Niering (1989) has postulated that continued self-maintenance of these wetlands may be due to superior competitive capabilities of cattail and/or allelopathic chemical inhibition of colonizing plant species. The absence of seed sources in the compost substrate, coupled with deeper water in many systems, further delays establishment of additional plant species. Consequently, long-term data on changes in plant community composition for mine drainage wetlands are needed to determine whether ingress of colonizing plant species is possible and/or preferred in wetlands where Typha has been planted as the dominant founder species.

In this paper we report the floral and faunal changes over a 5-yr period in a wetland constructed to treat a coal mine discharge. These findings are compared with data for a natural wetland on an abandoned beaver impoundment, immediately downstream from the construction site, to determine if biotic development in a nearby natural wetland could be used to predict changes over time in a constructed wetland. The data set used in this assessment represents a synthesis of previously reported findings with heretofore unpublished results.

Study Sites

Construction of the mine drainage wetland (Simco #4) was completed in November 1985 in Coshocton County, OH (Stark et al. 1988). The substrate in the three-celled wetland consisted of mushroom compost. Broadleaf cattail rhizomes were transplanted at a density of three to four rhizomes per square meter. The wetland was initially 0.26 ha in size and was receiving a deep mine discharge with a pH near 6.0 and an iron loading of 80 to 241 mg/L. Flow rates through the wetland average 328 L/min, but fluctuate with seasons. Overall, water treatment efficiency has improved with the age of the wetland (Stark et al. 1990).

The natural wetland developed on an abandoned beaver dam impoundment located directly downstream from the constructed wetland system. The natural wetland was 0.17 ha in size and was connected with a pond immediately downstream from the wetland. The wetland supported an emergent plant community dominated by broadleaf cattail. Hydrologically, the natural wetland received essentially the same water volume as that of the constructed wetland as only a series of three sedimentation ponds separated the two sites. The water chemistry in the natural wetland was improved by the upstream treatment of the mine drainage in the constructed wetland system. Mining activity ceased in the vicinity of the natural wetland in 1961, but unfortunately no historical information is available on the beaver dam and its wetland. The Simco #4 Mine began operation in 1970 and was sealed in 1979, with water treatment for the discharge commencing in 1980.

Methods

Forty permanent vegetation quadrats, 0.5 m^2 in size, were established in a grid pattern in the constructed wetland in 1985. The vegetation was annually inventoried from 1986 through 1990 for species coverage (Daubenmire 1970). Beginning in 1987, cattail density in numbers per square meter was also

quantified (Stark et al. 1990). In August 1988, the natural wetland vegetation was inventoried. Seventeen 1 m^2 quadrats were randomly located. Vascular plant species were identified, and their percent cover was recorded. Cattail density was quantified as in the constructed wetland.

Surveys for reptiles, amphibians, and mammals were conducted at both the constructed and natural wetlands from May 19 to 25 and June 10 to 15 in 1988, 1989, and 1990. All reptiles, amphibians, mammals, and signs of each were surveyed in the same manner (Lacki et al 1991a, 1991b). Coefficients of variation (CV) were then derived for total abundance and species richness (i.e., number of species), with data for reptiles and amphibians combined (herptofauna) and examined separately from those for mammals. Coefficients of variation are computed by taking the standard deviation as a percentage of the mean. Comparison by this approach demonstrates relative stability among mean values by removing any size or scale factor (Sokal and Rohlf 1969).

Results and Discussion

Thirteen vascular plant species were found in the permanent quadrats in the constructed wetland. Six of the 13 species were found at coverage values of 2% or greater for at least one growing season (Table 1). Besides broadleaf cattail,

Table 1. Percent cover of vascular plants and algae at the constructed and natural wetlands with species whose coverage values exceeded 2% in at least 1 sampling year. The natural wetland was sampled in 1988 only.

	Constructed Wetland					Natural Wetland
Species	1986	1987	1988	1989	1990	1988
Typha latifolia.....	30	42	54	62	[1]	30
Leersia oryzoides....	4	22	11	1	6	20
Algae.....	5	12	3	12	0	0
Lemna minor.....	4	2	5	1	1	0
Alisma aquaplantago..	0	1	[2]	5	0	0
Phalaris arundinacea.	[2]	2	0	0	0	0
Eleocharis sp.....	0	0	0	0	0	11
Impatiens biflora....	0	0	0	0	0	7
Carex sp.....	0	0	0	0	0	5
Polygonum sagittatum.	0	0	0	0	0	3
Eupatorium maculatum.	0	0	0	0	0	3
Scirpus cyperinus....	0	0	0	0	0	3

[1]Not measured for that particular year.

[2]Found at less than 1%.

only rice cutgrass (Leersia oryzoides) and duckweed (Lemna minor) consistently occurred in the quadrats across the 5-yr of study. A general trend of declining richness in the number of vascular plant species has occurred with time in the constructed wetland quadrats. The number of species recorded in quadrats from 1986 to 1990 was 8, 9, 6, 4, and 3, respectively. This decline does not reflect changes along the edge of the wetland, where several herbaceous and woody species have volunteered. Concurrent with the decline in species richness in the sample quadrats, the wetland depth increased and a slight increase in area occurred as the water surpassed the tops of the berms, particularly in the first cell.

The natural wetland was also dominated by broadleaf cattail and rice cutgrass, but differed from the constructed wetland in having a strong representation of three additional species: spikerush (Eleocharis sp.), touch-me-not (Impatiens biflora), and sedge (Carex sp.). The natural wetland also supported a richer vascular plant species community as 23 species were identified as present in at least one quadrat, with three additional dicot species detected but not identified. The natural wetland exhibited three zones or belts of vegetation along a gradient running from the drier wetland vegetation, near the source where a portal provided water from the upstream sedimentation ponds, to the ponded water adjacent to the natural wetland. These vegetation belts were reflected in shifts in the number of vascular plant species recorded in quadrats (fig. 1). Species richness was low adjacent to the

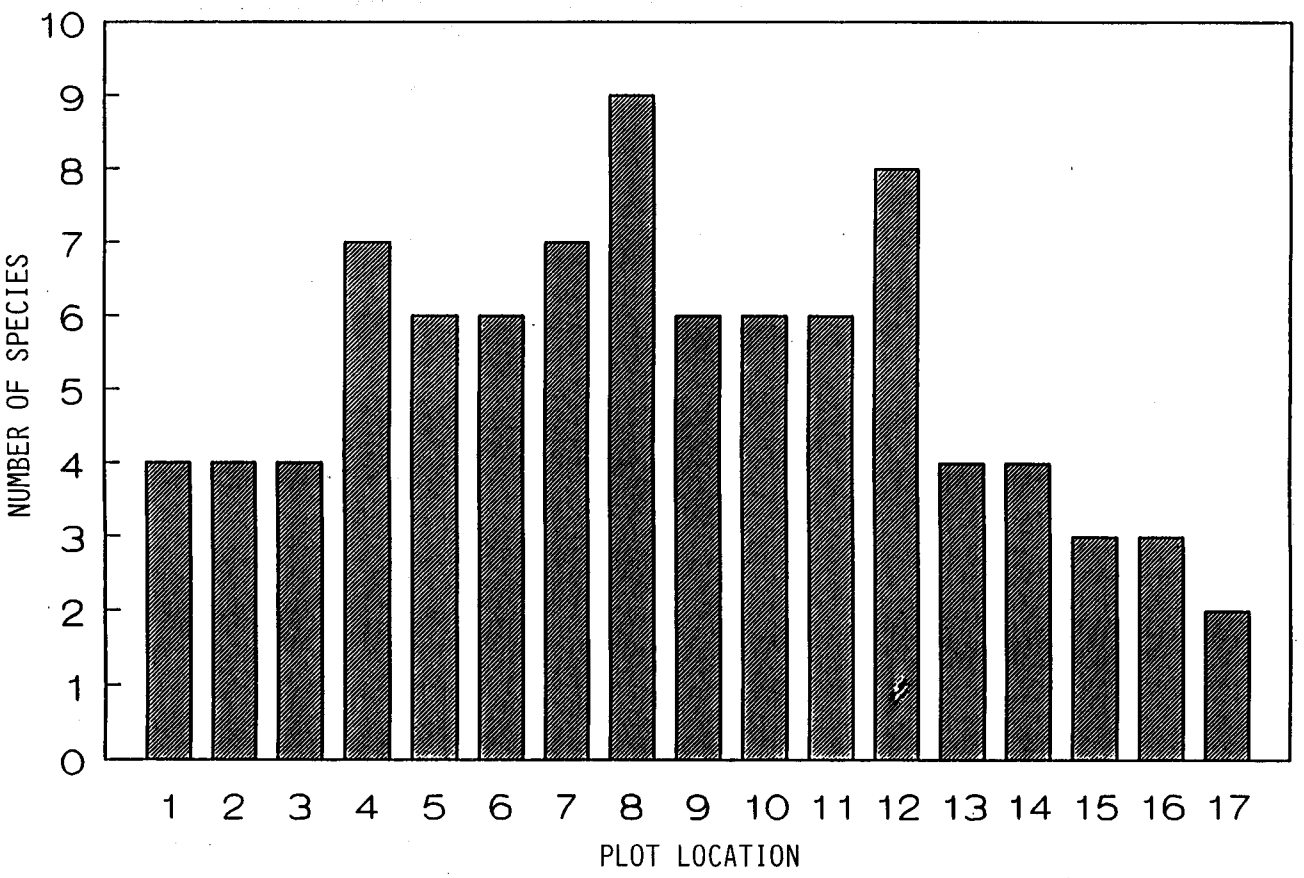

Figure 1. Number of species recorded per quadrat for samples obtained in the natural wetland, summer 1988. Plot locations represent a gradient starting from the upstream end (1) and continuing downstream to the edge of the wetland adjacent to the ponded water (17). Vegetation belts are noticeable by shifts in species richness at quadrat 4 and quadrat 12.

terrestrial habitat, increased to a maximum in the midportion of the wetland, and declined again as the wetland encroached upon the ponded water. At the upstream end, the natural wetland was more channelized and contained a firm substrate. Soils became more loosely compacted in a downstream direction, being extremely soft and unconsolidated adjacent to the ponded water. At this end the natural wetland exhibited an overall appearance not unlike that of the constructed wetland, with a reduced species richness and very deep mucky soils. The downstream end of the natural wetland was dominated by broadleaf cattail, rice cutgrass, and wool grass (Scirpus cyperinus).

Coverage values for broadleaf cattail increased in the constructed wetland as the system aged, surpassing the value for the natural wetland in 1988 by an additional 19%, suggesting that dominance by broadleaf cattail was increasing through time. Cattail density fluctuated among years in the constructed wetland, but remained at a density greater than that in the natural wetland across all years sampled (fig. 2). Narrowleaf cattail (Typha angustifolia) was observed to first invade the constructed wetland in 1988 and has continued to increase in area coverage since, particularly at the edges of the wetland cells. This species was absent from the permanent quadrats, thus actual coverage values are unavailable. Presently, narrowleaf cattail appears to be outcompeting broadleaf cattail at a microsite level. Algae was evident throughout the manmade wetland in the vegetation and vertebrate sampling. The four genera of algae recorded in the constructed wetland were Chlamydomonas, Euglena, Microspora, and Oscillatoria.

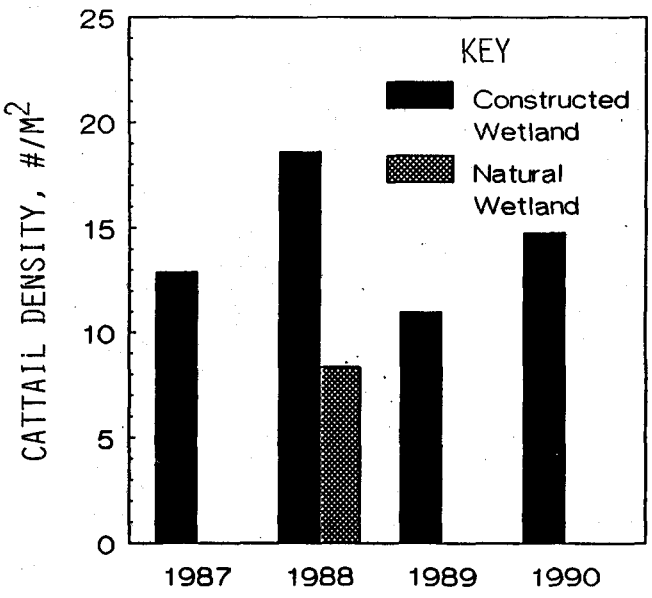

Figure 2. Density of cattails (number of stems per square meter) by year for the constructed and natural wetlands. (Natural wetland sampled in 1988 only.)

The presence of algae in the waters of the constructed wetland is probably essential to development of a food chain network. The constructed wetland is used as breeding habitat by green frogs (Rana clamitans) and gray tree frogs (Hyla versicolor), with tadpole larvae of both species present in the constructed wetland (Lacki et al. 1991b). Tadpoles of these species are herbivorous and depend heavily on algal food sources. A large amount of rubble and rock debris was also located in close proximity to the constructed wetland, providing denning habitat for an abundant snake population (Lacki et al. 1991b). Several snake species were found using the constructed wetland, including the northern water snake (Nerodia sipedon), eastern ribbon snake (Thamnophis sauritus), and northern black racer (Coluber constrictor). Exploitation of the abundant frog populations in the constructed wetland by snakes, which also utilize the adjacent denning habitat, contributed to the diversity of the herptofauna (Lacki et al. 1991b). Other predators, such as raccoons (Procyon lotor) and muskrats (Ondatra zibethicus) undoubtedly used these frog populations. Muskrats are commonly considered to be herbivores, although consumption of frogs has been documented.

Examination of CV values showed the natural wetland to support a more variable herptofaunal community than the constructed wetland, particularly in abundance levels, where the standard deviation across sample periods exceeded the value for the mean (table 2). During periods of low rainfall a noticeable

surface water depth remained visible in the constructed wetland; this may be an important factor in sustaining the abundant frog populations and subsequently the rich variety of snakes inhabiting this wetland. The natural wetland appeared much more susceptible to "drying out" in times of limited rainfall, such as summer 1988. Data for mammals indicated the two sites to be comparable overall, with the natural wetland showing slightly higher variability in species richness. No obvious patterns emerged to permit a ready explanation for this discrepancy (Lacki et al. 1991a).

Table 2. Coefficients of variation (CV) for abundance and species richness of herptofauna and mammals at the constructed and natural wetlands, 1988, 1989, and 1990 combined. All measures based on 6 survey periods per site.

Taxon by wetland type	Abundance (CV)	Species richness (CV)
Herptofauna		
Constructed...	35.9	32.4
Natural.....	102.0	41.3
Mammals		
Constructed...	48.7	23.3
Natural.....	43.9	34.8

Several fundamental differences existed between the constructed and natural wetlands. The constructed wetland was manually planted, thus modifying substantially the early stages in biotic development. With its volunteered vegetation, the natural wetland composition was more reflective of typical patterns found for Typha-dominated wetlands. The natural wetland had available a long length of time for colonization, although the age of the beaver dam is unknown. Substrate texture was primarily silt and organic matter in the natural wetland, whereas the constructed wetland was based upon mushroom compost with an accumulation of iron precipitates. The accumulation of metallic precipitates may inhibit colonizing plant species, but this is an area that requires further exploration. The natural wetland exhibited an irregular morphology, with water flow tending to meander through the site in defined channels. Morphological and hydrologic patterns for the constructed wetland were much more regular.

Kadlec (1989) has found that for wetlands constructed to treat secondary sewage, plant diversity declined with age of the wetland. He suggested that systems designed to maintain continual water flow and pollutant inputs favored cattails over other vascular plant species. Our data for the constructed wetland support this finding (table 1). Observations at the constructed wetland indicate that the water increased in depth and the system expanded in surface area. These hydrologic changes offer opportunities to examine responses by the peripherally established vegetation.

In contrast to the simplification of the vegetation at the constructed wetland, amphibian populations were afforded more optimal habitat conditions because of the availability of a free water surface. The stable measures of abundance and species richness observed at the constructed wetland support this contention (table 2). We predict that, as long as source water quality permits reproduction by frogs, wetlands constructed for the treatment of mine water drainge will sustain frog populations. Brooks et al. (1987) indicated that it is impractical to expect treatment wetlands to provide habitat for wildlife. Based on the performance of the Simco constructed wetland (Lacki et al. 1991a, 1991b, 1991c), we contend that such a position is premature and that each wetland construction project should be evaluated for its wildlife habitat potential.

Future effects of changes in the constructed wetland on other wildlife inhabiting the site is unclear. Meadow jumping mice (Zapus hudsonius) have been recorded at the constructed wetland in the grassy habitats adjacent to the wetland (Lacki et al. 1991a). Expansion of the wetland will likely lead to an

elimination of this cover type and a loss of this species from the immediate vicinity. Wetland expansion will also lead to input of iron-laden water into nearby vernal pools and pockets of unimpacted water located adjacent to the constructed wetland. The impact of a changed water quality in these microsites on other anuran species that do not use the constructed wetland for larval development remains unknown (Lacki et al. 1991b). These species include bullfrogs (Rana catesbeiana), pickerel frogs (R. palustris), American toads (Bufo americanus), and Fowler's toads (B. woodhousei). All other wildlife species recorded in the vicinity of the wetland should remain unaffected by wetland expansion.

More long-term studies are needed to fully evaluate the ultimate fate of broadleaf cattails in constructed wetlands. Although cattails are viewed as highly adaptive and capable of invading almost any wetland construction project (Odum 1987), data concerning cattail responses to water level changes are varied. Kadlec (1989) and this study found cattail respond favorably to sustained water depths. The possibility exists that in constructed wetlands where limited fluctuations in hydrologic patterns occur and iron precipitates accumulate, cattails may eventually decline in abundance if the precipitates suffocate the plants or form concretions at the surface. The time frame involved with such a phenomenon is unclear. Substrate consistency and water depth may also be important for the establishment of invading species, although comparable patterns of cattail dominance and species richness were observed in both the natural and constructed wetlands where a deep unconsolidated substrate occurred. Invasion of the Simco constructed wetland by narrowleaf cattail may reflect a lack of autotoxic effects from broadleaf cattail on germination and development of other Typha species (Sharma and Gopal 1978).

Wyngaard (1985) has suggested that wetlands be viewed from the perspective of performance, with wetland health assessed by examining wetland processes for possible malfunction. Bedford and Preston (1988) have indicated that optimization of all wetland functions within a given system is unrealistic, and Garbisch (1989) has pointed out that even existing wetlands seldom, if ever, maintain all wetland functions at a meaningful level. Alternatively, Cairns and Pratt (1985) recommend that constructed wetlands should only be expected to fall within the range of variability observed for natural wetlands. Based on the above arguments, we suggest that despite fundamental differences in substrate base, morphology, and water flow patterns, biotic trends for the constructed wetland coincided with succession-like patterns at the downstream end of the natural wetland. Further, the wildlife habitat utilization and food chain components were verified for the Simco wetland, documenting additional wetland functions for this system. A paucity of vascular plant species may be an artifact of the design of treatment wetlands, but Niering (1990) has indicated that wetlands should not be expected to support identical plant compositions, regardless of similarity in design, hydrology, and location within the landscape framework. We anticipate that further shifts in the biotic composition of the constructed wetland are likely, but the system should persist if primary production meets the microbial metabolic requirements for mine drainage treatment.

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ECOLOGICAL BENEFITS OF PASSIVE WETLAND TREATMENT SYSTEMS DESIGNED FOR ACID MINE DRAINAGE: WITH EMPHASIS ON WATERSHED RESTORATION[1]

Eric C. McCleary and Douglas A. Kepler[2]

Abstract: Western Pennsylvania has been a large source of coal for much of the United States since the late 1800's. During the extraction of the coal resources, acid mine drainage (AMD) often resulted. AMD from abandoned discharges has effectively rendered thousands of kilometers of streams lifeless in the Appalachian coal region. Restoration of these streams has been limited in previous years primarily because of the lack of cost-effective treatment for AMD. Conventional treatment (i.e., chemical addition) can treat AMD effectively but is costly to operate and maintain and is effective only when receiving human attention. Passive wetland treatment systems have proven to be the only realistic AMD treatment strategy, in terms of watershed restoration activities. If ecosystem health is the reason for implementing effluent standards then it can be reasonably argued that passive wetland treatment systems supply the most effective overall treatment, even if they do not meet one or more of the current effluent standards. Recent advancements in passive wetland treatment system technology have provided a management tool that could be used to treat the majority of AMD discharges cost-effectively, and when used strategically could reasonably be employed to restore the thousands of kilometers of AMD-affected streams in the coal regions of Appalachia. Secondary benefits that have been observed with passive wetland treatment systems suggest that these systems may be providing for accelerated ecological recovery independent of regulated effluent standards.

Additional Key Words: acid mine drainage, passive wetland treatment system, watershed restoration.

Introduction

Water of a quality consistent with biological functioning is critical to all life. Increasing human populations and the resulting consumption of our resources have necessitated that human society divert more attention to the preservation and restoration of our water resources.

The water resources of the Appalachian coal region have been particularly impacted, in terms of water quality degradation. Western Pennsylvania, for example, has been a large source of coal for much of the United States since the late 1800's. As of 1986, Pennsylvania contributed roughly 6.9% of all coal produced in the United States (Keystone Coal Industry Manual 1987). The extraction of coal contributed largely to the economic development in the coal mining regions, while leaving chronic environmental debts. These environmental debts have taken two basic forms: (1) incomplete or absent surface reclamation and (2) acid mine drainage (AMD). Surface reclamation today is not a serious problem largely because of restoration programs enacted by both State and Federal entities. AMD has been and remains the largest contributor to water quality degradation in most coal mining regions. The formation and cause of AMD are well documented (Hedin et al. 1994, Brodie 1993, Brodie et al. 1993, and Stumm and Morgan 1970). Surface and subsurface mining has left thousand of untreated AMD discharges flowing into thousands of streams in this region. AMD and its associated high metal and high acid content have rendered approximately 12,000 km of streams and 12,000 ha of impoundments biologically devastated (Appalachian Regional Commission 1969).

In 1977 the Surface Mining Control and Reclamation Act (SMCRA) was enacted creating the Office of Surface Mining, Reclamation, and Enforcement (OSMRE) within the U.S. Department of the Interior. OSMRE is the Agency responsible for regulating coal mining and reclamation in the United States through SMCRA. Basically, SMCRA requires that coal operators treat point source discharges that are regulated by the National Pollution Discharge Elimination System (NPDES) permits.

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[2]Eric C. McCleary, Restoration Ecologist, and Douglas A. Kepler, Restoration Ecologist, Damariscotta, Clarion, PA, USA

The Federal Clean Water Act of 1972 entitles the U.S. Environmental Protection Agency (EPA) to set standards that each AMD discharge from a coal mining activity must meet. These standards for mining Applications are between 6 and 9 for pH, 7 mg/L daily and less than 3.5 mg/L on a monthly average for Fe, 4 mg/L daily and less than 2 mg/L on a monthly average for Mn. State regulatory agencies may set more stringent standards.

Coal operators, while operating, have generally met the criteria set forth by the EPA. However, in cases where treatment ceases either temporarily or permanently the aquatic environment receiving the AMD discharge once again receives degraded water, adversely affecting the biota. With the prevailing economic situation, coal operations once thriving are now experiencing economic hardship. Because of this hardship, many coal operations have ceased and many more are in the process. This has left the States and Federal Government with the ultimate responsibility of AMD abatement under he Clean Streams Law of 1972. While bonds were allocated for each permitted coal extraction activity, the bond remaining after forfeiture is insufficient to allow for AMD treatment in perpetuity with conventional treatment (i.e., chemical). Thus, abandoned discharges, both those with bonds and those created prior to bonding requirements (1977), are flowing into the streams and rivers of the United States unabated.

Water affected by AMD require various forms of treatment before it can be considered an acceptable resource for most aquatic biota and human uses. In an effort to meet the standards imposed by the EPA, two types of treatment have emerged: active and passive. active treatment commonly consists of chemical addition in the form of soda ash briquettes, hydrated lime, quick lime (CaO), or sodium hydroxide. Being inherently acidic, AMD generally possesses low pH and elevated Fe and Mn concentrations. Chemical addition effectively raises pH, causing the chemical-physical precipitation of typical metals dissolved in AMD. Chemical treatment is effective but costly, in terms of the chemicals, maintenance of the treatment system, and the overall effects to the receiving aquatic environment (Hedin et al. 1994, Brodie et al. 1993). Passive treatment generally includes systems that are designed around constructed wetlands that require little in terms of operation and maintenance costs. These systems are generally less effective at meeting effluent criteria than chemical treatment but possess the secondary benefits of perpetual, year-around treatment, minimal operation and maintenance costs, and great potential for ecological restoration.

Restoration of AMD-affected waterways has been limited primarily because of insufficient funds allocated for such purposes, acceptability of passive treatment system technology by the controlling regulatory agencies, and the foresight of these agencies to predict the cost of chemical treatment to the coal industry and the environment. There are currently talks underway to appropriate existing funding, within the State and Federal Governments, to allow for treatment of existing AMD discharges; however, two questions will have to be answered: (1) who will ultimately pay for the abatement and how will it be prioritized and (2) what type of abatement will achieve the most cost-effective results.

This paper is intended to support the contention that passive wetland treatment systems can be used to successfully mitigate the effects that AMD has had on the ecology of the Appalachian coal region streams. Passive wetland treatment systems appear to provide the most cost-effective treatment of AMD discharges, with secondary benefits of providing for increased ecological recovery (Williams and Dalby 1993). The approach of this paper is to elucidate on the various passive treatment systems that can effectively provide a solution to the above second question. If the intent of the original law regarding effluent standards is the maintenance of overall ecosystem health, then it can be reasonably argued that passive wetland treatment systems would supply the most effective overall treatment, even if they do not meet one or more of the current effluent standards.

Discussion

Cost Comparison Rationale

One of the purported advantages of passive wetland treatment systems is the supposed costs savings in terms of perpetual treatment. The following example should illustrate the potential for cost savings, when properly designed passive wetland treatment systems are implemented site specifically:

The R.E.M. Orcutt/Smaill wetland was constructed in 1987 to deal primarily with the high iron loadings (>20,000 g/d). The initial design consisted of an aerobic wetland that was sized to the available area (roughly 0.12 hectares). The source AMD discharge was net acidic (possessing acidity values greater than 200 mg/L) allowing for little abiotic precipitation of Fe. Removal rates did not meet expectations, and in the spring of 1992 revisions were made to the system that incorporated an anoxic limestone drain (ALD) and a successive alkalinity producing system

Table 1. Water quality data for R.E.M.Orcutt/Smaill site, Union County, Jefferson County, PA.

Monitoring location	pH[1]	Fe, (mg/l)	Acid, (mg/l)	Acidity loading, (g/d)	Fe loading, (g/d)
L1 (ALD)...	6.18	155.0	213	31,561	22,967
L2.....	4.00	46.3	161	23,856	6,860
L3.....	3.75	29.8	152	22,523	4,416
L4.....	3.33	25.8	146	21,634	3,823
L5(Effluent)	3.98	9.0	64	9,483	1,333

[1]Field Measurements
Note: Flow measured at 102.9 L/min.
Source: U.S. Bureau of Mines laboratory data, January 18, 1993.

(SAPS). These passive system additions significantly affected water quality; Fe removal went from 70%-75% to 95%. Acidity values associated with this discharge were also reduced from source values out of the ALD of >200 mg/L to the effluent acidity of 50-60 mg/L (associated mostly with Mn).

The costs to reduce the acidity values were calculated for both the passive system and a hypothetical chemical treatment system that would reduce the acidity to the same level. A 25% level of error was calculated into each treatment scenario. Values (U.S. Bureau of Mines, laboratory data 1993) for this particular discharge, referred to as L1 are shown for one date (18 January 1993) in table 1. This date was chosen as representative of a date on which passive treatment systems are considered to be at their theoretical lowest effective treatment level.

To calculate the cost of treatment for the passive treatment system, the alkalinity value of 125 mg/L out of the ALD was utilized as the $CaCO_3$ equivalent for alkalinity production with a 25% error to allow for greater alkalinity production than observed. Assuming 90% $CaCO_3$ content of the limestone, approximately 21 kg of limestone is dissolved daily at this site. With a cost of $15 per ton of limestone delivered, the annual treatment cost for this discharge is $125 per year. The installation costs of the ALD and SAPS were roughly $10,000 for materials and labor.

The costs to chemically treat this discharge to the same effluent acidity can be calculated by determining the acidity loading and inserting the value 149 mg/L (from table 1; subtract L5 acidity from L1 acidity) into a common equation utilized by the chemical treatment industry to determine the amount of chemical needed to neutralize the AMD discharge acidity:

g/d of acidity ($CaCO_3$) x 0.80 ($CaCO_3$ equivalence conversion factor) x 1/0.99 (% pure) x 1/0.90 (% reactive) x 1/0.75 (% effective) = g/d of NaOH required to neutralize specified amount of acidity.

To neutralize the same 149 mg/L of acidity chemically that is neutralized in the passive system roughly 33 kg of NaOH would be required per day. Sodium Hydroxide currently costs $0.40/lb, on average, which translates to approximately $29/day and $10,600/year for chemical treatment alone. Additional funds would be required for staff to operate the treatment equipment.

Brodie et al. (1993) quoted figures similar to these for the chemical treatment of a discharge that had been converted from chemical to passive. While these are a few examples, it can be seen that passive treatment systems possess the ability to literally pay for themselves within 2 to 3 years (1 year in the R.E.M. Orcutt/Smaill system). Passive treatment systems, with this type of payoff and little or no operation and maintenance, can, when properly designed, operate at or below standard effluent criteria 365 days/year.

Table 2. Passive wetland treatment systems, construction dates, and the average influent and effluent characteristics from implementation to present.

Site	Construction year	Influent Effluent	Sample Size, n	pH	Fe, mg/L	Mn, mg/L	Acidity, mg/L	Alkalinity, mg/L	Flow, L/min
Somerset	1984	I	43	4.4	162	50	373	0	47
		E	40	5.5	18	33	69	NG	---
Donegal	1987	I	29	7.1	5	8	NG	202	501
		E	28	7.4	<1	2	NG	NG	---
Emlenton	1987	I	40	4.7	89	77	320	15	55
		E	40	3.2	15	73	271	0	---
Latrobe	1987	I	43	3.5	125	32	617	0	86
		E	43	3.7	56	29	343	NG	---
Piney	1987	I	39	5.8	1	15	NG	60	468
		E	39	5.8	1	15	NG	NG	---
FH	1988	I	73	2.6	153	9	929	0	15
		E	73	2.9	137	10	674	0	---
Blair	1989	I	12	6.2	52	30	NG	166	11
		E	8	7.0	<1	5	NG	NG	---
Cedar	1989	I	26	6.3	92	2	NG	336	156
		E	27	6.4	41	2	NG	NG	---
Keystone	1989	I	28	6.3	37	<1	NG	142	8,606
		E	28	6.4	32	1	NG	NG	---
Shade	1989	I	20	6.0	<2	23	NG	31	10
		E	20	6.8	<1	10	NG	NG	---
Morrison	1990	I	34	6.3	150	42	NG	271	7
		E	24	6.6	<1	11	NG	NG	---

Source: Hedin et. al. 1994.
FH Friendship Hill National Historic Site.
NG Not Given

Other Concerns

When the financial aspect of treatment is addressed, it is apparent that passive treatment systems are relatively inexpensive to construct, operate and maintain. Time or longevity of the passive treatment system operating life is a question that often follows. Longevity, however, is a question that remains theoretical at this point, because the oldest passive treatment systems still functioning effectively are generally less than 10 years old (Hedin et. al. 1994). Passive systems show no signs of physically failing and should operate well into the future. Table 2 gives a list of systems, when they were implemented, and average water quality data over their life span. The longevity of passive wetland treatment systems can be calculated for all components of the systems (ALD's, aerobic portions, and SAPS') but provides little comfort to those financially responsible for treatment. However, when chemical treatment is examined in this light the longevity question is just as applicable. Where passive treatment systems are limited by natural physical constraints (e.g., plugging of substrates and limestone, accumulation of Fe and other metals), chemical treatment systems are limited directly by operation and maintenance costs. Chemical treatment has been in place for about 16 years under the mandate of SMCRA. Generally, the only surviving treatment systems are those that are being operated by corporations able to withstand the economic burden of operating such systems. Total water treatment costs for the coal mining industry are estimated to exceed $1 million per day (Kleinmann 1989). The number of corporations that are still able to generate the funds necessary to operate and maintain their systems is rapidly dwindling. Operation and maintenance of these systems would then ultimately become the responsibility of the State and Federal Governments. Very little funding has been allocated for treatment systems by either of these entities, and the realization that chemical treatment would be cost prohibitive is apparent (Fabian 1993). Thus, while chemical treatment does provide the theoretical optimum treatment scenario the lack of funds necessary to maintain operation and the inability of most operators to treat to theoretical effectiveness with chemicals result in less than optimum AMD treatment.

A question that often follows the longevity concern is of passive wetland treatment limitations: What types of AMD discharges can be effectively treated? The quality of the AMD and the area relative to flow and quality have

been the limiting factors in the past. Available area for passive wetland construction (as it relates to AMD flow and quality) is now the only real limiting factor. In situations where large flows of severe quality must be treated, physical limitations in regard to area must be considered. In some cases there is no recourse but to either combine chemical and passive treatment or utilize chemical treatment alone. Cases in which AMD flow and quality are actually limiting are relatively rare. In most instances, passive treatment systems will provide the proper solution to the AMD problem.

Quality of the AMD discharge was a significant factor in the past when only net alkaline and low metal contaminated waters were dealt with adequately (Burris et al. 1984). Recent technological advancements have allowed the treatment of net acidic, low pH, and high dissolved metal concentration AMD (see table 3). While these examples are relatively recent, the technology that has allowed these systems to function are based on well-known chemical, physical, and biological processes (Hedin et al. 1994, Stumm and Morgan 1970, Brodie et al. 1993).

The quality of some AMD discharges can still produce limitations when considering a passive treatment system design. Some AMD source qualities have yet to be incorporated into successful designs (i.e., high ferric iron and aluminum concentrations and high flows of fair to poor-quality AMD). The lack of successful treatment designs when dealing with certain AMD discharges is probably due more to the concentrations of past treatment efforts than to actual treatment limitations (Hedin et al. 1994). Through the perseverance of research-oriented institutions (e.g., U.S. Bureau of Mines and the Tennessee Valley Authority) and individual groups that are concerned with the restoration of waters affected by AMD, AMD that could not be treated passively a few years ago is now being treated effectively with passive systems (see table 3, Hedin et al. 1994). Passive treatment system technology is 15 to 20 years old, and large advancements have been made in the last 3 years. Significant water quality improvements in AMD discharges seem to be directly related to passive treatment system design (Brodie et al. 1993, Hedin et al. 1994, Kepler and McCleary 1994). Designs that rely on " black boxes," "magical " microbes, and "special" additives, or are implemented by individuals or organizations not familiar with the fundamentals of AMD treatment mechanisms are less reliable and have seriously affected the advancement of scientifically designed passive treatment systems.

Wieder (1989) examined 146 wetlands to determine if trends could be uncovered that would allow for *a priori* predictions as to the constructed wetland treatment efficiency. No trends were noted, which led Wieder to question the reliability and predictability of wetlands as acceptable treatment mechanisms. He was a spokesperson

Table 3. Water data and locations of three recently constructed passive wetland treatment systems.

Site location[1]	Sampling location	Flow, L/min	pH	Alkalinity, mg/L	Acidity, mg/L	Iron, mg/L
Damariscotta:[2]						
Schnepp Road....	Influent	54.5	3.2	0	300	45
	Effluent	---	7.0	56	0	2
USBM:[3]						
Howe Bridge[4]....	Influent DRI	137.0	5.87	32.0	507	307.12
	Influent PO	84.5	6.10	40.0	344	227.61
	Effluent	---	6.49	75.0	<10	47.68
USBM:[3]						
Orcutt/Smaill[4]....	Influent L1	135.6	5.84	107.0	275	199
	Influent R1	105.7	5.19	48.0	895	508
	Effluent	---	5.48	37.0	109	37.3

[1]All sites in Union Township, Jefferson, PA.

[2]Data from Damariscotta, 23 August 1993.

[3]U.S. Bureau of Mines laboratory data (4 May 1993, Orcutt/Smail; 27 July 1993, Howe Bridge).

[4]Two source discharges are present at both Howe Bridge, DRI and PO; and Orcutt/Smaill, L1 and R1. These discharges combine for one effluent discharge at both sites.

for OSMRE at the time of this publication, and his conclusions led regulatory agencies to question passive wetland treatment system approaches. The failure of this study to partition out designs that were based on sound scientific principles from those that were implemented by those unfamiliar with the basic principles led Wieder to the correct conclusions; namely, that a high degree of variability exists within the designs of passive treatment systems and as a result the predictability of such systems prior to implementation is small. Hedin et al. (1994) and Stark et al. (1990), in contrast to Wieder found that a great deal of predictability does indeed exist when passive treatment systems are examined that were designed on well-known principles. Kepler and McCleary (1994), when applying these basic principles in atypical fashion to AMD considered untreatable (passively), produced results consistent with these principles (see table 3). These recent advancements in wetland treatment system design indicate that passive wetland treatment systems are capable of treating a wide variety of AMD discharges to higher degrees of success than were previously believed possible (Hedin et al. 1994) and that the systems efficiency is predictable (Hedin et al. 1994, Kepler and McCleary 1994). A modification (fig. 1) of a U.S. Bureau of Mines flow chart provides *a priori* criteria necessary to predict passive wetland treatment system behavior. With these advancements in passive wetland treatment system technology, attempts are being made to apply these treatment devices as tools for watershed restoration.

Watershed Restoration of AMD-Affected Waters

The intent of water treatment in the past has been to preserve the quality of the receiving aquatic resources (Clean Water Act of 1972). The means through which this was to be accomplished were the chemical parameters set as standards by the EPA. These standards when met would theoretically preserve the quality of the receiving waterway. Under ideal conditions this goal would have been achieved; however, with limitations on funds

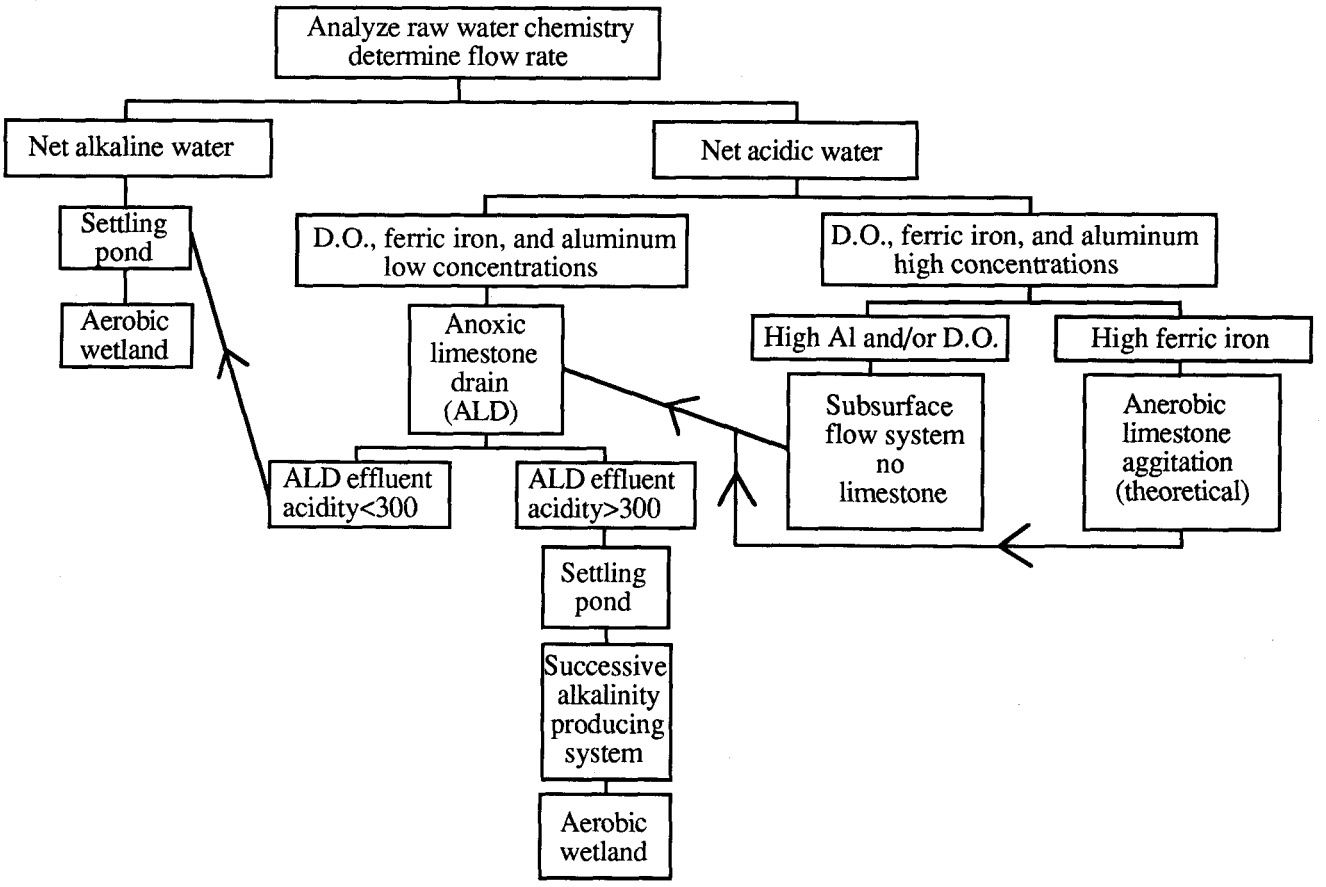

Figure 1. Flow chart showing dichotomous determinations necessary for design of passive wetland treatment systems. Adapted from Hedin et. al. (1994). D.O. dissolved oxygen.

necessary to implement and maintain treatment, these objectives were not met. In fact, the inability to react to the problem associated with these standard effluent criteria has arguably done more to negatively impact riparian health than it has to preserve the ecological health of existing waters (Environmental Protection Agency 1987, 1988a, 1988b, 1989a, 1990, General Accounting Office 1987).

Effective watershed restoration plans are being implemented by several watershed organizations in Pennsylvania, but the support from the regulatory agencies is instrumental in accomplishing the goals of these groups. This support is contingent upon the approval of treatment mechanisms that will be effective (cost and treatment) in abating AMD. The information presented in this paper suggests that passive treatment systems hold promise for such restoration efforts in terms of cost-effective, perpetual treatment. Passive treatment system efficiency can be predicted *a priori* (Hedin et al. 1994, Kepler and McCleary 1994), which can led to estimation of overall watershed impacts (McCleary and Kepler 1992).

A critical component of obtaining approval and implementing successful restoration programs is the ability to define success. Success is currently defined as the ability of the treatment mechanism to meet the Federal Clean Water Act (enacted 1972) mandate that all coal mining point source discharges meet water quality standards set by the EPA. While this definition of success seemingly incorporates an ecological component by proxy, the generality of the criteria does not account for local geographic variations, biological sensitivity, and/or cumulative impacts. Thus, treatment success is limited to the effluent of the treatment system with little regard for the receiving waterways.

Chemical and physical indicators have been the standard by which most forms of water pollution have been assessed (Karr 1991). AMD has not been an exception to this policy and despite massive regulatory efforts, continual declines in water quality and quantity have prevailed (Environmental Protection Agency 1987, 1988a, 1988b, 1989a, 1990, General Accounting Office 1987). A paradigm shift appears necessary in the monitoring and assessment of water resources from the chemical and physical to one that incorporates more of the biological component (Environmental Protection Agency 1984, 1987, 1988a, 1989b, 1989c) if watershed restoration is to become a reality. Some (Van Putten 1989) call directly for the development of biological criteria to protect water resources from negative impacts.

Restoration efforts require that all aspects of the environment associated with the water resource be measured in some capacity to provide an objective assessment of the restoration success. In some cases meeting a pH of 6.0 may not be necessary for the ecological recovery of the ecosystem, especially when background waters are typically less than 6.0. In other cases the addition of alkaline materials in sufficient quantities to neutralize the effects of the contaminants may actually contribute more to the receiving stream degradation than is easily observable.

Importantly, since restoration efforts are generally considered from a holistic standpoint, a measurement that incorporates many different types of parameters from the affected system should provide a more robust measurement of treatment success. The reduction of the definition of success to a few water quality parameters that may or may not reflect regional geographic and biological differences is not ecologically sound and may actually have contributed to the degradation of the riparian ecology in coal regions of Appalachia.

A primary concern of passive wetland treatment systems for AMD treatment is the ecological recovery of the streams affected by AMD. Recent data suggest that passive wetland treatment systems not only provide for water treatment, but can act to increase ecological recovery of the immediate area and when used strategically may act as an effective watershed management tool in areas affected by AMD (McCleary and Kepler 1992, Hedin et al. 1994, Williams and Dalby 1993) The restoration of the ecology in these devastated watersheds would provide positive impacts associated with the aquatic resources in these areas.

We are at a junction in AMD abatement that requires a definitive decision as to types of treatment mechanisms and scope of overall treatment. With the number of waterways that have been negatively impacted by AMD the scope of the problem is seemingly monumental. The cost of treatment is unrealistic if conventional chemical treatment is considered alone (Kleinmann 1989), and passive treatment cost is significant if all problems are considered collectively. For example, if each AMD discharge averages $20,000 to implement passive treatment and there are 1,000 discharges in the affected watershed then $20 million would be necessary to implement an abatement program for this watershed. Recent projections (U.S. Army Corps of Engineers 1981) have estimated restoration of AMD affected waters in the Clarion River Basin, located in western Pennsylvania, at $120 million (this study did not account for water treatment, only surface reclamation, which would have increased this figure

easily twofold.) Restoration programs would necessarily have to examine the benefit-to-cost ratio of the restoration before funds could be allocated for such projects. Recent conversations with the U.S. Department of Agriculture (Duncan 1993) suggest that the cost-benefit ratio for AMD watershed restoration would easily be higher than that normally required to initiate remedial actions (1.2 to 1.0).

Summary

The all-or-nothing response from the regulatory agencies in the coal mining regions of Appalachia must be reevaluated if ecological restoration is our goal. Years have passed with little being done to abate the AMD problems of this region, yet the technology exists that can positively impact streams receiving AMD. Passive wetland treatment systems , while not currently a solution for 100% of the AMD problems, hold the greatest promise for the effective, long-term treatment of such waters. While passive treatment systems may not always be consistent with current effluent criteria, their application as a tool for watershed management and restoration is great. Recent activities utilizing passive treatment systems as an approach to watershed restoration have met with great success. The restoration of thousands of kilometers of streams and hectares of impoundments in the Appalachian coal regions, affected by AMD, is dependent on the success of these early restoration efforts. The ability to objectively demonstrate that these restoration attempts will significantly enhance the quality of the receiving environment is imperative. Success must not be indicated by a few chemical parameters but a consideration of the intent of the Clean Water Act (1972), which includes secondary ecological benefits associated with passive wetland treatment systems. The secondary benefits associated with these systems include water resource recovery, wildlife habitat and restoration, and educational values.

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DRY FLUE GAS DESULFURIZATION BYPRODUCTS AS AMENDMENTS FOR ACID AGRICULTURAL SOILS[1]

Richard C. Stehouwer, Paul Sutton, and Warren A. Dick[2]

Abstract: Dry flue gas desulfurization (FGD) byproducts result from the removal of SO_2 from the stack gases of coal-fired boilers and are mixtures of coal fly-ash, $CaSO_4$, and unspent sorbent. Dry FGD byproducts frequently have neutralizing values greater than 50% $CaCO_3$ equivalency and thus have potential for neutralizing acid agricultural soils. Owing to the presence of soluble salts and various trace elements, however, soil application of dry FGD byproducts may have adverse effects on plant growth and soil and water quality. The use of a dry FGD by-product as a limestone substitute was investigated in a field study on three acid agricultural soils (pH 4.6, 4.8, and 5.8) in eastern Ohio. The by-product (60% $CaCO_3$ equivalency) was applied in September, 1992, at rates of 0, 0.5, 1.0, and 2.0 times the lime requirement of the soils, and alfalfa (*Medicago sativa* L.) and corn (*Zea mays* L.) were planted. Soils were sampled in April, 1993 and analyzed for pH and water soluble concentrations of 28 elements. Soil pH was increased by all FGD rates in the zone of incorporation (0-10 cm), with the highest rates giving a pH slightly above 7. At 10- to 20-cm, pH was increased from 4.7 to 5.2 in two soils; there was no effect on pH at 20- to 30-cm. Calcium, Mg, and S increased, and Al, Mn, and Fe decreased with increasing dry FGD application rates. No trace element concentrations were changed by dry FGD application except B which was increased in the zone of incorporation. Dry FGD increased alfalfa yield on the most acidic soil, and decreased corn grain yield on another soil. Application of dry FGD equivalent to the lime requirement of acid soils appears to be beneficial to acid-sensitive crops such as alfalfa. No detrimental effects on soil quality were observed in this study.

Additional Key Words: agricultural lime, beneficial use, coal ash, gypsum, liming, scrubber waste.

Introduction

The 1990 amendments to the Clean Air Act mandate a two-stage, 10-million ton reduction in annual SO_2 emissions in the United States by the year 2000. One strategy for meeting this standard is for utilities to scrub SO_2 from flue gases. Dry flue gas desulfurization (FGD) processes utilize lime-based sorbents in various scrubber vessels and produce highly alkaline byproducts which contain unspent sorbent, coal ash, and the SO_2 reaction product anhydrite ($CaSO_4$). These byproducts are generally disposed of in landfills. Because of limited landfill space and increased tipping fees, the development and demonstration of beneficial and environmentally safe uses for these byproducts would significantly reduce the cost of SO_2 scrubbing.

Because of their alkalinity, one possible beneficial use for these byproducts is as a limestone substitute for amendment of acid agricultural soils. Land application of fluidized bed combustion byproducts (one type of dry FGD byproduct) as a lime substitute and a source of Ca and S has been investigated in a number of studies (Terman et al. 1978, Holmes et al. 1979, Stout et al. 1979, Korcak 1980). These studies have generally reported positive effects on plant growth and crop yield, with negative effects occurring only at application rates of 25 wt % or higher. Most studies with fluidized bed materials have investigated soil pH and plant responses,

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[2]Richard C. Stehouwer, Senior Researcher, Paul Sutton, Professor or Agronomy, and Warren A. Dick, Professor of Agronomy, The Ohio State University, Ohio Agricultural Research and Development Center, Wooster, OH, USA.

with little emphasis on potential environmental impacts. In this study we have investigated the responses of two crops (alfalfa and corn) grown on three acid agricultural soils amended with a dry FGD byproduct applied at rates based on the liming requirement of the soils. In addition to crop responses, soil chemical effects and transport of the FGD material were monitored.

Materials and Methods

Field studies were conducted on three acid agricultural soils located at Wooster, Coshocton, and Canfield in eastern Ohio. Characteristics of these soils are given in table 1. Dry FGD byproduct was obtained from a pressurized fluidized bed combustion (PFBC) boiler. The PFBC boiler produced two byproduct streams: a coarse, granular bed material, and a much finer, particulate material collected in a primary cyclone (table 2). The PFBC byproduct used in this study was a 40:60 (wt/wt) mixture of the bed and cyclone materials.

The PFBC was applied in September 1992 at rates equivalent to 0, 0.5, 1, and 2 times the lime requirement (LR) of each soil. The soil LR was determined by use of the SMP buffer method (Shoemaker et al. 1962). The amount of calcium carbonate required to raise the soil pH to 7 was divided by the $CaCO_3$ equivalency of the PFBC (0.60) and multiplied by the lime requirement rate treatment factor to arrive at the amount of PFBC applied to each of the three soils (table 3). The PFBC was surface-applied using a lime spreader and then incorporated to a depth of 10 cm with a roto-tiller. Separate plots of each treatment were planted to alfalfa (*Medicago sativa* L.) and corn (*Zea mays* L.). Alfalfa was planted immediately after PFBC incorporation, while corn plots were seeded with a rye (*Secale cereale* L.) cover crop. In the spring the rye cover was killed with glyphosate (*N*-(phosphonomethyl)glycine), and corn was planted in early May. Corn plots were fertilized with N, P, and K, and alfalfa plots with P and K according to soil test and Ohio Agronomy Guide (Ohio Cooperative Extension Service 1990) recommendations.

All PFBC rate and crop treatment combinations were replicated four times at each location, using a randomized complete block experimental design. Corn and alfalfa were run as separate experiments.

Soil samples were collected from all plots at depths of 0 to 10, 10 to 20, and 20 to 30 cm in April 1993. Samples were air-dried, crushed, and passed through a 2-mm sieve. Soil pH was determined in a 1:2 (soil:water, wt/wt) paste, and water extracts were prepared by shaking a 1:10 (soil:water, wt/wt) mixture for 30 min, followed by filtering though a 0.45-μm membrane filter. Water extracts were analyzed for Al, As, B, Ba, Be, Ca, Cd, Co, Cr, Cu, Fe, Hg, K, Li, Mg, Mn, Mo, Na, Ni, P, Pb, S, Sb, Se, Si, Sr, V, and Zn by inductively coupled plasma emission spectrophotometry.

Alfalfa was harvested in July and September at the Wooster and Coshocton locations, and in July at the Canfield location. Corn grain was harvested at all three locations in October 1993.

Table 1. Soil characterization of 0 to 20 cm depth of Wooster, Coshocton, and Canfield soils.

Soil	Classification	LR[1], Mg/ha	pH	Bray P_1, mg/kg	Exchangeable, mg/kg			CEC[2], $cmol_c$/kg
					K	Ca	Mg	
Wooster	Aquic Fragiudalf	20.6	4.6	9	103	430	85	15.7
Coshocton	Aqultic Hapludalf	13.7	4.8	9	118	300	87	12.1
Canfield	Typic Fragiudalf	2.8	5.8	29	171	1130	29	8.7

[1] Lime requirement, amount of ag-lime required to raise soil pH to 7.
[2] Cation exchange capacity.

Table 2. Characterization of PFBC byproducts.

Parameter		PFBC, cyclone	PFBC, bed
pH (1:1, water)		10.5	12.2
Calcium carbonate equivalent, %		60.3	60.0
Particle size distribution, %			
>2 mm		0.0	4
2-0.1 mm		0.0	95
100-50 μm		23.5	1
50-2 μm		76.1	0
<2 μm		0.4	0
Major minerals, %			
Dolomite		25	ND[1]
Anhydrite		22	36
Periclase		12	27
Total Chemical Analysis:			
Al	%	3.93	2.75
Ca	%	17.53	24.44
Fe	%	5.17	2.08
K	%	0.53	0.14
Mg	%	10.64	16.28
Si	%	7.24	45.25
S	%	5.21	8.64
As	mg/kg	75.0	46.7
B	mg/kg	169.0	206.0
Cd	mg/kg	1.0	3.5
Cr	mg/kg	36.9	17.5
Cu	mg/kg	35.0	912.9
Pb	mg/kg	24.5	28.0
Coal-ash	%	32.1	10.0

[1]Not detected.

Table 3. Amount of PFBC by-product applied to the Wooster, Coshocton, and Canfield soils at each lime requirement rate factor.

Fraction of LR[1]	Amount of PFBC applied to soil, Mg/ha		
	Wooster	Coshocton	Canfield
0.0	0.0	0.0	0.0
0.5	17.2	11.4	2.4
1.0	34.3	22.8	4.7
2.0	68.6	45.6	9.4

[1]Lime requirement, amount of ag lime required to raise soil pH to 7.

Results

Soil Chemistry

Clearly the PFBC material was effective as a liming material. Soil pH was rapidly raised in the Wooster and Coshocton soils, and the increase was sustained through the winter (fig. 1). In the Canfield soil no effect on pH was seen immediately after incorporation, and the following spring there was only a small increase at the 2xLR rate. Apparently the SMP buffer underestimated the lime requirement of this moderately acid soil, and an insufficient amount of PFBC was applied.

The greatest effects on water soluble concentrations of Ca, Mg, and S occurred in the Wooster and Coshocton soils where the 1xLR rate was equivalent to 34.3 and 22.8 Mg/ha, respectively. In the Canfield soil, where the 1xLR rate was only 4.7 Mg/ha, the effects of PFBC application on Ca, Mg, and S were much less.

The major elements in the PFBC are Ca, S, and Mg (table 2), and in each soil application of PFBC increased the water-soluble levels of these elements. Surface soil concentrations of Ca were increased in each of the soils (fig. 2). Only in the Coshocton soil, however, was there significant downward movement of Ca. Much of the Ca in the PFBC is present as $CaSO_4$, which is much more soluble than $CaCO_3$ and thus more likely to be leached downward. The amount of winter precipitation at the two sites was similar, thus the amount of leaching was likely also similar. The differences in movement of Ca between the Wooster and Coshocton soils may be due to the higher cation exchange capacity and exchangeable acidity of the Wooster

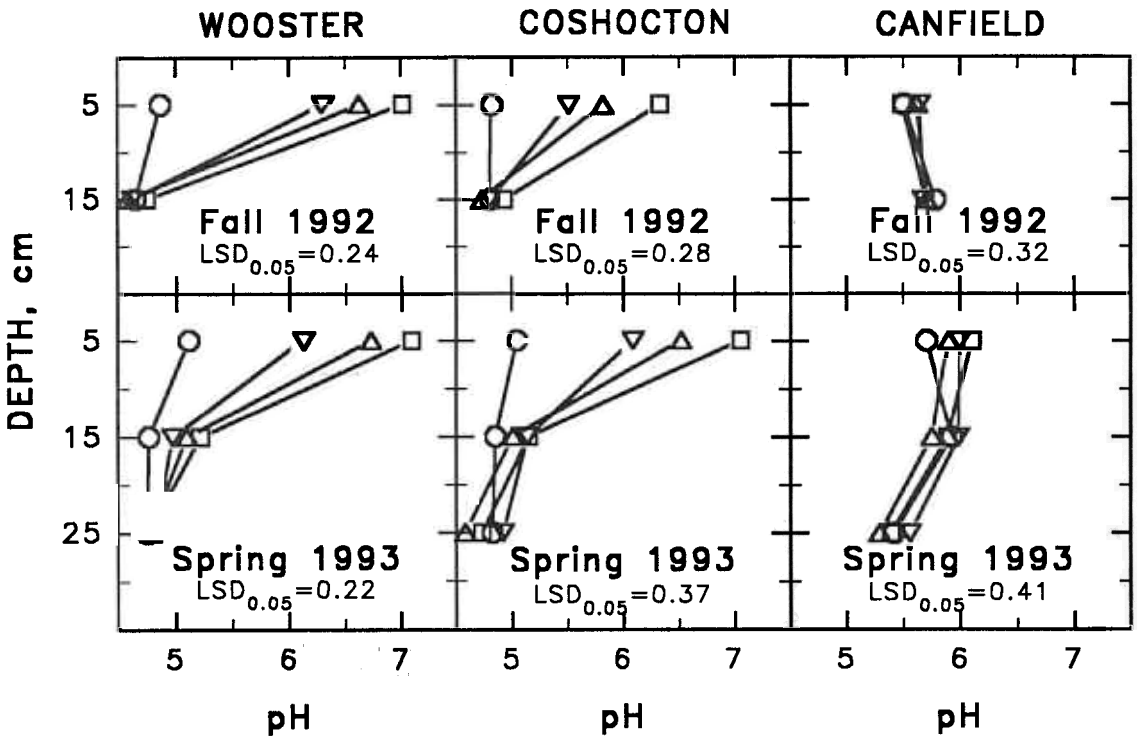

Figure 1. Distribution of pH in Wooster, Coshocton, and Canfield soils in fall and spring following fall application of PFBC (○ = 0xLR, ▽ = 0.5xLR, △ = 1xLR, □ = 2xLR).

soil. Reaction of $CaCO_3$ with exchangeable acidity in the Wooster soil may have led to greater sorption of Ca and, therefore, less movement. The downward movement of Ca during just one winter indicates there is potential for surface application of PFBC to improve the Ca status of soil below the zone of incorporation.

Sulfur is present in PFBC primarily as $CaSO_4$. In column leaching studies with PFBC-amended mine spoil, it was found that greater than 95% of leachate S was present as sulfate (SO_4^{2-}) (Stehouwer et al. 1992). Thus, because of its anionic form, it was expected that S would be highly mobile, as is clearly shown by the elevated S concentrations at all depths in all three soils (fig. 3).

All three soils showed increased water-soluble Mg concentrations at 0-10 cm following PFBC application (fig. 4). Distribution of Mg in the soil profiles was similar at Wooster and Coshocton, with very clear downward movement of Mg below the zone of incorporation. Even though the PFBC contained nearly twice as much Ca as Mg (table 2), movement of Mg on a mass basis was similar to that of Ca in the Coshocton soil, and greater than that of Ca in the Wooster soil. On a mole basis, therefore, movement of Mg was greater than that of Ca. Thus the mobility of the Mg in PFBC appears to be greater than that of the Ca. The same phenomenon was observed in column leaching studies with PFBC, where leachate concentrations of Mg were much higher than those of Ca (Stehouwer et al. 1992). Also, S concentrations in leachates from PFBC-amended spoils were much higher than those from spoil amended with an FGD byproduct containing no Mg. This is due to the much greater solubility of $MgSO_4$ compared with $CaSO_4$ (nearly 300 times more soluble). Movement of Mg relative to Ca was also likely increased by the greater ion selectivity of most soils for Ca relative to Mg (Bohn et al. 1985).

Surface soil concentrations of water-soluble Al, Mn, and Fe, ions which are frequently phytotoxic in acid soils, were decreased by PFBC amendment in all three soils with the exception of Fe in the Coshocton and Canfield soils (figs. 5- 7). This is consistent with the increase in pH that occurred in these soils (fig. 1). The smallest decreases were in the Canfield soil, where the lowest rates were applied, the pH increases were the least, and initial concentrations of Al, Mn, and Fe were the lowest of the three soils. There was no evidence that PFBC decreased concentrations of these elements below the zone of incorporation. On the contrary,

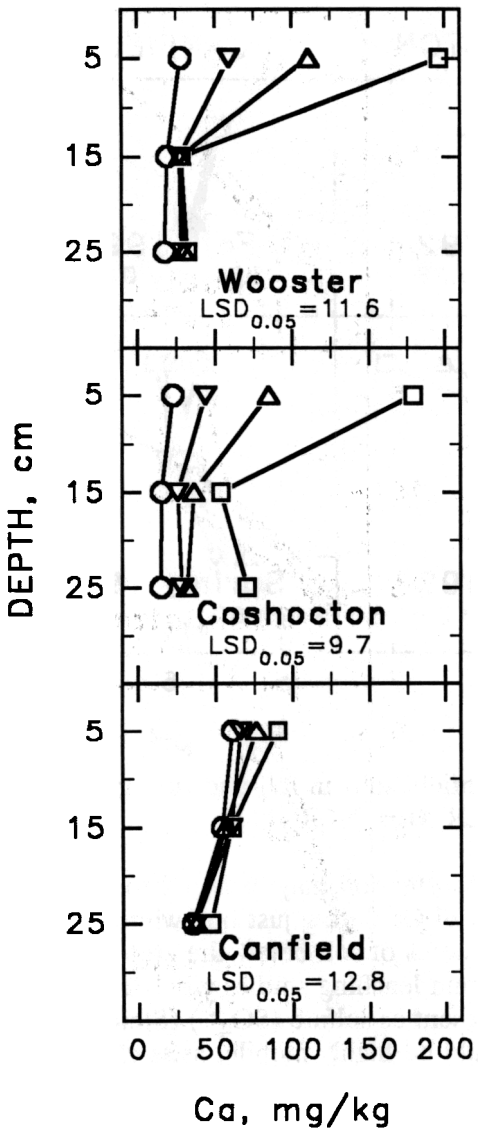

Figure 2. Distribution of water soluble Ca in three soils in the spring following fall application of PFBC (○ = 0xLR, ▽ = 0.5xLR, △ = 1xLR, □ = 2xLR).

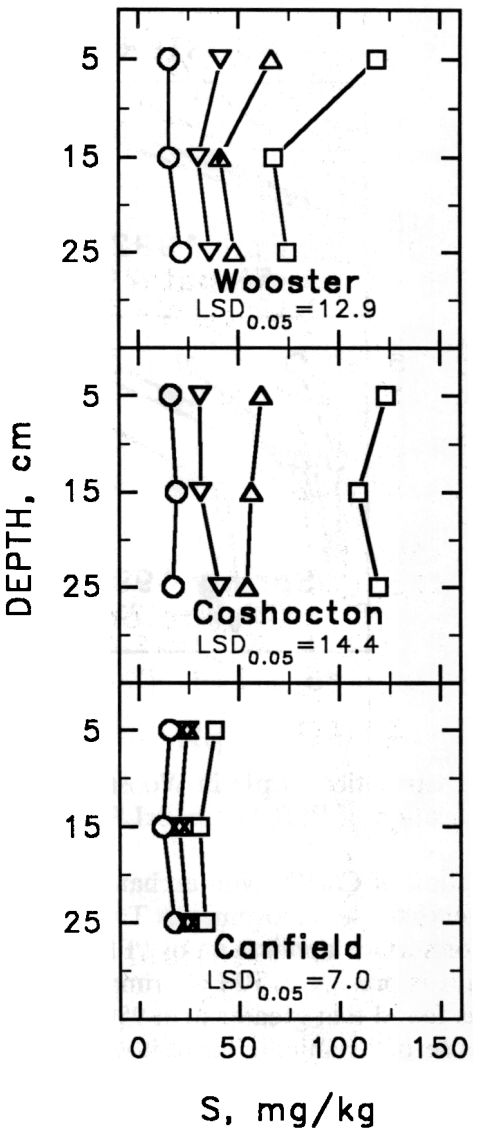

Figure 3. Distribution of water soluble S in three soils in the spring following fall application of PFBC (○ = 0xLR, ▽ = 0.5xLR, △ = 1xLR, □ = 2xLR).

particularly for Mn, and less so for Fe, there appeared to be mobilization from the zone of incorporation to underlying soil. The mobilization of Mn appeared to be the result of cation exchange between Mn^{2+} and Ca^{2+} and Mg^{2+}. Availability of Mn transported deeper in the profile would also be increased because of the lower soil pH below 10 cm. Of the trace elements, only B was affected by PFBC amendment (fig. 8). Concentrations of water-soluble B were increased in the zone of PFBC incorporation in all three soils. There was also evidence of downward movement of B in the Coshocton soil, as was observed with Ca, Mg, S, and Mn. The coal fly ash component of the PFBC material is the primary source for the B. Elevated B, or B phytotoxicity, has been reported in several studies involving land application of coal fly ash (Carlson and Adriano, 1993). It should be noted, however, that the water-soluble B concentrations observed in this study were well below phytotoxic levels; indeed, B toxicity was not observed on alfalfa or corn grown on these three PFBC-amended soils.

Water-soluble concentrations of all other measured elements showed no measurable changes in response

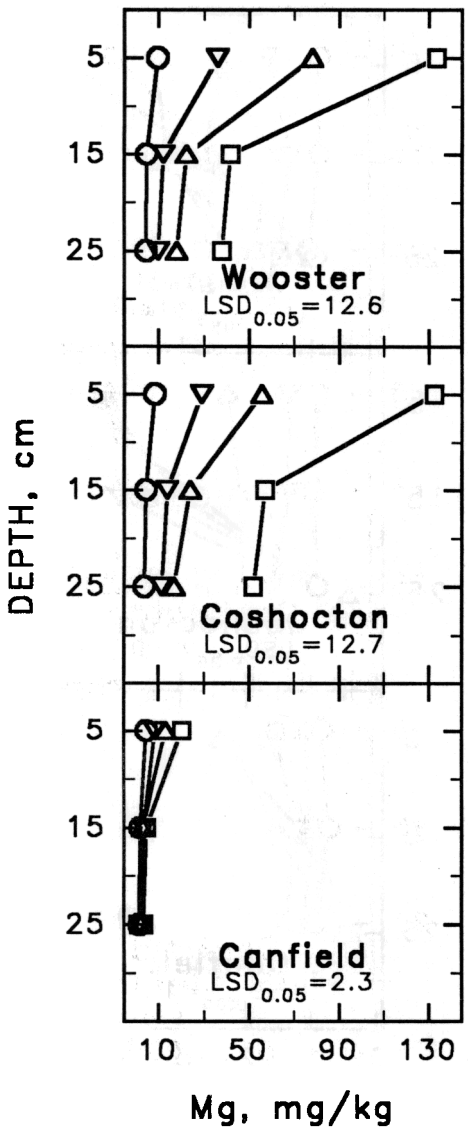

Figure 4. Distribution of water soluble Mg in three soils in the spring following fall application of PFBC (○ = 0xLR, ▽ = 0.5xLR, △ = 1xLR, □ = 2xLR).

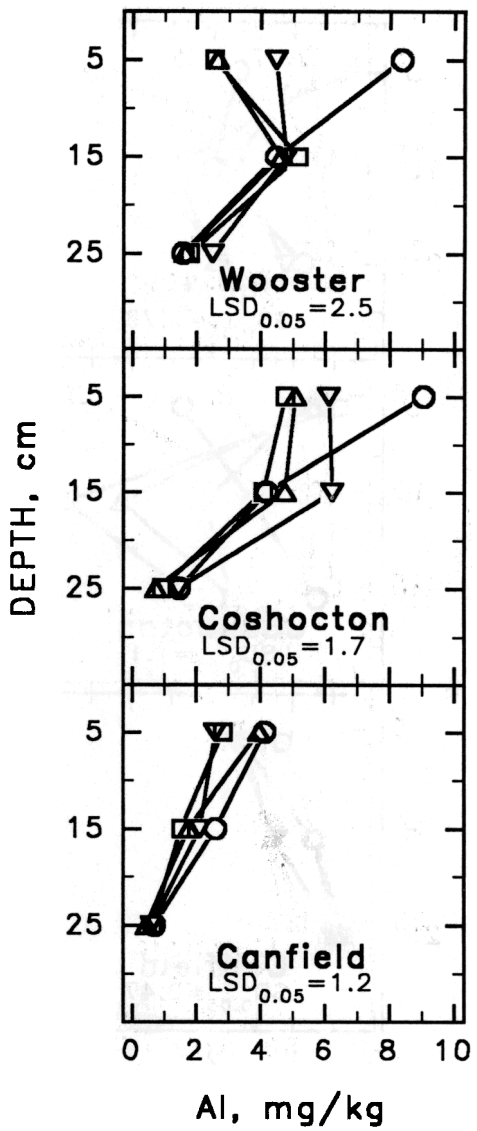

Figure 5. Distribution of water soluble Al in three soils in the spring following fall application of PFBC (○ = 0xLR, ▽ = 0.5xLR, △ = 1xLR, □ = 2xLR).

to PFBC application on any of the three soils (data not shown). Thus there appears to be a very low potential for surface- or ground-water contamination, or increased plant uptake of any of these potentially toxic elements, resulting from application of PFBC at agricultural rates.

Plant Growth

Much of eastern Ohio suffered from a midsummer drought in 1993. No significant precipitation occurred at any of the three sites from July 12 until September 2, with the exception of a 25 mm rain at Canfield on July 29. This drought severely reduced alfalfa and corn growth and yield and overshadowed any PFBC effects.

Corn grain yield was not affected by PFBC in the Wooster or Canfield soils (fig. 9). In the Coshocton

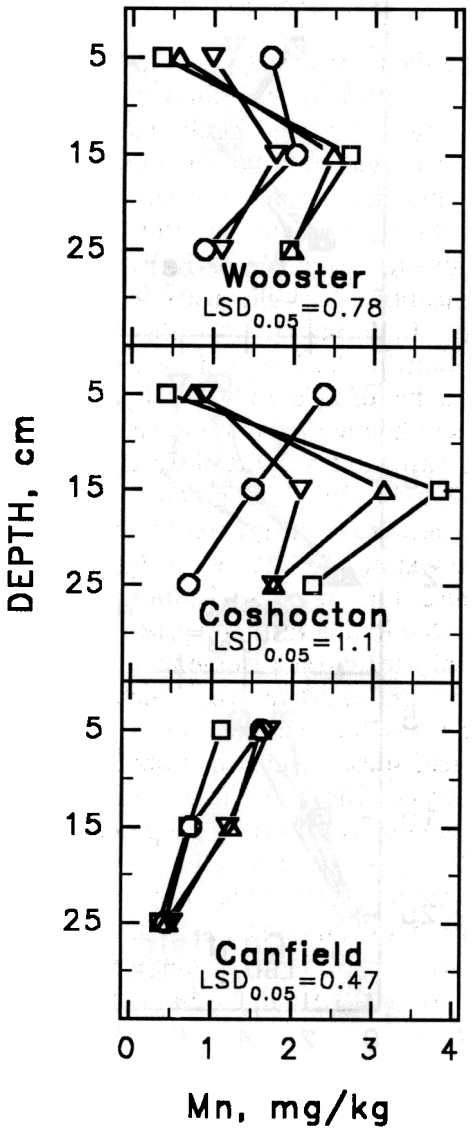

Figure 6. Distribution of water soluble Mn in three soils in the spring following fall application of PFBC (○ = 0xLR, ▽ = 0.5xLR, △ =1xLR, □ = 2xLR).

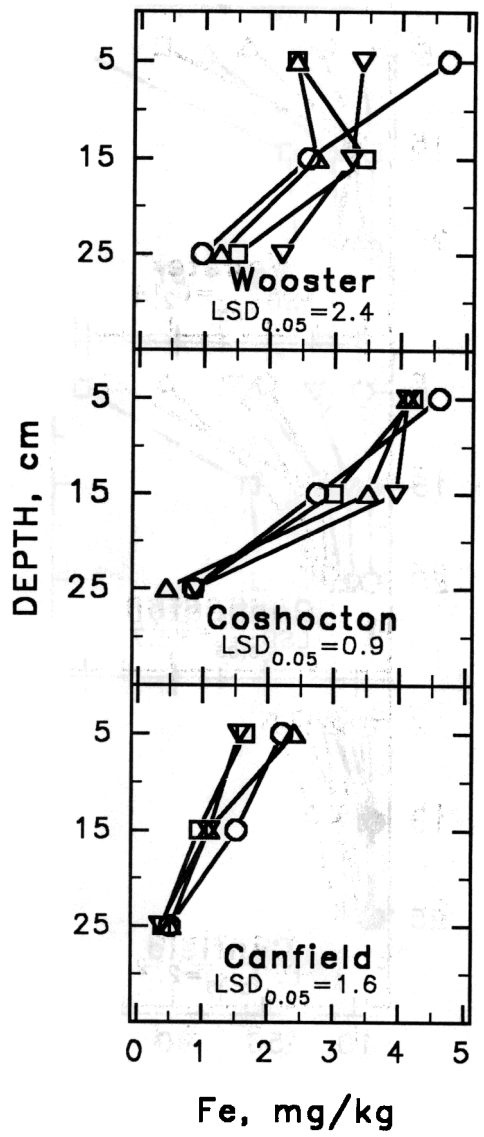

Figure. 7. Distribution of water soluble Fe in three soils in the spring following fall application of PFBC (○ = 0xLR, ▽ = 0.5xLR, △ =1xLR, □ = 2xLR).

soil, PFBC application decreased yield; at the 2xLR rate (45.7 Mg/ha) yield was lower than on the unamended soil. Yield suppression from application of high rates of fluidized bed byproducts has been reported with peanuts (Terman et al. 1979) and red clover (Stout et al. 1979) and has been attributed to high alkalinity and/or high soluble salt content. Given the low soil moisture conditions during the drought period, increased salts in the soil solution may have affected corn growth in our experiment. Good alfalfa growth was obtained only for the first harvest at Wooster, where a large beneficial effect of PFBC application was seen (fig. 9). The second cutting at the Wooster site also showed a positive effect from PFBC application, but yields were severely reduced because of the drought conditions. Poor stand establishment at Coshocton and Canfield, together with drought conditions, resulted in poor yields and no treatment effects on alfalfa yields.

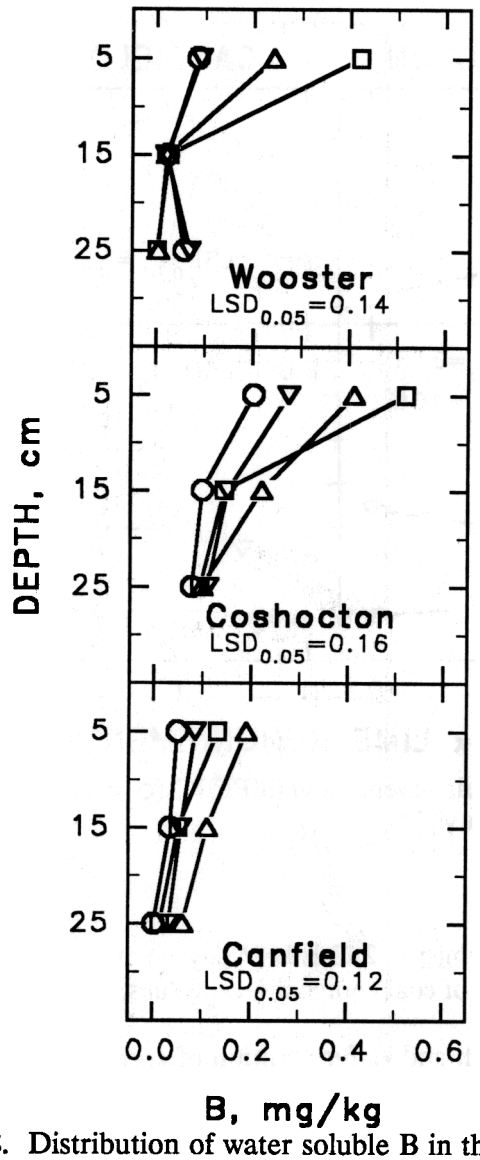

Figure 8. Distribution of water soluble B in three soils in the spring following fall application of PFBC (○ = 0xLR, ▽ = 0.5xLR, △ = 1xLR, □ = 2xLR).

Conclusions

The data presented in this paper represent only one year of data and the following conclusions must, therefore, be considered preliminary. Nevertheless, there is strong evidence that soil application of PFBC at the recommended liming rate effectively neutralized acidity in the zone of incorporation and showed some evidence of increasing pH in underlying soil 6 months after application. Soil chemistry was also made more favorable for plant growth. Water-soluble concentrations of Al, Mn, and Fe were decreased, and the base status of acid soils was improved. The base status of soil below the zone of incorporation was also improved owing to transport of Ca and Mg. Mobility of Ca and Mg in PFBC is greater than in dolomitic limestone because of the greater solubility of $CaSO_4$ and $MgSO_4$ compared with that of $CaMg(CO_3)_2$. There was no evidence that land application of PFBC at the recommended liming rate would lead to elevated levels of potentially toxic trace elements in soil or water.

With pH-sensitive crops such as alfalfa, PFBC application has the potential to improve growth and yield on acid soils. Even when applied at 2xLR there was no adverse effect on yield. With a crop such as corn that is less pH sensitive, the potential yield benefit from PFBC application may be less. While no yield improvement was seen in this study, we believe this was due to drought conditions. In a more normal rainfall growing season we believe there would be greater potential for corn yield to be improved by PFBC application.

Acknowledgements

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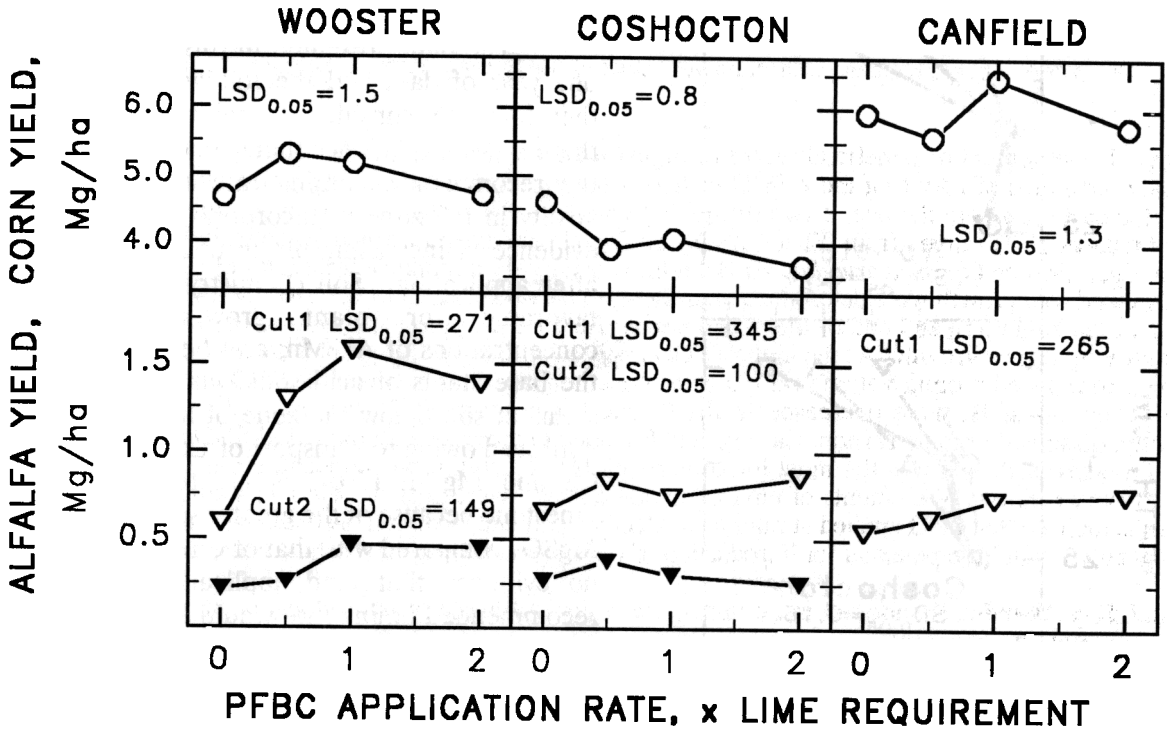

Figure 9. Alfalfa forage yield and corn grain yield on three soils amended with PFBC (○ = corn grain yield, ∇ = first cut alfalfa yield, ▼ = second cut alfalfa yield).

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DRY FLUE GAS DESULFURIZATION BYPRODUCTS AS AMENDMENTS FOR RECLAMATION OF ACID MINE SPOIL[1]

Warren A. Dick[2], Richard C. Stehouwer[2], Joel H. Beeghly[3], Jerry M. Bigham[4], and Rattan Lal[4]

Abstract: Development of beneficial reuses of highly alkaline, dry flue gas desulfurization (FGD) byproducts can impact the economics of adopting these FGD technologies for retrofit on existing powerplants. Greenhouse studies were conducted to evaluate the use of two dry FGD byproducts for reclamation of acid mine spoil (pH, 3.1 to 5.8). Treatment rates of FGD ranged from 0% to 32% by dry weight and most treatments also included 6% by dry weight of sewage sludge. Fescue (*Festuca arundinacea* Schreb.) was harvested monthly for a total of six harvests. Plant tissue composition and root growth were determined after the sixth harvest. Leachate analyses and pH determination of mixes were done at the beginning and end of the experiments. Both FGD byproducts were effective in raising the spoil pH and in improving fescue growth. At the highest FGD application rate, fescue growth decreased from the optimum due to high pH and reduced rooting volume caused by cementation reactions between the FGD and spoil. Trace elements, with the exception of B, were decreased in the fescue tissue when FGD was applied. Leachate pH, electrical conductivity, dissolved organic carbon, Ca, Mg, and S tended to increase with increased FGD application rate; Al, Fe, Mn, and Zn decreased. pH was the most important variable controlling the concentrations of these elements in the leachate. Concentrations of elements of environmental concern were near or below drinking water standard levels. These results indicate that FGD applied at rates equivalent to spoil neutralization needs can aid in the revegetation of acid spoil revegetation with little potential for introduction of toxic elements into the leachate water or into the food chain.

Additional Key Words: abandoned mine lands, clean coal technology byproduct, plant growth response, water quality, heavy metals, oxyanions

Introduction

The U.S. Department of Energy's Clean Coal Technology program and the 1990 amendments to the Clean Air Act, mandating a two-stage 10-million short ton reduction in SO_2 emissions in the United States, have encouraged the development of various SO_2 scrubbing technologies. A number of dry FGD technologies have been developed in the past decade. These technologies are generally smaller in scale and require a lower capital investment than do the wet FGD processes. Dry FGD technologies are also generally designed for retrofit on existing coal-fired powerplants and, therefore, represent an option for bringing older plants into compliance with clean air legislation.

A recent ruling by the U.S. Environmental Protection Agency, exempting most FGD byproducts from the label of being hazardous wastes, provides incentive to develop beneficial uses of these materials. Thus, plants burning high-sulfur coal and using dry FGD technologies may avoid the increasingly expensive landfill disposal costs and at the same time create a product that can be used for beneficial purposes.

We have recently completed a comprehensive study of the chemical, physical, mineralogical, and engineering properties of 58 dry FGD byproduct samples from Ohio (Ohio State University 1993). Based upon this information, a greenhouse study was conducted to demonstrate potential reuses of dry FGD byproducts for reclaiming acid mine spoil.

Dry FGD byproducts are composed of a mixture of conventional coal combustion ash (either bed or fly ash), the SO_2 reaction product (primarily anhydrite, $CaSO_4$), and unspent sorbent (generally lime, limestone, or dolomite). Because of the presence of unspent sorbent, dry FGD byproducts are usually highly alkaline with significant neutralization potential (Carlson and Adriano 1993, Fowler et al. 1992, Terman et al. 1978). Anhydrite does not

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[2]Warren A. Dick, Professor of Agronomy and Richard C. Stehouwer, Senior Researcher in Agronomy, The Ohio State University, Wooster, OH, USA.

[3]Joel H. Beeghly, Project Manager, Dravo Lime Company Research Center, Pittsburgh, PA, USA.

[4]Jerry M. Bigham, Professor of Agronomy and Rattan Lal, Professor of Agronomy, The Ohio State University, Columbus, OH, USA.

neutralize acidity but may ameliorate problems of Al toxicity through the formation of $Al(SO_4)_X$ complexes. It is also a source of both Ca and S for plant nutrition. Fly ash may supply other plant nutrients (e.g., B, Mo, Zn, P, K). These FGD byproducts may be high in soluble salts and contain some trace elements of environmental concern.

Of the several dry FGD technologies tested in Ohio, byproduct from two of these technologies - from lime injection multistage burner (LIMB) and from pressurized fluidized bed combustion (PFBC) - were chosen for use in our greenhouse studies. In the LIMB process, calcium-based sorbent is injected directly into the boiler where it calcines to CaO and reacts with SO_2 and O_2 in the combustion gases to produce $CaSO_4$. The reaction product and unspent sorbent are collected with the fly ash. In PFBC systems, a calcium-based sorbent (usually limestone or dolomite) and crushed coal are introduced together into the boiler bed, where they are "fluidized" or suspended by jets of air. This mixes the coal and sorbent and allows for reaction of SO_2 and sorbent. Two byproduct streams are created. One is the heavier, granular bed ash material and the other is the finer materials removed by the particulate emission control equipment.

Several studies have been conducted to investigate land application of FGD byproducts. Generally these studies have focused on using atmospheric fluidized bed combustion (AFBC) byproducts applied to soils at rates equivalent to agricultural limestone needs (Carlson and Adriano 1993). Byproduct from AFBC, when used as a limestone substitute, was an effective source of Ca and S. However, when it was applied at rates well in excess of the soil's lime requirement, high alkalinity and salinity adversely affected plant growth. Korcak (1988) applied high rates (112 mt/ha) of AFBC as a mulch between apple trees and yield increased in three of four tree types and decreased in one. No nutritional disorders were found in any of the tree types.

A much higher volume potential beneficial use for dry FGD byproducts is the reclamation of active and abandoned surface coal mines with acid spoils. Drainage water from such sites can cause severe offsite environmental damage owing to its acidity, high salt content, and concentrations of soluble metals (Sutton and Dick 1987). Reclaiming spoils, by addition of alkaline amendments, must neutralization the acidity but not introduce other factors that may have adverse environmental impacts. Because of the high potential for acid development of many acid minespoils, however, alkaline application rates must be much higher than for acid agricultural soils.

The objectives of our greenhouse study were to investigate the use of two different FGD byproducts as an amendment material for reclamation of acid mine spoil. The effectiveness of this reclamation was determined by measuring plant growth responses and leachate water quality.

Materials and Methods

Experiment 1

Spoil was obtained from the Fleming abandoned mined land (AML) site near New Philadelphia in eastern Ohio. This site contains approximately 10 ha (25 acres) of exposed, highly erodible underclay bordered on two sides by approximately 18 ha (45 acres) of unreclaimed overburden spoil and 2 ha (5 acres) of coal refuse. Overburden refers to material removed to expose the coal seam mined at the AML site and underclay refers to the material beneath the coal seam. The term "spoil" will be used to refer to both the overburden and the underclay materials. These materials were air-dried and passed through a 10-mm sieve prior to use in the greenhouse experiments. Characteristics of the spoil materials from the Fleming AML site are given in Table 1.

Dry FGD byproducts from the LIMB and PFBC scrubber technologies were used. The LIMB byproduct was collected from the full-scale commercial LIMB demonstration at Ohio Edison's Edgewater plant in Lorain, OH. The PFBC byproduct was obtained from the demonstration facility operated by American Electric Power at the Tidd plant located near Brilliant, OH. Digested sewage sludge from Rahway, NJ was used as an amendment for the AML spoils and was air-dried and passed through a 10-mm screen before use. Detailed characterization data for the LIMB and PFBC byproducts and for the sewage sludge are summarized in table 2.

Spoil materials were combined with the FGD and sewage sludge amendments to simulate their placement during AML reclamation. Dry FGD byproducts were applied at rates of 0%, 3%, 6%, 12%, and 24% by dry weight to overburden (3 kg) and underclay (4 kg) materials from the AML site. Sewage sludge was added to each of these treatments at a constant rate of 6% by dry weight. Supplemental P (0.64 g P_2O_5) and K (0.25 g K_2O) fertilizers were mixed with each column containing sewage sludge. In addition to the treatments with sewage sludge, spoil material without sewage sludge was combined with the two dry FGD byproducts at rates of 0% and 12% by dry weight. Nitrogen (0.15 g N), P (0.75 g P_2O_5), and K (0.74 g K_2O) fertilizers were mixed with the no-sewage sludge treatments. All materials were mixed when in an air-dry condition and were poured into 30 cm tall polyvinyl chloride (PVC) columns (15 cm diameter).

Table 1. Characterization of coal mine spoil materials and topsoil used in greenhouse column studies.

Parameter	AML spoil	AML underclay	Active mine spoil	Active mine topsoil
Particle size, %:				
Sand (0.05 to 2 mm)	17.3	5.0	14.4	27.0
Silt (2 to 50 μm)	34.7	61.0	54.9	59.3
Clay:				
(0.2-2 to μm)	40.8	28.9	26.0	10.6
(< 0.2 μm)	7.2	5.2	4.7	3.1
Extractable cations, mg/kg:				
Ca	50	130	1,180	260
Mg	27	38	370	60
K	65	76	181	63
pH (1:1, water)	3.1	3.4	5.8	4.4
Total chemical analysis:				
Al %	8.7	9.5	9.2	NA
Ba %	0.05	0.03	0.04	NA
Ca %	<0.01	<0.01	<0.01	NA
Fe %	2.5	<0.01	1.3	NA
K %	2.0	1.3	2.3	NA
Mg %	<0.01	<0.01	0.18	NA
Organic C %	11.1	0.7	1.7	0.8
S %	1.02	0.27	1.06	NA
Si %	17.6	27.2	23.5	NA
As mg/kg	46.3	14.4	13.7	NA
Cd mg/kg	0.8	0.8	0.8	NA
Cr mg/kg	94.4	95.6	79.8	NA
Cu mg/kg	26.8	37.3	79.0	NA
Pb mg/kg	78.0	35.0	30.5	NA
Mo mg/kg	14.0	1.0	<0.9	NA
Ni mg/kg	28.5	33.2	65.1	NA
P mg/kg	1,027	351	371	NA
Se mg/kg	4.5	5.6	4.3	NA
Zn mg/kg	<0.03	<0.03	<0.03	NA

NA, not analyzed.

Columns were wetted by adding enough deionized water to produce 150 to 200 mL of leachate, and were then planted with 30 seeds of Kentucky 31 tall fescue (*Festuca arundinacea* Schreb.). After the initial wetting, columns were weighed to determine gravimetric moisture content, which was assumed to represent field capacity. Columns were watered with sufficient deionized water to replace evapotranspiration losses, and gravimetric soil moisture was adjusted to approximately 75% of field capacity by weighing the columns and adding needed water on a weekly basis. Artificial lighting was used to provide a minimum of 14 hours of light per day (400 μmol/m/s) and temperature was maintained at 25°C during the day and at 20°C during the night. Following an initial 3-month growth period, fescue was harvested once each month for a total of six harvests. After each harvest, each column with no sewage sludge received a surface application of N (0.18 g N), P (0.46 g P_2O_5), and K (0.23 g K_2O) to replace nutrients removed by the plant.

Pots were leached a second time following the third harvest and again following the sixth harvest using the same procedure as previously described. Leachates were analyzed for pH, for electrical conductivity, for As, B, Ba, Be, Ca, Cd, Co, Cr, Cu, Fe, Hg, K, Li, Mg, Mn, Mo, Na, Ni, P, Pb, S, Sb, Se, and Zn by inductively coupled plasma emission spectrometry, for SO_4^{2-} by ion chromatography, and for dissolved organic C using a carbon analyzer.

Table 2. Characterization of LIMB, PFBC, and sewage sludge amendments.

Parameter	LIMB	PFBC	Sewage sludge
Particle size, %:			
Sand (0.05 to 2 mm) --	0	25.5	NA
Silt (2 to 50 μm) -------	90	74.1	NA
Clay (<2 μm) ----------	10	0.4	NA
Mineralogy, %:			
Anhydrite ($CaSO_4$) ----	25	22	NA
Calcite ($CaCO_3$) --------	15	11	NA
Dolomite ($CaMg(CO_3)_2$)	ND	23	NA
Lime (CaO) ----------	21	ND	NA
Portlandite ($Ca(OH)_2$)-	5	ND	NA
Periclase (MgO) --------	ND	13	NA
Flyash ----------	30	32	NA
$CaCO_3$ equivalent, % ---	59.4	60.3	0
pH (1:1,water) ----------	12.5	10.5	6.5
Total chemical analysis:			
Al -------- % ----------	3.52	3.93	3.40
Ba --------% ----------	.03	.02	.01
Ca --------% ----------	36.0	17.5	2.76
Fe -------- % ----------	5.56	5.17	1.24
K ---------% ----------	.91	.50	.15
Organic C-% ----------	NA	NA	31.2
Mg -------- % ----------	.60	10.6	.34
P --------- % ----------	0.02	0.02	1.78
S --------- % ----------	5.77	5.21	1.41
Si -------- % ----------	6.58	7.24	NA
As -------mg/kg --------	55.1	1.9	< .03
B --------mg/kg --------	233	171	31.1
Cd ------ mg/kg --------	1.0	1.9	6.3
Cr -------mg/kg --------	28.0	36.9	315
Cu ------ mg/kg --------	21.0	52.5	1,174
Pb ------ mg/kg --------	16.0	16.0	16.1
Mo ------mg/kg --------	5.9	6.6	11.2
Ni -------mg/kg --------	31.1	52.5	166
Se -------mg/kg --------	8.1	5.6	< .03
Zn ------ mg/kg --------	86.0	74.0	1,494

NA, not analyzed. ND, not determined.

Harvested plant tissue was dried at 60°C for 48 h, weighed, and digested by heating in concentrated HNO_3 and $HClO_4$. Digests were analyzed for As, B, Ba, Be, Ca, Cd, Co, Cr, Cu, Fe, Hg, K, Li, Mg, Mn, Mo, Na, Ni, P, Pb, S, Sb, Se, and Zn by inductively coupled plasma emission spectrometry. Separate samples were analyzed for total N using a nitrogen analyzer. Following the final harvest, spoil in the columns was sampled at various depths and analyzed for root density. Also following the final harvest and leachate collection, the columns were sampled at various depth intervals for pH determination, and selected samples were analyzed for crystalline mineral phases by x-ray diffraction.

Treatments were replicated four times and, excluding the no-sludge treatment, the experimental design was a complete factorial with randomized complete blocks. Data analysis was conducted using analysis of variance, and regression analysis was used to assess the responses to amendment rate within spoils and FGD byproducts. A separate sludge versus no sludge response comparison was also made.

Experiment 2

Overburden and topsoil were collected from an active surface coal mine site located in east-central Ohio. Topsoil refers to the upper 20 cm of the soil profile which was removed separately from the overburden at the active mine site. Characteristics of the overburden and topsoil from the active mine site are given in table 1. Also used were dry FGD byproducts from the LIMB and PFBC scrubber technologies (table 2)

Overburden (6.2 kg) from the active mine site was mixed with the dry FGD byproducts at rates of 0%, 4%, 8%, 16%, and 32% by weight and poured into 60-cm PVC columns (15-cm diameter) mounted as previously described. Topsoil (5.4 kg) from the active mine site was mixed with LIMB (0.4%) and PFBC (0.8%) and placed in the columns above overburden treated with the same FGD by-product. This resulted in a 20-cm layer of FGD-amended topsoil in each column above the FGD-amended overburden. Rates of FGD amendment to the topsoil were based on results from a prior incubation study with LIMB and PFBC which indicated these rates would raise the topsoil pH to 7. Topsoil in each column was fertilized with N (0.27 g N), P (1.23 g P_2O_5), and K (0.54 g K_2O). These columns also received surface applications of N (0.18 g N), P (0.46 g P_2O_5), and K (0.23 g K_2O) after each harvest to replace nutrients removed by the harvested plants.

All other procedures were the same as described previously for Experiment 1 with one exception. After the initial leaching at time of planting, only a single additional leaching was conducted following the sixth harvest at the end of the growth experiment.

Results and Discussion

Experiment 1

Plant Growth Response. No fescue could be grown on unamended AML overburden. Fescue seeds planted in unamended AML overburden did germinate, but seedlings died within 2 to 3 days of emergence. Addition of either sewage sludge alone (see zero rate of FGD in fig. 1) or FGD alone (data not shown) permitted fescue growth on the otherwise phytotoxic AML overburden. However, applying both sludge and FGD byproduct together significantly improved growth compared to when FGD was applied alone. Growth response to PFBC was similar at all amendment rates, while with LIMB there was growth suppression at the highest rate. This growth suppression was greatest at the beginning of the experiment and had disappeared by the end of the experiment. It was associated with the initially excessively high overburden pH of 9.6, which decreased with time to pH 7.8.

On the underclay material, all LIMB amendment rates initially suppressed growth of fescue (fig. 1). By the end of the experiment, however, the lower LIMB amendment rates increased fescue yield, while the yield suppression persisted at the 24% LIMB rate. A similar response was seen with PFBC amendment in that initially negative effects on

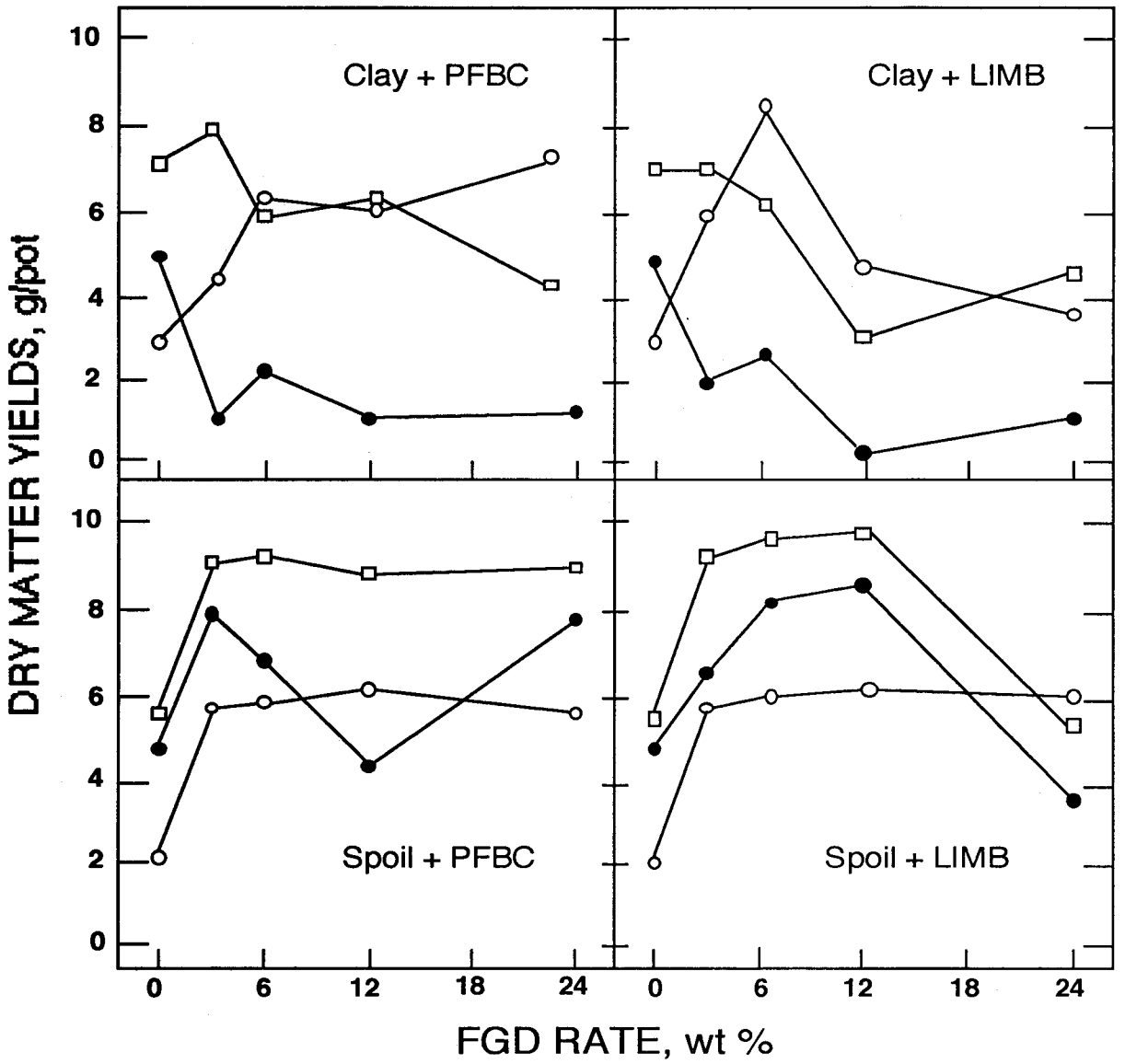

Figure 1. Fescue plant yields (topgrowth only) from first (●), third (□) and sixth (○) harvests on underclay and overburden from the Fleming AML site that was amended with PFBC or LIMB byproduct.

yield changed to positive effects in the last two harvests. Similar growth suppression of peanuts with 10% to 25% application rates of AFBC was attributed to high pH and salinity (Terman et al. 1978).

Root growth was also affected by FGD amendments (data not shown). In the highly toxic AML overburden, rooting density at the 10- to 15-cm depth was increased by FGD amendment up to the 12% rate. With the 24% FGD amendment rate, however, root growth was concentrated in the upper 10 cm of the columns and was inhibited below this depth by the combination of cementing and high pH. In the less toxic underclay, LIMB amendment above the 3% rate decreased root growth. With 24% LIMB amendment, a thick mat of roots grew in the upper 10 cm. Below this depth no root growth was possible because the formation of the secondary mineral ettringite ($Ca_6Al_2(SO_4)_3(OH)_{12}\cdot 26H_2O$) cemented the underclay into a solid, highly impermeable mass. With 24% PFBC amendment, there was less extensive cementing, primarily caused by gypsum and not ettringite, and thus root growth was not completely inhibited.

Tissue composition of fescue grown on AML spoils was affected by FGD and sewage sludge amendment (data not shown). Concentrations of most elements remained within sufficiency levels and below toxic levels and the fescue plants did not show any toxicity symptoms. Differences in the Ca and Mg content of the FGD byproducts were reflected in tissue concentrations, with LIMB amendment producing higher Ca, and PFBC producing higher Mg tissue concentrations. Magnesium toxicity has been reported in plants grown on high sulfate soils amended with dolomite (Evangelou and Thom 1984). However, in spite of leachate Mg concentrations as high as 7,000 mg/L, tissue Mg remained within a normal range and there were no signs of Mg or salt toxicity. Tissue S concentrations were largely unaffected by the large increases in spoil S due to FGD amendment. There was a tendency for the PFBC amendment to produce higher tissue S concentrations which reflected the much higher SO_4^{2-} solubility in the PFBC material. Tissue Al and Mn concentrations were decreased from near phytotoxic levels to normal levels by FGD amendment of the AML spoils. Similar responses were seen with Cu, Ni, and Zn. Each of these responses was associated with decreases in leachate concentrations of the same elements. The trace elements As, Cd, Cr, and Se, which are of concern because of their potential toxicity to higher animals, all showed decreasing tissue concentrations with FGD amendment (data not shown), while no changes in tissue Hg or Pb concentrations were observed. One exception was an increase in tissue Se at intermediate LIMB amendment rates. Total Se in the LIMB byproduct was higher than in the PFBC byproduct and may account for some of this effect. Tissue B concentrations were unaffected by PFBC, but increased with increasing LIMB amendment rate to levels in excess of 100 mg/kg. Boron toxicity symptoms were not observed in the fescue plants; however, more susceptible plant species would likely be adversely affected by these B concentrations.

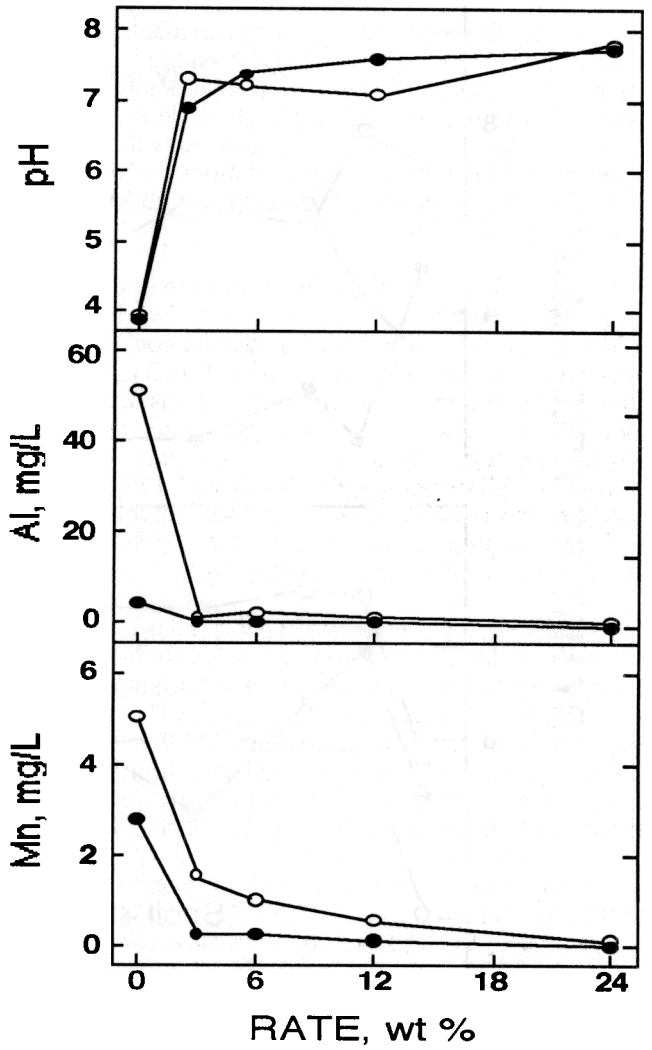

Figure 2. pH, aluminum, and manganese leachate concentrations in AML overburden treated with PFBC byproduct. The first leaching (o) was conducted after seeding of the columns and the last leaching (●) was after 9 months of growth.

Spoil and Leachate Chemistry. The FGD byproducts were very effective in rapidly raising the pH of each of the mine spoil materials (data for PFBC byproduct only is shown, fig 2). Because of its portlandite and lime content the LIMB byproduct raised both spoil and leachate pH to higher levels than the PFBC byproduct, in which alkalinity was due primarily to Ca and Mg carbonates with some periclase (table 2).

At application rates of 4% to 6% these FGD byproducts, with a neutralization potential of about 60%

$CaCO_3$ equivalency, increased minespoil pH to around 7 and sustained this pH over time. Application of alkaline materials in excess of that needed to neutralize acidity in the treated zone may be desirable if downward leaching will increase the pH and base status of spoil below the treated zone. It should be noted, however, that dry FGD byproducts vary considerably in their total neutralization potential (Ohio State University 1993). Therefore, the amount of material needed to neutralize acidity in the treated spoil is also expected to vary from one type of FGD byproduct to another.

The pH of the treated spoil materials in the columns decreased with time, (data not shown) with the largest decreases occurring near column surfaces. The decrease was apparently the result of carbonation of lime, portlandite, and periclase since the pH was tending toward that of free carbonates in equilibrium with atmospheric CO_2.

Concentrations of dissolved organic C (DOC) in the leachates increased with increasing LIMB and PFBC amendment in all AML spoil materials (data not shown). Leachate DOC levels were also increased by 6% sewage sludge amendment. As was seen with pH, LIMB amendment produced higher leachate DOC concentrations than PFBC amendment, and AML underclay gave higher DOC than AML overburden. In fact, the increases in DOC were strongly correlated with increases in pH above 7 (r^2=0.82). These results are expected since it is well established that the solubility of organic matter increases substantially as pH increases above the neutral range (Stevenson 1982).

Leachate concentrations of the elements Al and Mn (fig. 2), and of Mn and Zn, which are frequently toxic in acid spoils, all decreased with FGD amendment. This was expected since the solubility of each of these elements is greatest under acid conditions and decreases rapidly with increasing pH (Bohn et al. 1985). Amendment of the AML overburden with sewage sludge gave large decreases in leachate Al and Fe, even though the corresponding pH increase was small (from 2.98 to 4.05). This reflects the ability of organic C to form strong complexes with these elements (Stevenson 1982), and the inverse relationship between soil organic matter and Al^{3+} solubility and toxicity (Hue et al. 1986). The same effect was not observed with Mn and Zn because concentrations of these metals were relatively high in the sewage sludge (table 2).

Increasing FGD amendment increased leachate soluble salt concentrations with the largest increases resulting from PFBC amendment (data not shown). Much of the salt was in the form of sulfate as this species comprised more that 90% of the total S leached. Leachate concentrations of Ca and S in LIMB-amended minespoils appeared to be controlled by the solubility of gypsum ($CaSO_4 \cdot 2H_2O$) where ettringite did not form. Because ettringite is much less soluble than gypsum, Ca and S concentrations decreased in leachates where ettringite was present even though the total amount of Ca and S in the columns increased. With PFBC amendment, leachate Ca appeared to be controlled by the solubility of calcite, while Mg and S were likely controlled by the solubility of epsomite ($MgSO_4 \cdot 6H_2O$). Leachate S and Mg could go to much higher levels with PFBC amendment than leachate S and Ca with LIMB amendment because epsomite is approximately 300 times more soluble than gypsum. Thus the presence of Mg in the PFBC byproduct gives it a greater potential for excessive salt loading than the LIMB byproduct. The decrease in leachate S and Mg at the 32% PFBC amendment rates was apparently caused by gypsum cementing in the overburden column.

Tables 3 gives leachate concentrations of elements that are of environmental concern and have been regulated with respect to land application of sewage sludge (Environmental Protection Agency 1993). Not listed are Hg and Pb. Mercury was below detection limit (<0.04 mg/L) in all of the column leachates and was also not detected (<0.0002 mg/ L) in Toxicity Characteristic Leaching Procedure (Environmental Protection Agency 1991) extracts of LIMB or PFBC byproduct. Lead was detected at a level of 0.12 mg/L only in leachates from unamended active overburden. In all other leachates, Pb was below detection limit (0.04 mg/L). Leachate Cd showed either no effect (AML overburden and clay) or a decrease (active mine overburden) in concentration with increasing FGD amendment. Thus with respect to these metals there appears to be little potential for environmental contamination as a result of using FGD byproducts.

Leachate concentrations of the oxyanions As, B, and Se tended to increase with increasing FGD amendment, with the highest concentrations occurring in leachates from active mine overburden. Boron is of concern because of the potential for the development of soil solution concentrations which are phytotoxic. Although there was some increase in leachate B with PFBC amendment, much larger increases were observed with LIMB amendment up to the 16% rate. These increases were seen in both the AML and the active mine overburden. This is not surprising since total B was much higher in the LIMB byproduct than in the PFBC byproduct. The most phytotoxic B species, however, are relatively water soluble (Woodbury 1992). In an actual mine reclamation site, leaching due to natural precipitation may rapidly move soluble B below the root zone, thereby decreasing the potential for B phytotoxicity.

Experiment 2

Plant Growth Response. With the active mine spoil, FGD byproduct treatments were applied to overburden material which was then covered by a 20-cm layer of topsoil amended with either LIMB (0.4%) or PFBC (0.8%).

Table 3. Trace element composition of final leachates from columns of AML overburden and underclay amended with sewage sludge and LIMB and PFBC byproducts.

Treatment	FGD amendment rate, wt %	Composition, mg/L								
		As	B	Cd	Cr	Cu	Mo	Ni	Se	Zn
		OVERBURDEN WITH 6% SEWAGE SLUDGE								
LIMB	0	<0.05	0.41	<0.003	0.009	0.04	<0.018	0.24	<0.27	3.34
	3	< .05	.84	.003	.005	.02	< .018	.01	< .27	.10
	6	< .05	1.02	< .003	.004	.01	< .018	< .01	< .27	.03
	12	< .05	3.72	< .003	.008	.03	< .018	.01	< .27	.05
	24	.14	.61	< .003	.146	5.55	3.417	.85	.28	.07
PFBC	0	< .05	.41	< .003	.009	.04	< .018	.24	< .27	3.34
	3	< .05	.55	< .003	.014	.16	< .018	.03	< .27	.20
	6	< .05	.57	< .003	.011	.08	< .018	.02	< .27	.16
	12	< .05	.93	< .003	.021	.06	.089	.01	< .27	.06
	24	.07	.97	.004	.043	.21	.394	.11	.54	.12
		UNDERCLAY WITH 6% SEWAGE SLUDGE								
LIMB	0	<0.05	0.53	<0.003	0.010	0.25	<0.018	0.96	<0.27	4.30
	3	< .05	1.42	< .003	.009	.06	.045	.01	< .27	.17
	6	.06	2.68	.017	.005	.19	.051	.05	< .27	.14
	12	.22	2.64	< .003	.069	11.08	1.941	1.04	.38	.08
	24	.06	.62	< .003	.022	26.07	1.215	3.96	.27	.06
PFBC	0	< .05	.53	< .003	.010	.25	< .018	.96	< .27	4.30
	3	.05	.74	.006	.016	.13	.120	.07	< .27	.16
	6	.05	1.04	< .003	.025	.15	.344	.06	< .27	.20
	12	.06	1.60	< .003	.016	.19	.786	.06	.32	.09
	24	.22	1.30	< .003	.041	.46	1.086	.14	.62	.15

There were relatively small differences in growth response among treatments (fig. 3) suggesting the topsoil layer had a larger effect on fescue growth than did treating the underlying overburden.

The experiment involving treatment of spoil from the active mine site (data not shown) revealed that root growth was most extensive in the topsoil layer and was not affected by the FGD treatments in the underlying spoil. Roots were able to grow into the unamended overburden and root mass density at 20 to 25 cm was not affected by PFBC amendment. With 32% PFBC amendment, root growth below 25 cm was severely restricted due to cementation and high salt concentrations. With LIMB amendment, root mass density at 20 to 25 cm was increased up to the 8% rate and declined at higher rates due to cementation and high pH. Cementing was most extensive with 32% LIMB where it limited root growth to the upper 1 cm of the overburden layer. While there was only a slight yield depression due to these root restricting layers in this experiment, changes in rooting volume could have a much greater impact in the field where the decreased rooting volume would increase drought susceptibility.

Spoil and Leachate Chemistry. Selenium concentrations were much higher in the active mine than in the AML spoil leachates. In the active mine overburden leachates, Se tended to decrease with LIMB amendment (table 4), although it remained above the solid waste limit (0.30 mg L^{-1}). With PFBC amendment, Se concentrations increased at all amendment rates except 32% where concentrations decreased, apparently due to secondary mineral formation. Total Se concentration in the PFBC byproduct, however, was less than the LIMB byproduct (table 2), thus it does not appear that the PFBC material was the source of the Se. Therefore it appears that Se was mobilized from the minespoil materials. If Se in the spoils was present in part as adsorbed SeO_4^{2-} or SeO_3^{2-}, these species could have been mobilized by exchange with SO_4^{2-}. Due to their weak acid character, SeO_4^{2-} and SeO_3^{2-} should be held more strongly by soils than SO_4^{2-} (Bohn et al., 1985), thus high SO_4^{2-} concentrations would be necessary to exchange with SeO_4^{2-} and

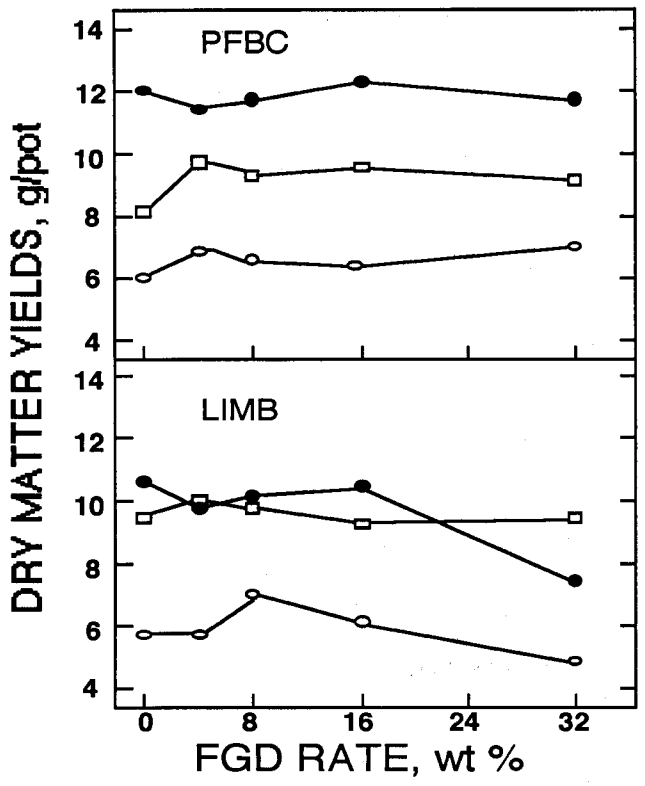

Figure 3. Fescue plant yields (topgrowth only) from first (●), third (□) and sixth (○) harvests on spoil from an active surface coal mine in east-central Ohio that was amended with PFBC or LIMB byproduct.

SeO_3^{2-}. This would account for greater Se mobilization with PFBC amendment where SO_4^{2-} concentrations went to very high levels than with LIMB amendment where SO_4^{2-} concentrations decreased. The decrease in Se in the leachate from 32% PFBC amended overburden, where there was also a large decrease in S concentration, is consistent with this mechanism.

Conclusions

FGD byproducts appear to be highly effective as alkaline amendments for acid soil and spoil. Addition of FGD byproduct effectively increased pH to near neutrality or to slightly alkaline pH and decreased concentrations of soluble Al and Mn. The result of these chemical changes is an improved growing medium for plants. Improved plant growth was, indeed, observed when FGD byproduct was mixed with acid soil or spoil. The addition of 6% sewage sludge with the FGD byproduct created the best conditions for plant growth. Concentrations of trace elements in fescue tissue generally were decreased by FGD amendment. Thus there seems to be little potential for introduction of potentially toxic elements into the food chain from the two FGD materials included in this study.

Leachate composition indicated that at application rates of 12% or lower, concentrations of elements of environmental and regulatory concern remained very low. Most, in fact, were below drinking water standard levels. Boron was an exception when LIMB byproduct was used to amend the spoil. The limiting factor for application rates of Mg-containing FGD byproducts, such as the PFBC byproduct, is more likely to be high soluble salt

Table 4. Trace element composition of leachates from columns of active mine topsoil and overburden amended with LIMB and PFBC byproducts.

FGD amendment rate, wt %	Composition, mg/L								
	As	B	Cd	Cr	Cu	Mo	Ni	Se	Zn
				LIMB AMENDED TOPSOIL AND OVERBURDEN					
0	0.05	1.40	0.046	0.021	0.04	0.018	2.45	0.94	2.67
4	< .05	1.34	.013	.018	.04	.583	< .01	1.17	.02
8	< .05	.55	.013	.036	.06	1.499	.01	.65	.01
16	< .05	4.16	.010	.021	.03	0.879	< .01	< .27	.02
32	.09	2.67	.010	.008	.23	0.964	.03	.55	.01
				PFBC AMENDED TOPSOIL AND OVERBURDEN					
0	<0.05	1.53	0.032	0.020	0.02	<0.018	1.77	1.33	1.33
4	< .05	1.00	.007	.024	.01	.090	.02	2.17	.04
8	< .05	.91	.007	.034	.01	.202	.01	3.03	.03
16	< .05	.79	.005	.036	< .01	.343	< .01	2.83	< .01
32	< .05	1.35	< .003	.009	< .01	.252	< .01	.55	.01

concentrations, which may inhibit growth and impact water quality. For the LIMB byproduct the factors limiting use are the initially high pH values that were observed and the potential for ettringite formation. Both would inhibit plant growth. However, it must be stressed that these limitations occurred only at very high application rates.

There seems to be little potential for adverse effect on water, soil, and plant quality when FGD application rates, based on the amount required to neutralize spoil acidity, are not exceeded. This last statement is made in the context that the neutralization potential of the FGD byproducts is approximately equivalent to 50% calcium carbonate. FGD byproducts with lower neutralization potentials would require higher application rates; thus, their environmental impact would differ from what is reported here.

Acknowledgments

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INFLUENCE OF AMENDMENTS ON PHOSPHOGYPSUM CHEMICAL CHARACTERISTICS AND BERMUDAGRASS GROWTH[1]

Shailesh K. Patel[2], Jerry B. Sartain[2], and Steven G. Richardson[3]

Abstract: Phosphogypsum (PG) is primarily a calcium sulfate dihydrate ($CaSO_4.2H_2O$) byproduct of the production of phosphoric acid from phosphate ore. Approximately 5 mt of PG is produced per metric ton of phosphoric acid. Acidity and nutrient deficiencies are concerns that must be addressed for successful establishment of vegetation in PG. Therefore, a glasshouse study was conducted to determine the effects of dolomite, phosphatic clay (PC), and sewage sludge (SS) on PG chemical characteristics and bermudagrass (Cynodon dactylon [L.] Pers.) growth in PG. Phosphogypsum from an active stack, International Mineral Corp. (IMC-PG), and an old, inactive stack, Estech (ET-PG), was used. The pH and EC in water (2:1, V:W) were 2.78 and 4.16 dS/m for IMC-PG, and 4.82 and 2.29 dS/m for ET-PG. According to Mehlich-I extraction interpretations, both PG sources are classified as deficient in magnesium (Mg) and potassium (K) for optimum plant growth. Dolomite, ranging from 0.87 to 6.98 g/kg, PC from 21.7 to 174 g/kg, and SS from 4.35 to 17.4 g/kg were applied to IMC-PG. Similarly, dolomite at 0.22 to 1.74 g/kg, PC at 0.87 to 6.96 g/kg, and SS at 0.87 to 3.48 g/kg were applied to ET-PG. Each rate of amendment was increased successively by a factor of 2 to give 4 rates of dolomite and PC, respectively, and 3 rates of SS, and was mixed with the PG source and incubated for six months prior to this study. At the beginning of this study, ammonium nitrate (NH_4NO_3), potassium-magnesium sulfate (K_2SO_4-$MgSO_4$), and potassium sulfate (K_2SO_4) fertilizers were selectively added to supply 50 mg/kg N, 20 mg/kg K, and 12 mg/kg Mg before sowing (0.14 g seeds/pot). Germination and growth occurred at pH 4.3 despite moderate salinity (15.6 dS/m), suggesting that attaining a desired pH is more critical than salinity for germination and growth. Phosphatic clays reduced salinity in recently decommissioned stacks that are generally higher in salts. The rates of amendments applied to ET-PG did not supply sufficient K for plant growth. Additionally, SS rates did not supply sufficient Mg. Dolomite, PC, and SS each have potential for use as nutrient sources and for raising pH. However, higher rates would be required to provide adequate nutrients in older, leached stacks and to increase pH in recently deactivated stacks.

Additional Key Words: dolomite, phosphatic clay, sewage sludge, acidity, electrical conductivity.

Introduction

Phosphogypsum (PG) is a solid byproduct of phosphoric acid production from phosphate rock. For each metric ton of phosphoric acid produced, there is a byproduct of approximately 5 mt PG. The PG is pumped as a slurry to an impoundment and allowed to settle. Small draglines are used to move some of the settled PG to raise the height of the dikes. This process results in an impoundment or "stack", which can grow up to a height of 60 m. Area of the stacks range from 20 ha to 200 ha; however, stacks currently occupy over 2,000 ha of land in Florida. Florida mining operations produce 33 million mt of PG annually (May and Sweeney 1980), with over 600 million mt already accumulated in stacks.

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[2]Shailesh K. Patel, Graduate Research Assistant, and Jerry B. Sartain, Professor of Soil Fertility, University of Florida, Gainesville, FL, USA.

[3]Steven G. Richardson, Director of Reclamation Research, Florida Institute of Phosphate Research, Bartow, FL, USA.

Owing to the inherent low-level radioactivity present in PG, mostly from radium-226, the Environmental Protection Agency (EPA) has restricted PG use both for agricultural, and research and development purposes (Federal Register 1992). The EPA's main concern is with the emanation of radon-222 gas (a radioactive decay product of Ra-226) and possible increased concentrations of radon in dwellings built on PG treated sites. Most PG, therefore, remains in approximately 20 stacks throughout Florida. When PG stacks cease usefulness as deposition sites, they will have to 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, improve surface runoff quality, and promote PG stack use as a wildlife habitat.

Sparse natural vegetation on PG surfaces indicates that establishment of a vegetative cover is possible. Reasons for sparse vegetation on PG, based on preliminary PG analyses, include residual acidity and nutrient deficiencies or imbalances, along with low nutrient-holding capacity.

Phosphogypsum is comprised primarily of the moderately soluble neutral salt, $CaSO_4.2H_2O$, and as such should have minimal effect on soil acidity. However, residual phosphoric and sulfuric acids from the acidulation process, and secondary reaction products such as hydrofluoric and hydrofluorosilicic acids, render PG highly acidic (Smith and Wrench 1984). Acidity of different PG stacks in Florida commonly ranges from 2.1 to 3.8 (May and Sweeney 1980).

Studies have shown that bermudagrass can tolerate high acidity. Lundberg et al. (1977) reported bermudagrass growth under acidic conditions (pH 3.4). Similarly, growth of "Tifgreen" bermudagrass was observed at a pH of 3.5 by Sartain (1985). Therefore, bermudagrass establishment and growth on PG may be possible.

Phosphatic clay (PC) and sewage sludge (SS) may also have the potential to reduce PG acidity and supply nutrients. Application of limestone or dolomite is a common practice for reclaiming acidic mine spoils or processing wastes (Mays and Bengston 1978). Phosphatic clay, a byproduct washed from the phosphate ore during the beneficiation process, has a neutral to slightly alkaline pH, and may be used as a possible PG amendment for reducing acidity and as a plant-nutrient source. In reclaiming quartz sand-tailings, Mislevy and Blue (1981) reported higher yields using PC (pH = 7.8) with sand-tailings than when using sand-tailings alone due to nutrients and improved water-holding capacity.

Sewage sludge generally has a near-neutral pH (Jing and Logan 1992) and, therefore, should increase PG pH upon application. Cunningham et al. (1975) reported that N, P, and K released from SS application to a sandy loam soil increased yields of corn (Zea mays L.) and rye (Secale cereale L.). In reclaiming coal stripmines with either SS or chemical fertilizer, Seaker and Sopper (1988) reported that the rate of soil development and ecosystem recovery was enhanced on SS-amended sites when compared to fertilizer-amended sites.

The objective of this study was to determine the effects of selected quantities of dolomite, SS, and PC on the chemical characteristics of PG and on bermudagrass growth.

Materials and Methods

Very acidic PG from an active stack (IMC-PG) and less acidic PG from a stack that had been closed for more than 20 yr (ET-PG) were used. Partially consolidated PC was collected from a PC settling pond at the IMC Phosphoria mine. Anaerobically digested SS was collected from an advanced wastewater treatment plant (AWTP), near the City of Tampa, FL. The dolomite was agricultural-grade crushed dolomitic limestone, mined in central Florida.

Selected dolomite, PC, and SS rates were singly mixed with each PG source and incubated for 6 months

prior to this study. Rates of dolomite mixed with IMC-PG ranged from 0.87 to 6.98 g/kg, PC from 21.7 to 174 g/kg, and SS from 4.35 to 17.4 g/kg. Similarly, dolomite at 0.22 to 1.74 g/kg, PC at 0.87 to 6.96 g/kg, and SS at 0.87 to 3.48 g/kg were mixed with ET-PG. The rates were increased successively by a factor of 2 to give 4 rates of dolomite and PC, respectively, and 3 rates of SS. Rates were selected to provide a range in pH.

Samples of raw and amended PG were analyzed for pH and electrical conductivity (EC) using a 2:1 volume to weight ratio (water:PG), and for various elements using Mehlich-I extractant according to Hanlon and DeVore (1989). According to Hanlon et al. (1993), the EC values (tables 2 and 3) were converted to estimated saturation extract EC (EC_e) as follows:

$$EC_e = (EC - 2.20) \times 8 + 2.20 \quad (1)$$

where the factor 8 was obtained by taking the ratio of water content at EC measurement to that of the material at field capacity, which for this study was 26% by weight. The constant 2.20 is the EC of water saturated with gypsum (2.41 g/L). Dolomite was analyzed for HCl-extractable elements using Method M-110 of the Florida Department of Agriculture and Consumer Services Fertilizer Laboratory. Due to the high pH of PC, it was extracted instead with Mehlich-3 extractant (Mehlich 1984) using a procedure similar to that for the Mehlich-I extraction. Elemental analysis was accomplished using inductively coupled argon plasma spectrometry. Sewage sludge samples were dry-ashed and analyzed for total elemental concentrations. Analytical results for various elements in IMC-PG, ET-PG, dolomite, PC, and SS are shown in table 1, along with Kjeldahl-N content (Bremner and Mulvaney 1982) in SS, which was determined using an autoanalyzer.

Table 1. Concentrations of selected elements in the IMC-PG, ET-PG, dolomite, phosphatic clay, and sewage sludge samples used, mg/kg.[1]

Element	IMC-PG Mehlich-I extractable	ET-PG Mehlich-I extractable	Dolomite HCl extractable	Phosphatic clay Mehlich-III extractable	Sewage sludge Total conc. Dry ashing
N	ND	ND	ND	ND	59,000
P	1,904	106	190	74	14,237
K	55	1	140	245	807
Ca	4,008	4,234	208,833	ND	ND
Mg	76	3	104,666	1,967	2,114
Zn	2	0.4	7	3	902
Cu	1	0.3	6	2	428
Mn	4	0.1	31	5	67
Fe	51	5	1,155	100	11,605
Al	102	42	ND	250	4,904
B	ND	ND	ND	ND	34
Cd	ND	ND	ND	ND	16
Pb	ND	ND	ND	ND	154
Ni	ND	ND	ND	ND	41

[1]Mean of 5 replicates.
ND = Not Determined.

Two kilograms of air-dried material from previously incubated, amended-PG were transferred into a plastic pot (12 by 15 cm). The contents of each pot were moistened using de-ionized (DI) water to which fertilizer had been applied, as solutions, to the surface. Nitrogen as NH_4NO_3 was applied at 50 mg/kg to all

pots not receiving SS. Similarly, K_2SO_4-$MgSO_4$ was applied to control units containing ET-PG to supply 20 mg/kg K and 12 mg/kg Mg, whereas K_2SO_4 was applied to dolomite-treated ET-PG to supply 20 mg/kg K. Common bermudagrass was used as a test species owing to its moderately high tolerance to salinity (Bernstein 1964), tolerance to low pH, and commercial seed availability, as well as its value as a livestock feed. Seeds were sown at a rate of 0.14 g seeds per pot. Bermudagrass aerial biomass was harvested after 10 weeks, oven-dried at 70° C, and the weight recorded. Biomass was digested using techniques similar to those for SS (dry ash), and analyzed for P, K, Mg, Zn, Mn, Cu, and Fe.

The single-factor experiment was arranged in a randomized complete block design, with four replicates. Analysis of variance, regression analyses, and single degree of freedom contrasts were used to determine significant differences within rates (Statistical Analysis System 1985). All of the analyses were conducted at the 0.05 level of probability.

Results and Discussion

Preplant Amended-Phosphogypsum Analyses

Analyses of Mehlich-I extractable elements from amended-PG prior to fertilization (tables 2 and 3) showed inadequate Mg and K concentration for optimum plant growth according to Kidder et al. (1990). Richardson (1992) also reported PG to be low in Mg and K. A comparison of the two PG sources indicates that ET-PG is consistently lower in extractable nutrient levels. This finding may be attributed to the stack having been deactivated for the past 20 yr. It thus has been subject to weathering and leaching for a greater period of time than the still-active IMC-PG stack. These differences persisted through the amendment treatments for the two PG sources as well. A linear increase in extractable Mg was observed by the addition of dolomite to both PG sources, though dolomite application to ET-PG actually decreased extractable P. Phosphatic clay applications increased extractable Mg, P, Mn, and Fe concentrations in both PG sources, and K concentration when applied to IMC-PG. Sewage sludge application increased extractable Zn, Cu, and Fe concentrations in both PG sources.

Influence of Electrical Conductivity and Acidity on Germination and Growth

Conversion of EC measurements made in a 2:1 (water:PG) ratio to estimated saturation extract EC (EC_e) values according to Hanlon et al. (1993) shows that the EC_e values for amended ET-PG ranged between 2.84 (EC = 2.28) and 3.56 (EC = 2.37 dS/m) (table 3). The protocol for EC conversion agreed well with actual EC_e measurements (Patel, unpublished data). Lower EC_e (essentially that of a saturated PG solution alone) along with a favorable pH (>4.3) contribute to germination and growth of bermudagrass in ET-PG. According to Hanlon et. al. (1993), the salinity effect threshold for bermudagrass is 6.9 dS/m, above which a yield decrease should be expected. Any 2:1 (V:W) EC in the tables in excess of 2.79 dS/m would exceed this threshold EC_e value. Low pH (<4.3) coupled with high EC_e (>6.9 dS/m) may have contributed to the lack of seed germination for several rates of SS and dolomite-amended IMC-PG. However, seed germination and growth at the highest dolomite and PC application rates for IMC-PG indicated that the influence of pH on seed germination and growth may be greater than that of EC. Dolomite and PC applications to ET-PG did not affect the EC, though SS produced a linear increase in EC (EC = 2.28 + 2.59E-5(SS), r^2 =0.76). For the IMC-PG, dolomite and SS applications did not affect EC, though PC application actually decreased EC compared to the unamended PG (table 2). Within PC rates, a linear increase in EC (EC= 2.73 + 1.92E-6(PC), r^2 = 0.48) was observed. This indicates that PC is effective in reducing the influence of salinity on seed germination when reclaiming recently deactivated stacks.

Dolomite and SS rates produced linear responses in pH (pH = 2.48 + 2.54E-4(Dolo), r^2 = 0.93 and pH = 2.72 + 2.89E-5(SS), r^2 = 0.55, respectively), whereas PC-rates produced a quadratic response in pH (pH = 4.22 + 1.33E-5(PC) - 4.95E-11$(PC)^2$, R^2 = 0.82) when applied to IMC-PG. Similarly, a linear increase in pH

Table 2. Preplant pH, electrical conductivity, and Mehlich-I extractable nutrient data for IMC-PG treated with dolomite, phosphatic clay, and sewage sludge.[1]

Amendment Rate, g/kg	pH	EC, dS/m	Calculated[2] EC_e, dS/m	Concentration, mg/kg Mg	K	P	Zn	Cu	Mn	Fe
None										
0	2.78	4.16	17.9	25	39	664	1.3	0.79	1.2	18
Dolomite										
0.87	2.84	4.34	19.3	61	42	792	1.1	0.91	1.4	18
1.74	2.91	4.21	18.3	90	41	923	1.2	1.09	2.0	30
3.48	3.18	3.93	16.0	158	41	980	0.7	0.75	1.2	28
6.96	4.34	3.87	15.6	326	48	1193	1.4	1.13	1.3	65
C.V.	2.0	7.0		32	17	22	21	23	34	47
Contrasts										
0 vs Dolo[3]	*	ns		*	ns	ns	ns	ns	ns	ns
0.87 vs 1.74	ns	ns		ns	ns	ns	ns	ns	ns	ns
1.74 vs 3.84	*	ns		*	ns	ns	*	*	*	ns
3.48 vs 6.96	*	ns		*	ns	ns	*	*	ns	*
PC										
21.7	4.44	2.84	7.32	95	45	1129	1.5	1.12	1.8	76
43.5	4.80	2.72	6.36	184	50	1515	1.5	0.90	2.7	94
87.0	4.95	2.90	7.80	304	63	2066	1.8	0.83	4.3	92
174.0	5.03	3.07	9.16	346	62	2039	1.7	0.73	4.6	58
C.V.	1.2	3.8		23	17	17	26	21	21	14
Contrasts										
0 vs PC[3]	*	*		*	*	*	ns	ns	*	*
21.7 vs 43.5	*	ns		*	ns	ns	ns	ns	ns	*
43.5 vs 87.0	*	ns		*	ns	*	ns	ns	*	ns
87.0 vs 174.0	ns	ns		ns	ns	ns	ns	ns	ns	*
SS										
4.35	2.88	4.10	17.4	30	42	765	3.1	1.38	1.4	23
8.70	2.93	3.92	16.0	33	38	737	4.1	1.94	1.4	24
17.40	3.24	4.14	17.7	45	44	843	8.1	2.79	1.8	53
C.V.	3.0	9.7		31	21	24	44	17	33	59
Contrasts										
0 vs SS[3]	*	ns		ns	ns	ns	*	*	ns	*
4.35 vs 8.70	ns	ns		ns	ns	ns	ns	*	ns	ns
8.70 vs 17.40	*	ns		ns	ns	ns	*	*	ns	*

[1] Mean of 4 replicates.
[2] Calculated EC_e = (EC - 2.20) x 8 + 2.20.
[3] Dolo, PC, SS = compared with the average of all dolomite, all PC, or all SS rates.
* = Significantly different at 0.05 level of probability.
ns = Not significantly different at 0.05 level of probability.

Table 3. Preplant pH, electrical conductivity, and Mehlich-I extractable nutrient data for ET-PG treated with dolomite, phosphatic clay, and sewage sludge.[1]

Amendment Rate, g/kg	pH	EC, dS/m	Calculated[2] EC_e, dS/m	Concentration, mg/kg Mg	K	P	Zn	Cu	Mn	Fe
None										
0	4.82	2.29	2.92	8	3.3	89	0.44	0.33	0.15	5
Dolomite										
0.22	5.03	2.29	2.92	15	4.6	76	0.61	0.40	0.15	4
0.43	4.95	2.28	2.84	21	5.3	74	0.57	0.47	0.15	4
0.87	5.13	2.28	2.84	49	2.2	70	0.74	0.38	0.19	5
1.74	5.12	2.29	2.92	64	3.0	71	0.64	0.42	0.20	6
C.V.	2.0	0.6		17	65	11	34	39	29	21
Contrasts										
0 vs Dolo[3]	*	ns		*	ns	*	ns	ns	ns	ns
0.22 vs 0.43	ns	ns		ns	ns	ns	ns	ns	ns	ns
0.43 vs 0.87	*	ns		*	ns	ns	ns	ns	ns	ns
0.87 vs 1.74	ns	ns		*	ns	ns	ns	ns	ns	ns
PC										
0.87	5.04	2.28	2.84	5	3.2	93	0.46	0.32	0.17	6
1.74	5.06	2.28	2.84	8	2.8	119	0.44	0.28	0.20	6
3.48	5.19	2.28	2.84	14	3.2	156	0.46	0.32	0.25	7
6.96	5.27	2.30	3.00	25	3.9	231	0.47	0.32	0.44	12
C.V.	1.0	0.6		11	46	7	30	33	19	17
Contrasts										
0 vs PC[3]	*	ns		*	ns	*	ns	ns	*	*
0.87 vs 1.74	ns	ns		*	ns	*	ns	ns	ns	ns
1.74 vs 3.48	*	ns		*	ns	*	ns	ns	ns	ns
3.48 vs 6.96	*	*		*	ns	*	ns	ns	*	*
SS										
0.87	4.98	2.30	3.00	5	3.7	84	1.48	0.44	0.25	10
1.74	5.04	2.33	3.24	5	3.4	111	1.89	0.62	0.23	47
3.48	5.07	2.37	3.56	6	3.6	133	3.09	0.82	0.32	83
C.V.	2.1	0.7		20	42	14	34	26	63	34
Contrasts										
0 vs SS[3]	*	*		ns	ns	*	*	*	ns	*
0.87 vs 1.74	ns	ns		ns	ns	*	ns	ns	ns	*
1.74 vs 3.48	ns	*		ns	ns	ns	*	ns	ns	*

[1] Mean of 4 replicates.
[2] Calculated EC_e = (EC - 2.20) x 8 + 2.20.
[3] Dolo, PC, SS = compared with the average of all dolomite, all PC, or all SS rates.
* = Significantly different at 0.05 level of probability.
ns = Not significantly different at 0.05 level of probability.

was observed for PC rates (pH = 5.01 + 3.86E-5(PC), r^2 = 0.54) added to ET-PG.

For the IMC-PG, approximately five times the quantity of SS as compared to dolomite was required to raise the pH to an equivalent level. Similarly, for the ET-PG, four times the quantity of PC and SS was required to raise the pH, compared to dolomite. For the IMC-PG, approximately 16 mt/ha dolomite and 50 mt/ha PC would be required to raise the pH above 4.0.

Dry Matter Yield (DMY) and Aerial Biomass Nutrient Content

For the IMC-PG (table 4), a linear increase in DMY was produced by the PC rates (DMY = 0.46 + 6.20E-5(PC), r^2 = 0.81). However, no growth at all was observed on the SS-treated PG, nor on any but the highest dolomite rate. The DMY for the highest dolomite rate was low despite sufficient nutrients, suggesting that salinity (15.6 dS/m) may have affected growth after germination. Both PC and SS rates produced a linear increase in DMY with application to ET-PG (DMY = - 0.039 + 1.00E-4(PC), r^2 =0.50 and DMY = -0.015 + 1.27E-4(SS), r^2 = 0.46, respectively), though DMY for both the PC and SS rates were lower than for the inorganically fertilized PG. When the pH was less than 4.3, no growth was observed. Richardson (1992) observed bermudagrass growth at pH 3.9 or higher on limed PG, but found no difference in yield despite higher pH values.

Tissue-nutrient levels from IMC-PG grown plants were within sufficiency ranges suggested by Jones (1980), whose nutrient sufficiency ranges were used for comparative purposes in this study. However, Mg was much lower than the critical level of 2.0 g/kg for the lowest PC rate. Low DMY at this rate may have been due to Mg deficiency as suggested by the concentration in the tissue. For ET-PG, tissue Mg concentrations were below the critical level for all plants grown on SS, for the lowest PC rate, and for the inorganic-fertilizer treated PG. Tissue K was also below the critical level of 10 g/kg for all amendments, including PG that

Table 4. Dry matter yield and nutrient concentrations of bermudagrass grown on IMC-PG treated with dolomite and phosphatic clay.[1]

Amendment Rate, g/kg	DMY, g/pot	Concentration Mg	K	P	Zn	Cu	Mn	Fe
		g/kg			mg/kg			
Dolomite								
6.96	0.17	2.1	32	6.5	55	16.4	148	52
PC								
21.7	0.30	0.8	15	10.7	40	16.1	68	34
43.5	4.31	1.2	13	8.3	32	6.0	53	33
87.0	6.85	1.8	11	8.8	21	5.8	54	31
174.0	10.74	2.8	11	7.2	27	6.0	63	38
C.V.	33	14	42	21	16	47	14	8
Contrasts								
21.7 vs 43.5	*	*	ns	ns	*	*	*	ns
43.5 vs 87.0	*	*	ns	ns	*	ns	ns	ns
87.0 vs 174.0	*	*	ns	ns	ns	ns	ns	*

[1] No growth or dry matter yield obtained for any of the other amendments or rates.
* = Significantly different at 0.05 level of probability.
ns = Not significantly different at 0.05 level of probability.

Table 5. Dry matter yield and nutrient concentrations of bermudagrass grown on ET-PG treated with dolomite, phosphatic clay, and sewage sludge.

Amendment Rate, g/kg	DMY, g/pot	Concentration Mg	K	P	Zn	Cu	Mn	Fe
		g/kg			mg/kg			
None								
0	4.18	1.3	5.6	6.3	51	6.9	12	28
Dolomite								
0.22	4.39	1.6	4.2	5.8	57	6.8	16	24
0.43	4.62	1.9	3.3	4.5	40	5.2	16	27
0.87	4.04	2.1	4.2	4.2	37	4.8	15	29
1.74	3.32	1.9	5.2	4.4	56	5.2	19	27
C.V.	20	16	17	12	21	16	17	16
Contrasts								
0 vs Dolo[2]	ns	*	*	*	ns	*	*	ns
0.22 vs 0.43	ns	ns	ns	*	*	*	ns	ns
0.43 vs 0.87	ns	ns	ns	ns	ns	ns	ns	ns
0.87 vs 1.74	ns	ns	ns	ns	*	ns	ns	ns
PC								
0.87	0.10	1.2	3.6	4.3	101	4.5	25	41
1.74	0.05	2.3	3.6	5.1	167	2.5	60	48
3.48	0.34	1.8	3.0	4.4	246	5.3	39	38
6.96	0.60	2.1	2.5	4.3	144	4.8	38	37
C.V.	29	17	18	9	34	34	16	21
Contrasts								
0 vs PC[2]	*	*	*	*	*	*	ns	*
0.87 vs 1.74	ns	*	ns	ns	ns	ns	*	ns
1.74 vs 3.48	ns	ns	ns	ns	*	ns	*	ns
3.48 vs 6.96	ns	ns	ns	ns	*	ns	ns	ns
SS								
0.87	0.17	0.9	4.1	5.1	183	6.6	26	46
1.74	0.10	0.8	2.3	4.0	187	6.8	20	45
3.48	0.46	0.8	3.2	3.8	236	7.6	33	58
C.V.	18	20	17	15	22	23	15	35
Contrasts								
0 vs SS[2]	*	*	*	*	*	ns	*	*
0.87 vs 1.74	ns	ns	*	ns	ns	ns	*	ns
1.74 vs 3.48	*	ns	ns	ns	ns	ns	ns	ns

[2] Dolo, PC, SS = compared with the average of all dolomite, all PC, or all SS rates.
* = Significantly different at 0.05 level of probability.
ns = Not significantly different at 0.05 level of probability.

received only inorganic fertilizer. The low DMY in PC-amended ET-PG may be attributed to K deficiency, while that in SS-amended ET-PG may be due to both Mg and K deficiencies. Recently, Shahandeh and

Sumner (1993) reported that fertilization with 150 mg/kg K, along with 100 mg/kg N and 12 mg/kg Mg, resulted in the best total plant growth for a mixture of weeping lovegrass (Eragrostis curvula) and lespedeza (Lespedeza cuneata) grown in artificially leached PG that had a pH of 4.1. Bermudagrass tissue concentrations in PC- and SS-treated ET-PG showed Mn and Fe to be adequate, but Mn and Fe were below the sufficiency range for all dolomite rates. Despite having comparable Mg and K concentrations in its tissue, the PC treated ET-PG showed much less growth than the dolomite treated PG. A twofold to fivefold increase in Zn concentration was also observed with PC and SS applications, which seems to suggest possible Zn toxicity to bermudagrass. However, Sartain (1992) reported that Zn applications did not affect bermudagrass growth despite tissue concentrations ranging from 124 to 355 mg/kg Zn.

Conclusions

Bermudagrass germination and growth on PG occurs at pH around 4.0. This study showed germination and growth on PG at pH 4.3 while Richardson (1992) reported growth at pH 3.9. Phosphogypsum pH influences seed germination and growth to a larger extent than salinity. Chemical-fertilizer rates of Mg and K used in this study were not sufficient to adequately provide the needed amounts of these nutrients for optimum plant growth, based on tissue analyses. Dolomite was effective in raising the pH and supplied Mg. Phosphatic clays reduced the effects of salinity on plants growing in recently decommissioned stacks that are generally higher in salts than older, leached stacks. The PC was also effective in increasing pH and added nutrients such as Mg, K, Mn, and Fe. Sewage sludge also increased pH and supplied nutrients such as Zn, Cu, and Fe. Dolomite, PC, and SS have potential for use as nutrient sources and also for raising pH. Application rates of amendments will have to be higher than those used in this study to provide adequate nutrients for plant growth in older, leached stacks, and to increase pH in recently deactivated stacks.

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GROWTH AND DEVELOPMENT OF SNOW BUCKWHEAT (*ERIOGONUM NIVEUM*) ON XERIC MINE SPOILS OF AN ABANDONED URANIUM MINE IN EASTERN WASHINGTON[1]

Benjamin A. Zamora and James W. Leier[2]

Abstract: Snow buckwheat (*Eriogonum niveum*) is a native of the steppe region of the northern intermountain region in the Pacific Northwest. The habitat types in which this species is abundant represent some of the most xeric environments of the region. The species is an important dietary plant to wildlife (elk, deer, and birds) and has a vigorous colonizing ability. The Soil Conservation Service Plant Materials Center at Washington State University, Pullman, WA, developed and released an accession of 'Umatilla' snow buckwheat as conservation plant material for use in rangeland rehabilitation, wildlife upland habitat improvement, and critical area stabilization. The plant is a half-shrub, growing to 7 dm under ideal conditions, but generally averaging 3-6 dm maximum canopy height with a spreading canopy of 3-9 dm. A soil amendment and mulching study on xeric mine spoil sites was started in 1991 to determine the suitability of this species for use in native plant revegetation efforts on reconfigured and resoiled mine spoils. By the end of the 1993 growing season, 87% of the original plants had survived. Composted sewage sludge amendment at rates of 13.45, 26.9, and 53.8 t/ha (overdry weight) combined with mulch covers enhanced canopy development and reproductive vigor of the plants. Seedling establishment around parent plants is most evident on treatments with no mulch and high application rates of composted sewage sludge. Mulching with woodchips substantially reduces seedling establishment, suggesting that site amelioration to promote stand development of this species must consider soil surface characteristics created by the treatment process.

Additional Key Words: mine spoil revegetation, drought-tolerant shrub, composted sewage sludge, woodchip mulching.

Introduction

A substantial amount of the hard rock mining in the state of Washington occurs within forested areas. The forests of these areas represent vegetation types typical of the temperate xerophytic and mesophytic forest regions. Soil moisture is primarily recharged annually by winter rain and snow which varies from 460-1,000+ mm per year depending on geographic location. Mining within these kinds of ecosystems presents some rather difficult reclamation problems in terms of the hydrology and the complexity of microsite mosaics created by the disturbance. Commonly, the disturbance of stratigraphy creates highly permeable, deep mine spoil deposits, and radical changes in microtopography and relief. These changes commonly result in more xeric surface conditions than normal. Under these conditions, revegetation strategies emphasizing native plant re-establishment may need to consider species more adapted to drier habitats than the native species that previously occupied the sites. Selection of species, however, does not preclude the use of native species of the *area* (not specific to the original site) to revegetate and potentially establish advanced stages of native vegetation succession on the reclaimed site.

The purpose of this study was to evaluate snow buckwheat (*Eriogonum niveum*) for use in

[1]Paper presented at the International Land Reclamation and Mine Drainage Conference and the Third International Conference on the Abatement of Acidic Drainage, Pittsburgh, PA, April 24-29, 1994.

[2]Benjamin A. Zamora, Associate Professor of Range Management, and James W. Leier, Graduate Research Assistant, Department of Natural Resource Sciences, Washington State University, Pullman, WA, USA.

revegetating droughty site conditions created by mine spoil deposition in forest areas of the inland Pacific Northwest. Snow buckwheat is a native of the steppe region of the northern intermountain region of the Pacific Northwest (fig. 1). The species range extends from southern British Columbia to extreme northern Nevada and across the Columbia Plateau between the Cascade Range of Washington and the Bitterroot Range of northern Idaho. All of the habitat types in which this species is found represent some of the most xeric environments of the region. The species has been identified as a common food component in diets of big game species in the Pacific Northwest (Burrell 1982, Tiedemann and Driver 1983). An accession of 'Umatilla' snow buckwheat was released by the Soil Conservation Service Plant Materials Center at Pullman, WA, for use in rangeland rehabilitation, wildlife upland habitat improvement, and critical area stabilization (Lambert 1991, 1992).

Figure 1. An illustration of snow buckwheat (*Eriogonum niveum*) (*from Hitchcock et al. 1964, with permission*).

Generally the species is best adapted to precipitation zones of 170-380 mm, though the plant may be found in higher precipitation zones on sites with less effective moisture resulting from soil and/or topographic conditions.

The plant is a half-shrub, growing to 7 dm under ideal conditions, but generally averaging 3-6 dm maximum canopy height with a spreading canopy of 3-9 dm. This species is drought tolerant, has a root system adapted to gravelly and rocky soils, and can colonize grass-covered surfaces of disturbed sites. Because of its winter persistent leaves, continual production of leaves throughout the summer, and extensive root system, the plant can potentially contribute significantly to the transpiration component of the water budget of disturbed moist sites. The plant is common within the ponderosa pine/bitterbrush (*Pinus ponderosa / Purshia tridentata*) zone at elevations within 150 m below the mine site. The soil materials of these natural habitat types are comparable in terms of moisture conditions to the mine spoil deposits of the study area.

The specific objective of the study was to measure the establishment and growth of plantings of snow buckwheat on mine spoil deposits to determine its adaptability to the higher moisture climate and response to composted sewage sludge amendments and mulching treatments of the mine spoil material.

Materials and Methods

Study Area

The study is being conducted on an abandoned, open-pit, hardrock uranium mine on the Spokane Indian Reservation, Stevens County, in east-central Washington. Precipitation at the site averages 450 mm per year, which falls primarily in the winter. The principal native vegetation of the area surrounding the mine is of the ponderosa pine / bluebunch wheatgrass (*Pinus ponderosa / Agropyron spicatum*) and ponderosa pine/snowberry(*Pinus ponderosa / Symphoricarpos albus*) plant associations. The mine lies

within an ore body that is located in a metasedimentary formation, which contains substantial amounts of pryitic mica phyllite. Exposure of the pyrite has initiated acid rock drainage. The groundwater flow system and leaching characteristics of wastes at this mine could have significant effects on the quantity and quality of impounded and soil water and impact the quality of ground water leaving the mine site. Consequently, our revegetation and resoiling research at this site is focused on options for installation and on-site development of soil and vegetation covers that would provide long-term surface stability and act as deterrents to deep percolation of meteoric water to buried, acid-generating waste rock deposits.

Planting Design

Field plantings of six month old containerized snow buckwheat plants were established during July of 1991 with the aid of drip irrigation. Drip irrigation was not used after the fall of 1991. The plot area was first graded and ripped with heavy equipment to break up compacted mine spoil, then topdressed with 2 dm of clay subsoil designated for eventual use in the reclamation of the mined area. Three rates of composted sewage sludge (13.45, 26.9, and 53.8 t/ha, oven-dry weight metric tons) were combined with a 5 cm deep, topdressed woodchip mulch to create eight treatments in a randomized complete-block design (table 1). The chemical character of the composted sludge was as follows: pH 5.2, NO^3-N 5.1 ug/g, NH^4-N 95.6 ug/g, %N 2.32, %C 30.14. No additional nitrogen was added to the sludge to balance the carbon/nitrogen ratio. Ten containerized seedlings were planted per replication at 6 dm spacings within a 1.8x3.6 m plot in which the soil amendment treatment was applied.

Table 1. Percent survival of snow buckwheat (*Eriogonum niveum*) plants on amendment treatments of mine spoils in 1992 and 1993.

	% Survival	
Treatments	1992	1993
No mulch, no compost	94	91
Mulch, no compost	92	92
No mulch, 13.45 t/ha[1]	96	85
Mulch, 13.45 t/ha	90	85
No mulch, 26.9 t/ha	90	88
Mulch, 26.9 t/ha	82	78
No mulch, 53.8 t/ha	95	95
Mulch, 53.8 t/ha	88	83

[1] Oven-dry weight composted sludge

Measurements and Analyses

Measurement and evaluation of the plantings were initiated in 1992. Survival of plants and seedling establishment was assessed for each treatment at the end of the growing season each year. Canopy area and volumes, number of floral bolts per plant, and average maximum height of the vegetative and floral components of canopies were made at the time of full floral development in late August to early September. Canopy areas of both the vegetative part of the canopy and floral component of the plants were made by measuring the longest diameter of the canopy ellipse and the longest perpendicular diameter to this long axis. Within this elliptical area, the coverage of live canopy was estimated. The estimate of canopy area for each plant was then calculated as the area of the ellipse minus the percentage of unoccupied space and expressed in dm^2. Floral canopy area was estimated in the same manner. Volume (dm^3) estimates of the vegetative canopy were based on an elliptical cylinder by incorporating an average maximum height dimension into the canopy area calculation. For the floral canopy component, the frustum of an elliptical cone was used to determine volumetric space occupied by that portion of the canopy occupied solely by flowering stems (Mawson et al. 1976). Canopy volume is a multidimensional measure of canopy structure that may provide additional insight about species dominance of plant communities (Zamora 1981). Statistical analyses of the data were accomplished with the Tukey HSD simultaneous pairwise mean comparison method in the Systat Statistical Program (SYSTAT 1992).

Results and Discussion

Growth Pattern

The developmental pattern of both vegetative and floral growth of snow buckwheat over the two full growing seasons of observation did not differ between the eight treatments. Snow buckwheat is indeterminate in its floral and seed development, producing flowers and mature seed over an extended period of time. The enriched soil nutrient conditions of some treatments may have accentuated this attribute in some plants by prolonging the period of floral development.

Snow buckwheat has an herbaceous but winter-persistent leaf canopy that develops from a base of woody prostrate stems. Growth of the plant begins in March or early April and accelerates to a maximum leaf area in the canopy by the first part of July. Floral bolt initiation may be evident in late May or early June but generally does not accelerate until after the majority of the maximum leaf area has developed in the canopy. It is at this point that enriched habitats of plants may stimulate an accelerated rate of floral stem initiation. With the rapid growth of floral stems, leaf area significantly declines with older leaves senescing and dropping out of the functional leaf pool of the canopy. The carbon balance within the plant shifts dramatically to support floral development, and remains in this status until after seed shatter, which occurs in September. There is evidence within the genus that the floral stems actively carry on photosynthesis, replacing that particular role of leaves during the reproductive phase of the annual growth cycle (Smith and Osmond 1987, Osmond et al. 1987). The floral stems of snow buckwheat do have a significant number of leafy bracts, which may add to the photosynthetic capability of the stem. Following seed shatter, growth modules, which form the coming year's canopy growth, are rapidly initiated before fall/winter temperatures preclude further meristem development. Some proportion of the larger and later developing leaves of the current year's growth will persist through the winter to form a pool of leaf tissue that will be photosynthetically active at the start of next year's growth cycle.

Survival of Plants

Survival of the container grown plants from 1991 into the 1992 growing season averaged 96% across all treatments. By the end of the 1992 growing season, an average of 91% of the plants had survived across all treatments (table 1). By the end of the 1993 growing season, only an additional 4% of the plants had been lost. Some plant loss was due to browsing and uprooting of plants by elk. No statistically significant differences were evident between treatments in either year.

Canopy Structure

Average total canopy volume per plant increased between 1992 and 1993 in seven of the eight treatments, but was not found to be statistically different between the two years across the treatments (fig. 2). During the first year of establishment in 1991, plants developed substantially more vegetative canopy than floral canopy. This vegetative canopy overwintered into the 1992 growing cycle, contributing significantly to the vigorous establishment of the plants. The vegetative volume component of the canopies, however, changed very little between 1992 and 1993 when the plants entered into a typical reproductive phase. The floral volume component of the canopy did demonstrate a greater degree of increase between 1992 and 1993 indicating the degree to which the plants were allocating greater resources to the reproductive process. In 1992, there was a stronger tendency for treatments with mulch covers and higher rates of composted sewage sludge amendment to produce statistically greater canopy volumes than treatments with lesser rates of amendment with or without mulching. In 1993, the number of significant differences between treatments declined with the most evident difference being between the mulch, no compost amendment and the mulch, high compost amendment treatments.

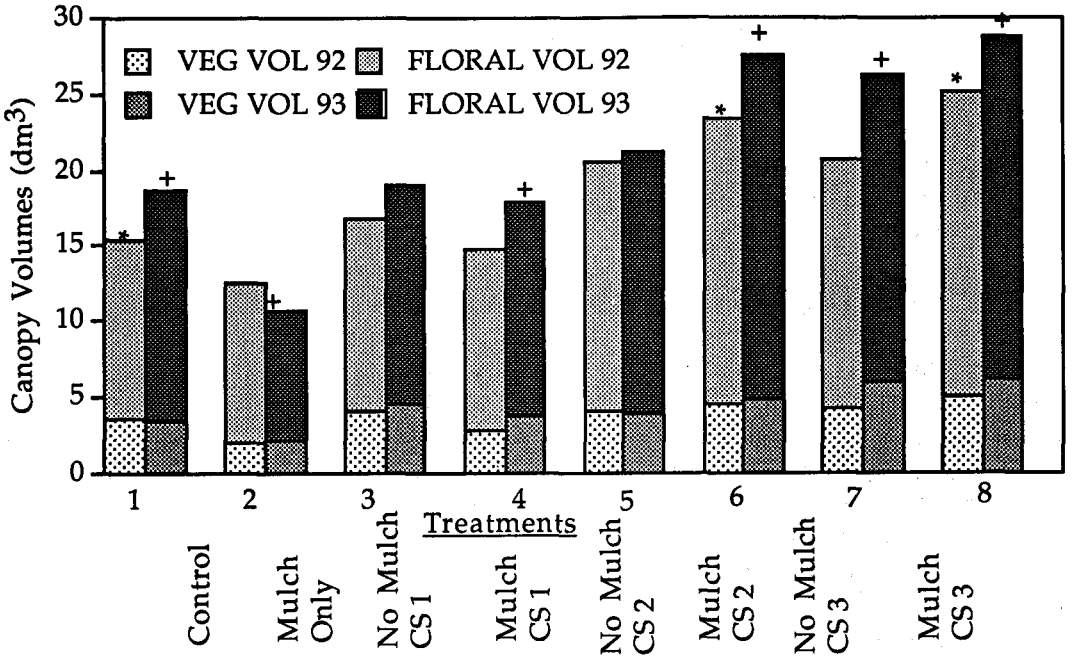

Figure 2. Average vegetative and floral canopy volumes of snow buckwheat (*Eriogonum niveum*) among soil amendment treatments in 1992 and 1993. Composted sludge levels, CS1 = 13.45 t/ha., CS 2 = 26.9 t/ha., CS 3 = 53.8 t/ha. (*)Treatments 1, 2, and 4 differ from treatments 6 and 8; (+)Treatment 2 differs from treatments 6, 7, and 8 at the 0.1 probability level using Tukey HSD simultaneous pairwise mean comparisons.

Both vegetative and floral canopy area showed little change between years although some significant differences were discernible between some treatments within years (fig. 3 and 4). Canopy area was distinguished from other canopy measures as a means of identifying lateral canopy expansion on a yearly basis. Since snowy buckwheat appears to be self limiting in terms of height, lateral expansion of the canopy should dictate changing canopy structure and leaf area. Average height of the vegetative canopy, depth of the floral component of the total canopy, and total height did not significantly differ between treatments and differed only to a small degree between years (table 2). In general, the trend in average canopy area was from smaller average canopy areas in treatments with no compost amendment or low rates of compost to larger canopy areas with high rates of compost amendment with mulch covering (fig. 3 and 4).

Reproductive Vigor

Reproductive vigor of the plants, as measured by the number of floral stems per plant, was similar in response to treatments as measured with canopy volume and area (fig. 5). Generally, plants averaged fewer flowering stems in treatments with no compost or low rates of compost amendment than plants growing in higher rates of compost amendment with mulch covering.

Seedling Establishment

Seedling establishment was evident among the eight treatments at the start of the 1993 growing season. No seedlings were observed on the treatment sites prior to persistent snow cover the previous

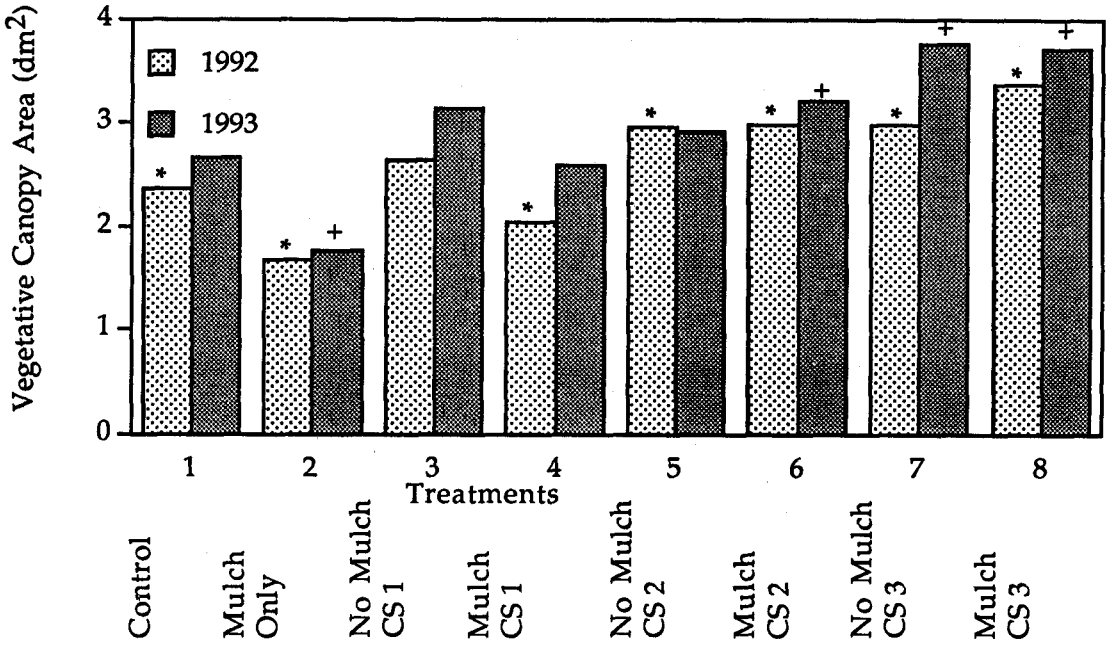

Figure 3. Vegetative canopy areas of snow buckwheat (*Eriogonum niveum*) in 1992 and 1993 among soil amendment treatments. Composted sludge levels, CS1 = 13.45 t/ha., CS2 = 26.9 t/ha., CS3 = 53.8 t/ha. (*)Treatments 1 and 4 differ from treatment 8, treatment 2 differs from treatments 5, 6, 7, and 8; (+)treatment 2 differs from treatments 6, 7, and 8. Significant differences at the 0.1 probability level, Tukey HSD simultaneous pairwise mean comparisons.

Table 2. Mean heights of canopy components of snow buckwheat (*Eriogonum niveum*) plants on amendment treatments of mine spoils in 1992 and 1993.

	Mean Height (cm) of Canopy Component					
	Vegetative		Floral		Total	
Treatment	1992	1993	1992	1993	1992	1993
No mulch, no compost	13	11	25	27	38	38
Mulch, no compost	10	11	25	22	35	33
No mulch, 13.45 t/ha[1]	13	13	26	26	39	39
Mulch, 13.45 t/ha	12	12	26	25	38	37
No mulch, 26.9 t/ha	13	12	28	28	40	40
Mulch, 26.9 t/ha	13	12	27	27	40	40
No mulch, 53.8 t/ha	12	13	26	26	39	39
Mulch, 53.8 t/ha	13	14	27	27	40	41

[1] Oven-dry weight composted sludge

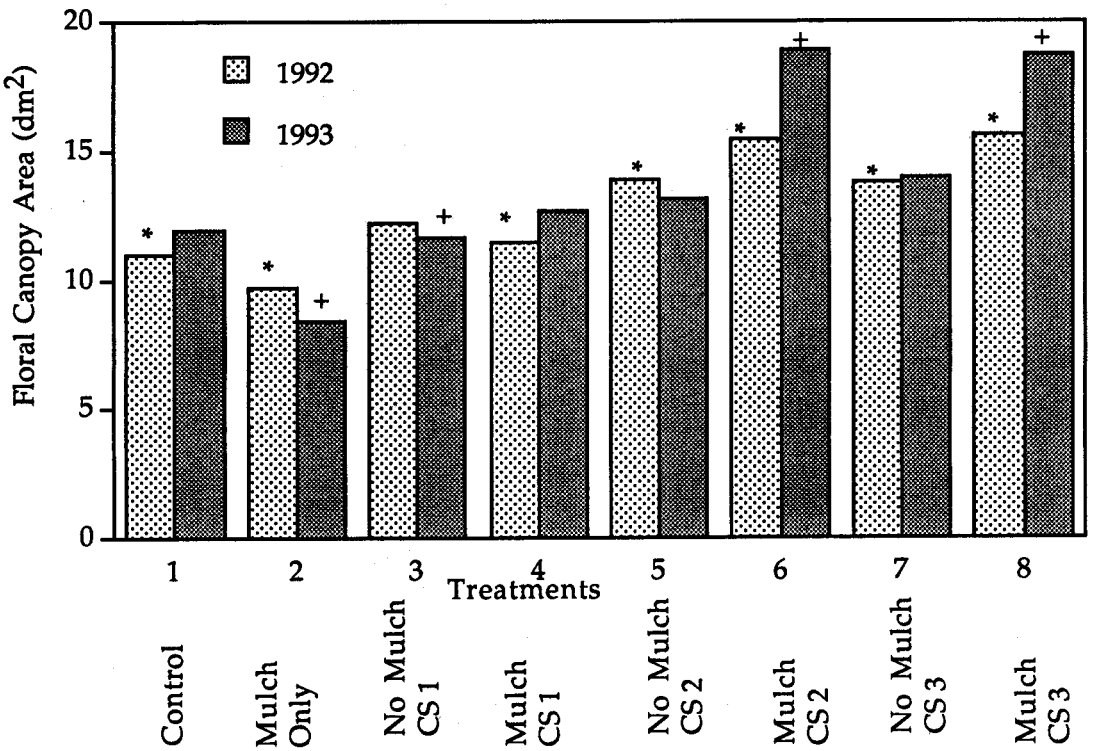

Figure 4. Floral canopy areas of snow buckwheat (*Eriogonum niveum*) in 1992 and 1993 amomg soil amendments treatments. Composted sludge levels, CS1 =13.45 t/ha., CS2 = 26.9 t/ha., CS3 = 53.8 t/ha. (*)Treatments 1 and 4 differ from treatments 6 and 8, treatment 2 differs from treatments 5, 6, 7, and 8; (+)treatment 2 differs from treatments 6 and 8, treatment 3 differs from treatment 6. Significant differences at the 0.1 probability level, Tukey HSD simultaneous pairwise mean comparisons.

fall so germination may have taken place under a heavy snowpack or very rapidly following snowmelt. Germination trials with this species suggest that if germination does not take place within days following seed deposition on the soil surface, secondary dormancy develops and the seed becomes a dormant element of a long-lived seed bank in the soil until disturbance occurs. Seedling establishment appeared to be influenced by surface conditions within the treatment site (table 3). Seedling establishment was inhibited by mulch cover. However, the higher rates of compost amendment (26.9 and 53.8 t/ha) significantly enhanced seedling establishment. After incorporation, sludge residues are still evident on the surface of the plots. These residues darken the soil surface, produce a considerable range of soil surface textures, and contribute to a diverse mosaic of surface conditions and microtopography. These conditions may contribute to an enhanced environment for seedling establishment.

Conclusions

Survival of snow buckwheat on amended mine spoils exceeded expectations after two growing seasons with an average loss of only 13% of the plants among all treatment plantings in the experiment. Vegetative and reproductive growth of all plants was vigorous across all amendment treatments. Generally, composted sewage sludge amendment rates of 26.9 and 53.8 t/ha produced greater canopy growth and reproductive vigor among the plants. Since 1992 and 1993 were years of average to above

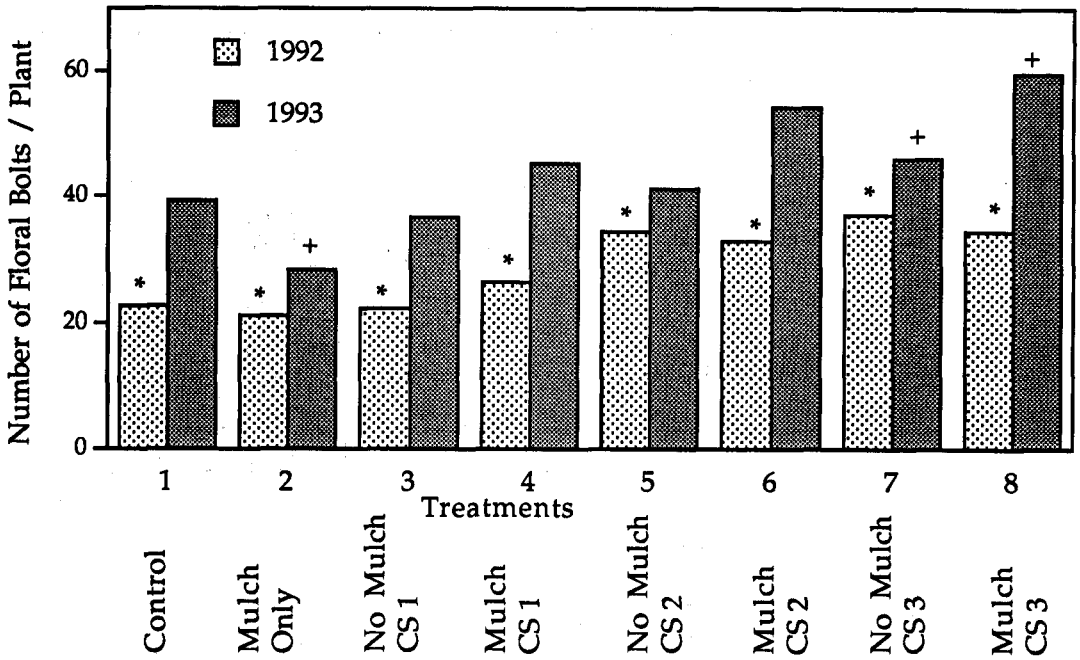

Figure 5. Average number floral bolts of snow buckwheat (*Eriogonum niveum*) per plant in 1992 and 1993 among soil amendment treatments. Composted sludge levels, CS1 = 13.45 t/ha., CS2 = 26.9 t/ha., CS3 = 53.8 t/ha. (*)Treatment 1 and 3 differ from treatments 5, 7, 8, treatment 2 differs from treatments 5, 6, 7, 8, treatment 4 differs from treatment 7; (+)treatment 2 differs from treatments 6 and 8. Significant differences at the 0.1 probability level, Tukey HSD simultaneous pairwise mean comparisons.

average precipitation, the mulch treatment probably did not contribute substantially to an improved environmental microsite condition for each plant. The enriched soil N condition however, likely caused the increase in canopy development and reproductive vigor. Seedling establishment was significantly reduced by the mulching treatment but enhanced by the higher rates of composted sewage sludge

Table 3. Percent snow buckwheat (*Eriogonum niveum*) seedling occurrence on amendment treatments of mine spoils in 1992 and 1993.

	Seedling Occurrance		
Treatments	Absent	P[1]	A[2]
No mulch, no compost	63	17	20
Mulch, no compost	82	14	4
No mulch, 13.45 t/ha[3]	60	30	10
Mulch, 13.45 t/ha	79	21	0
No mulch, 26.9 t/ha	59	16	25
Mulch, 26.9 t/ha	82	18	0
No mulch, 53.8 t/ha	38	25	37
Mulch, 53.8 t/ha	80	16	4

[1] P= <10 seedlings within 3 dm radius of plant.
[2] A = >10 seedlings within 3 dm radius of plant.
[3] Oven-dry weight composted sludge

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DIRECT REVEGETATION OF FLY ASH--A GREENHOUSE STUDY[1]

Richard I. Barnhisel[2]

Abstract: Two greenhouse experiments were conducted on fly ash to determine if such material could be vegetated without covering it with soil, in anticipation of a field research study. Corn was used as a test crop in both studies. In the first study, very poor growth was obtained. In spite of high soil test levels for P, plants exhibited extreme P deficiencies. This was intensified when K was applied. Plants appeared normal for the first 3 weeks, then rapidly exhibited deficiency symptoms with total P often <0.1 ppm. In the second experiment, the same fly ash source was mixed with either sewage sludge or soil or both and three rates of each soil and sewage sludge were used with a fly ash control. Unlike the first experiment, only one rate of P and K was used. The plants of the control were similar to those in the first experiment; however, both sewage sludge and soil amendments produced significantly greater yields as well as reduced P deficiency symptoms. After the second crop, K was applied to all pots because plant tissue levels were approaching deficient levels. By the fourth crop, plants began to exhibit low P levels and were approaching a critical level in spite of the fact soil test levels remained high. It is apparent that the soil tests were extracting greater levels of P than were available to the corn test crop.

Additional Key Words: phosphorus, soil test, Bray-1P, Mehlich III-P.

Introduction

Environmentally acceptable, cost-effective disposal of large quantities of fly ash produced by coal-burning power plants poses a significant challenge, especially in view of changes in the regulations associated with landfills. When landfill closure requires a 4-ft soil cover, removal of adjacent soils may cause significant environmental problems, with impacts approaching those of the landfill itself.

Direct revegetation of fly ash disposal sites would reduce land disturbances, provided suitable methods can be developed. Significant amounts of information are available in the literature that indicate direct seeding of fly ash may offer a successful alternative to current and proposed practices.

As a result, a demonstration study was conducted on a fly ash disposal site by East Kentucky Power Cooperative (EKPC). Questions were raised concerning the obvious P deficiency symptoms observed for some plant species in spite of the very high soil test levels for this element. In an attempt to answer this question, a greenhouse study was initiated. This first experiment had two main objectives: determining (1) if the P soil tests were valid, and (2) if the apparent differences could be corrected by adding P fertilizers, in spite of high soil test levels.

Literature Review

A feasibility study conducted in 1976 by the U.S. Environmental Protection Agency (Dougherty and Holzen) concluded pulverized fuel fly ash could be used effectively to produce a soil cover capable of sustaining grasses and legumes on regraded acidic surface mine spoils. Such reclamation was found effective in producing usable land and in improving water quality.

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[1]Paper presented at the International Land Reclamation and Mine Drainage Conference and the Third International Conference on the Abatement of Acidic Drainage, Pittsburgh, PA, April 24-29, 1994.

[2]Richard I. Barnhisel, Professor of Agronomy and Geology, University of Kentucky, Lexington, KY, USA.

Wendell et al. (1992) conducted a 3-year study to examine the suitability of lime-stabilized fly ash scrubber sludge (FASS) in surface mine reclamation. The primary concern of using FASS in row crop production was the effect of B concentrations on plants that could reach toxicity levels with subsequent yield reduction.

In an associated study using the same FASS, Zhang et al. (1992) evaluated the impact of this soil amendment upon the growth of legumes and grasses. An addition of 2.5% to 5.0% FASS by weight to an acidic topsoil produced an increase in soil pH and salt level, and an associated increase in yield for alfalfa, birdsfoot trefoil, and tall fescue. FASS was contributed to improved soil water retention, supplied supplemental Ca, S, Cl, B, Mo, and Mn, neutralization of acidity, and increased plant yields.

Growth and elemental analysis studies were conducted on trees growing on abandoned coal fly ash basins (Carlson and Adriano 1991). Trees found to contain the greatest trace element concentrations also attained the greatest height and diameter growth. Size differences between trees grown in soil and fly ashes were attributed to increased water-holding capacity in the fly ash substrate. Raw fly ash from bituminous coal-fired power plants was used by Adams et al. (1971) in the reclamation of coal surface-mined lands. Fly ash amendment was found to increase pH and improve soil texture and available water-holding capacity. Of the grasses and legumes planted, Kentucky 31 tall fescue, rye, and red top grasses, and birdsfoot trefoil survived the best.

Fly ashes have been applied in the reclamation of abandoned mine spoils to produce agricultural crops (Ghazi et al. 1987). Excellent yields of alfalfa were produced when as much as 350 t/a of fly ash had been applied, as long as adequate P was supplied to these acidic spoils. In this case, fly ashes were serving as a liming agent as well as a source of some plant nutrients.

Usually fly ash contains virtually no N and has little plant-available P (Martens 1971, Adriano et al. 1978). Appropriate fertilizer rates to accommodate these deficiencies would be indicated.

Oyler (1989) tested the use of sewage sludge, fly ash, lime, and potash in mixtures to vegetate areas containing high concentrations of particular metals in the soil. A 1:1 ratio of sludge to fly ash was found to be most successful. The metals-tolerant herbaceous species used were switchgrass, big bluestem, tall fescue, birdsfoot trefoil, flatpea, perennial ryegrass, and intermediate wheatgrass.

Francis et al. (1985) found that ryegrass grown in unlimed and unfertilized coal gasification fly ash and soil mixtures contained very high concentrations of Al, B, Cd, Co, Mo, Ni, and Zn, many of which were considered to be phytotoxic. However, it was demonstrated that with proper liming and fertilizing practices, risk of transport of these metals through the food chain was eliminated. Similar conclusions were reported by Keefer and Singh (1985).

In a similar study, Sopper and McMahon (1988) used mixtures of sludge and fly ash to successfully aid in the establishment of grasses and legumes in soils highly contaminated from operations of a zinc smelter. A 2:1 ratio of sludge to fly ash resulted in the highest average percent vegetation cover and dry matter production. The seed mixture exhibiting the best performance was orchardgrass, tall fescue, and crown vetch.

Methods and Materials

Greenhouse Experiment I

Several kilograms of fly ash, which had been previously pumped to sediment ponds, were collected at the John Sherman Cooper Plant near Somerset, KY. At this plant, two ponds exist, and ash samples were collected from both. One pond contains old ashes and was filled to capacity at least 10 years ago. The second pond was currently receiving ashes. These two bulk samples were dried, sieved through a 2-mm screen, and mixed to provide a uniform sample.

Pots large enough to hold 2 kg each were lined with plastic. Three separate studies were conducted, two using ashes from each pond and the third using fly ash collected directly from the precipitators at the power plant. A 3 x 3 factorial design was used in each case.

Three rates of P and K were chosen, representing 0, 56, and 112 kg/ha for P, and 0, 140, and 280 kg/ha for K, assuming a furrow slice weighs 2 x 10^6 kg. For each study, three replications were used. Nitrogen was initially added to each pot at a rate equivalent to 200 kg N per hectare. The fertilizers were mixed with the ashes and placed in the pots. Midway during the growing period, week 3, an additional application equivalent to 75 kg N per hectare was applied.

Deionized water was added to raise the ashes to field capacity or about 38% water on a dry weight basis. In each pot, holes were dug 2 cm deep, six corn seeds were placed in a symmetrical pattern and covered. Corn plants were grown 6 weeks and harvested. A second cropping was started after a 100-g portion of ashes was removed from each pot for soil testing. This process was repeated two more times.

The plant tissue was dried at 60°C, ground to <10 mesh, and analyzed using standard wet-ash for P, K, Ca, Mg, and Na, and micro-Kjeldahl N tests published in American Society Agronomy (ASA) Monograph 9. The ash samples were analyzed for available P by both the Bray-1P and Mehlich III extracting solutions. Exchangeable Ca, K, Na, and Mg were also determined using methods in ASA Monograph 9. The texture, available water-holding capacity (AWC), and pH were determined by commonly used methods described in ASA Monograph 9.

Greenhouse Experiment II

In general, the procedures were the same, except in this case ashes from only the old pond were used. The main treatments were equivalent to 0, 18, and 30 Mg/ha of sewage sludge added to the following: (1) a fly ash control, (2) a 25:75 mixture of subsoil to fly ash, and (3) a 50:50 mixture of subsoil to fly ash. The subsoil was obtained from a stockpile of unknown source adjacent to the old ash pond; it is believed that it had been removed during the construction of the pond and was likely from the Pembroke soil (fine-silty, mixed, mesic Mollic Paleudalfs). If soil was to be required for covering the ash pond, it is likely that at least some of it will come from this stockpile.

Results and Discussion

Greenhouse Experiment I

Data for selected physical and chemical properties are given in table 1 for the "old" and "new" fly ashes. The "new" ashes had a lower AWC because they had a higher sand content. This is most likely a function of the location within the respective ponds from which the ashes were collected with respect to the outlet of the pipe carrying the ashes. Both fly ash and bottom ashes were pumped into both of these ponds. The "new" ashes had higher nutrient levels of P, K, Mg, and Ca. Any value for P >68 kg/ha would be considered high had this been an agricultural soil. In spite of these high P levels, the plants were experiencing severe P deficiencies. For that reason, the Mehlich III P test was used to test P availability. A large discrepancy exists between the two methods for P. In fact the Mehlich III-P was proportionally higher and exceeded the upper detection limit (224 kg/ha) without dilution of the extract. Since the Mehlich III extract was measured with an inductively coupled plasma emission spectrometer (ICP), it is unlikely that this high P value would represent soluble silica from the ashes as was originally believed to be the case for the Bray-1P measurement. For Bray-1P, an ammonium molybdate was used to determine P, under some conditions it can also react with Si to give erroneous values.

The average biomass produced in each of the four growth periods from both of the P and K treatment combinations for fly ashes from the "old" sediment pond are given in table 2. Similar trends for corn growth were obtained for all four harvest dates. The largest response was for added K, especially between the 0 and 140 kg/ha,

with little if any additional yield response between 140 and 280 kg/ha. The response to K increased somewhat for each rate of applied P, with the largest values occurring for the 100 kg/ha P and 280 kg/ha K treatments three of the four harvests.

Little if any response in yield was observed for applied P at any K rate for the first harvest. However, there appeared to be some response to P for subsequent harvests, at least for those treatments that also received K. In other words, there was an interaction between P and K. Although yield responses to P and K were obtained, yields were much lower than normal (as compared with a response commonly observed for agricultural soils).

Table 1. Physical and Chemical Properties of Fly Ash Samples.

Sample	AWC[3]	Texture				
		Sand	Silt		Clay	
	%	----------	%	----------		
Old Ashes	40.0	16.1	82.7		1.2	
New Ashes	27.7	35.9	60.3		3.8	
	Bray-1P	P[4]	K	Ca	Mg	pH
	----------	----------	kg/ha	----------	----------	
Old Ashes	57.3	224+	87	683	53	6.8
New Ashes	96.3	224+	126	974	87	6.7

[3]AWC = available water-holding capacity, i.e., field capacity - wilting point.
[4] P = Mehlich III available phosphorus.

Table 2. Yield of corn (forage) in response to P and K additions to fly ash from old pond.

Treatment		------- Harvest Date -------			
P	K	1	2	3	4
		------ g dry wt/treatment ------			
0	0	2.92	3.00	1.76	2.47
0	140	3.99	5.33	6.45	4.32
0	280	4.03	5.16	6.93	4.90
50	0	2.96	2.80	2.39	1.85
50	140	4.12	6.77	7.70	5.99
50	280	4.38	7.37	9.87	7.83
100	0	2.88	2.93	2.15	2.24
100	140	4.24	6.46	8.17	6.88
100	280	5.02	7.06	12.30	9.04

Severe P deficiency symptoms were observed for all treatments in which K was applied. The corn plants had stopped growing the last week before harvest and by that time, most of the leaves had turned purple. The 0 K treatments had much smaller plants, as indicated by the yield values, and all plants were yellow, indicative of K deficiency. These observations were confirmed by plant tissue analyses for P and K as given in table 3.

The concentration of K was slightly below normal for the 0 K treatment, but it was within the normal range for higher K rates. The corn plants responded to applied K, and it did not appear that an interaction existed between P and K as was observed for forage yields reported in table 2.

Since few positive conclusions could be drawn from the first greenhouse study, the second experiment was developed, the data of which are given in the following section.

Greenhouse Experiment II

Data for biomass produced from this study are given in table 4. The first harvest date had the greatest biomass, especially for the 100% and 50% fly ash treatments, as compared with subsequent harvests. The biomass was about 2 times that for later harvests. There was a trend for increased biomass as a function of sewage sludge rate for the first harvest, but this did not seem to be the case for later harvests. It is unlikely that N could have been limited in any growth period.

Table 3. P and K levels of corn forage in response to P and K additions to fly ash from the old pond.

Treatment P	Treatment K	Harvest Date 1	2	3	4
		P in plant,%			
0	0	0.16	0.24	0.30	0.28
0	140	0.10	0.10	0.13	0.10
0	280	0.09	0.10	0.13	0.10
50	0	0.16	0.21	0.42	0.38
50	140	0.09	0.10	0.13	0.11
50	280	0.10	0.10	0.13	0.11
100	0	0.18	0.25	0.43	0.40
100	140	0.10	0.10	0.14	0.16
100	280	0.09	0.10	0.12	0.12
		K in plant,.%			
0	0	1.04	1.39	2.24	1.09
0	140	2.36	3.77	4.16	3.71
0	280	3.87	4.42	5.59	4.93
50	0	1.00	1.42	1.69	1.02
50	140	2.63	3.48	3.10	4.62
50	280	3.73	4.35	4.68	4.60
100	0	0.97	1.53	1.89	1.13
100	140	2.45	3.57	2.80	3.94
100	280	3.67	4.23	4.28	4.77

Table 4. Effect of adding sewage sludge to mixtures of fly ash and subsoil on biomass production of corn.

FA[5]	SS[6]	Harvest Date 1	2	3	4
%	Mg/ha	g/plant			
100	0	1.7	0.6	0.5	0.4
100	0	1.9	1.0	1.0	0.6
100	0	2.0	1.8	1.3	0.6
75	15	1.4	1.3	1.0	1.2
75	15	2.1	1.2	1.2	0.9
75	15	2.2	1.1	1.1	0.7
50	30	1.5	1.2	1.7	1.5
50	30	2.5	1.3	1.5	1.1
50	30	2.4	1.1	1.7	1.2

[5] Percent fly ash with remaining being subsoil.

[6] Sewage sludge equivalent in terms of megagrams per hectare

As the percentage of soil in the fly ash mixture increased, there was a trend for the biomass to increase for each comparable sewage sludge treatment. This effect was more pronounced between the 0% and 25% soil treatments and was observed for all four growing periods.

The effects of sewage sludge and subsoil additions on the percent P and K of corn grown on fly ash are given in table 5. The addition of sewage sludge to fly ash increased the percent P and K in the plant tissues for all harvest dates and for all fly ash and soil mixtures. The magnitude of this trend diminished with harvest date. This is in spite of the fact that extractable P was high for all mixtures. All treatments had >224 kg/ha P based on the Mehlich III soil test (data not shown).

The level of K decreased in most cases between the first and second harvest, especially for the two fly ash and soil mixtures. As a result, K fertilizer was added to each pot between the second and third growing periods. This had an effect of increasing the level of K in the plants. Although K concentrations in the plant tissue would not appear to be limiting biomass production for any of the treatments, the concentration of this element changed over time to a greater degree than all others measured. Data for Ca, Mg, N, and several trace elements are not given since none of these varied significantly as a result of any treatment.

Table 5. Effect of adding sewage sludge to mixtures of fly ash and subsoil on P and K levels of the plant tissue.

FA	SS	Harvest Date 1	2	3	4
%	Mg/ha	P in plant, %			
100	0	0.12	0.14	0.11	0.11
100	0	0.15	0.16	0.13	0.15
100	0	0.20	0.16	0.13	0.18
75	15	0.13	0.27	0.14	0.15
75	15	0.19	0.18	0.14	0.14
75	15	0.18	0.22	0.14	0.15
25	30	0.13	0.29	0.13	0.15
25	30	0.14	0.24	0.14	0.16
25	30	0.18	0.20	0.14	0.15
		K in plant, %			
100	0	1.35	1.78	2.35	1.88
100	0	1.57	1.69	2.34	1.70
100	0	1.81	1.89	2.14	1.99
75	15	2.00	0.87	2.30	1.55
75	15	2.11	1.42	2.42	1.58
75	15	2.09	2.11	2.46	1.83
25	30	2.35	1.06	1.42	1.36
25	30	1.86	1.26	1.40	1.55
25	30	2.34	1.61	1.82	1.56

Conclusions

Knowledge gained from the first greenhouse study indicated that P may be limited for plant uptake even when data from either Bray-1P or Mehlich III extractions would indicate adequate levels should be in the high range. A three-fold difference between these two extractants was observed. The amount of P that corn could remove from fly ashes was reduced when K was applied.

Data from the second greenhouse experiment were encouraging that fly ash could be directly vegetated. The use of sewage sludge or some other organic materials is likely to be necessary to serve as a slow release N fertilizer. It is also likely that soil will be mixed with the fly ash to provide more cation exchange capacity and plant nutrients. It appears that sewage sludge and soil has P in a more available form than that found in fly ashes.

Although corn was used as a test crop, it is highly unlikely to be grown on the site, hence other grasses such as Kentucky 31 will need to be tested. It would appear the fly ashes at this site could be directly vegetated without the 4-ft soil cover. It has been proposed that a field study be established to verify these conclusions.

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INFLUENCES OF HUMIC AND FULVIC ACIDS AND ORGANIC MATTER ON LEACHATE CHEMISTRY FROM ACID COAL SPOIL[1]

Claude E. Gentry,[2] Gerald S. Walton,[3]
Walter H. Davidson[4] and Gary L. Wade[5]

Abstract: Column-leaching experiments were conducted on an acid pyritic coal spoil to determine the influence of acid rain, humic acid (HA), fulvic acid (FA), and undecomposed organic matter (OM) on pH and Al, Fe, Mn, and SO_4 concentrations in the spoil leachate and on the spoil. Simulated acid rain of pH 4.0 was applied for 50 weeks under laboratory conditions to spoil columns modified with 0.5% FA or HA, or 2.0% OM from four forest trees and two herbs. Quality-control methods were used to evaluate treatment effects. Addition of HA and tall fescue leaf material to a Lily, KY, spoil created a greater and longer lasting desirable effect on leachate pH and Al, Fe, Mn, and SO_4 than additions of FA or OM of five other species. Results suggest that revegetation resulting in rapid production of matured soil OM may reduce the amount of some ions commonly leached from acid mine spoils.

Additional Key Words: simulated acid rain, quality-control statistics

Introduction

Acid atmospheric deposition probably has increased the rate of cation and anion dissolution and leaching in many strip mine spoils (Gentry et al. 1991), mine tailings (Doepker 1988), and soils (Bergkvist 1987). Neutralization by soils or mine soils of acids (H_2SO_4, $HN0_3$, and HCl) from the atmosphere can help to prevent acidification of streams, ground water, capillary water, or surface runoff and creation of heavy-metal concentrations that are toxic to fish (Sharpe et al. 1984), agricultural crops (Irving 1983), and microorganisms (Miller et al. 1991). Soil lacking natural buffering by alkaline earth carbonates and low-intensity management may be sensitive to acid rain (Reuss 1983). Therefore, surface mine spoils low in carbonates and with or without OM produced by reclamation and pioneer species warrant study with regard to acid rain.

Most mined areas in Appalachia have been revegetated naturally or through reclamation, but not all have been revegetated effectively. Many poorly vegetated surface mine spoils in the Appalachian area have extremely low pH (Davidson et al. 1988) and high concentrations of available Al (Berg and Vogel 1973) and Fe (Cummins et al. 1965).

This study was designed to investigate the effects of sources of raw and "matured" OM on the quality of leachates produced by simulated acid rain from an acid mine spoil and the composition of the spoil.

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[2]Claude E. Gentry, Professor, Department of Agriculture, Berea College, Berea, KY, USA

[3]Gerald S. Walton, Statistician, U.S. Department of Agriculture, Forest Service, Hamden, CT, USA

[4]Walter H. Davidson, Research Forester, U.S. Department of Agriculture, Forest Service, University Park, PA, USA.

[5]Gary L. Wade, Ecologist, U.S. Department of Agriculture, Forest Service, Burlington, VT, USA

Material and Methods

Spoil

An extremely acid (pH 3.8, 1:2 spoil-water) silty clay, pyritic spoil material from an unreclaimed surface mine that was mined in 1964 near Lily, KY, was selected for study. It had an electrical conductivity (EC) of 171.1 dS/m and a catgion-exchange capacity.(CEC) 32.4 of cmol/kg. Exchangeable ions included Al, Fe, Mn, and SO_4 at 223, 204, 141, and 4,700 mg/kg, respectively. Organic matter totaled 1.32%, HA was 0.38%, and FA was 0.27%. The spoil material was air-dried and passed through a 2-mm sieve. It was then well mixed according to the accepted procedures of Sobek et al. (1978).

Simulated Acid Rain

Simulated acid rain leaching solutions were based on the average chemical composition of natural precipitation in Berea, KY, from 1985 through 1990. The chemical composition of the simulated acid rain is NO_3 (0.836 mg/L), SO_4 (1.262 mg/L), Ca (0.186 mg/L), Mg (0.028 mg/L), K (0.156 mg/L), Na (0.176 mg/L), NH_4 (0.216 mg/L), and PO_4 (0.012 mg/L). It was acidified to pH 4.0 with HCl. Owing to the instability of the simulated acid rain, it was prepared with freshly distilled water at 21°C and applied to the soils within 2 h.

Effects of Fulvic and Humic Acids on Leachate Chemistry

To investigate the effects of humic and fulvic acids, 1.6 kg lots of Lily spoil were carefully mixed with 8 g of FA or HA (5 g/kg or 0.5%) and then passed through a soil splitter 25 times to ensure an even mix. The FA and HA were extracted from a Pennsylvania revegetated mine soil (Gentry et al. 1993) and purified by the method of Schnitzer (1982). Control spoils received no organic acids. The amended spoils were placed into 20.3-cm-long, 10.4-cm-diameter polyethylene leaching columns of spoil packed (avoiding stratification) to a bulk density of approximately 1.45 g/cm^3, and retained by an inert fiberglass mat at the bottom of each column. Columns were placed inside a Buchner funnel on a leaching rack. Three columns were prepared for each treatment.

Weekly, each column was brought to field capacity with distilled water, then irrigated with 200 mL freshly prepared pH 4.0 simulated acid rain (2.54-cm equivalent) from a 250-mL plastic bottle at a constant drip rate of 50 mL/h. Spoil columns were treated every Thursday for 50 weeks beginning on August 13, 1991, and allowed to drain freely. Leachate water from each column was collected in a new sterile polyethylene bottle every other week. The collected leachate was analyzed for pH and SO_4 immediately, and within 36 h for the other constituents.

Effects of Organic Matter Source on Leachate Chemistry

Leaf materials were collected at Lily on August 5, 1991, from six naturally established plant species growing in the immediate area surrounding the unvegetated spoil source--tall fescue (Festuca arundinacea Schreb.), pokeweed (Phytolacca americana L.), black cherry (Prunus serotina Ehrh.), Virginia pine (Pinus virginiana P. Mill.), red pine (P. resinosa Ait.), and white pine (P. strobus L.). The leaves were rinsed with 0.01M HCl solution and distilled water, dried for 48 h at 68°C in a forced-air oven, and ground to pass through a 2-mm screen. The ground tissue was mixed by hand and passed through a soil splitter.

Seven soil organic-matter treatments were set up: none (no OM), tall fescue, poke, Virginia pine, red pine, black cherry, and white pine. Thirty-two g of plant tissue were added to 1.6 kg of the Lily spoil and mixed, then mixed again by passing the material 25 times through a soil splitter to ensure an even mix of spoil and OM. Spoil from each treatment was added to a randomly assigned leaching column. For 50 weeks beginning on August 13, 1991, each column received weekly applications of pH 4.0 simulated acid rain as described earlier.

Spoil and Leachate Analysis

Spoil material was analyzed in the soil laboratory at Berea (KY) College. It was air-dried, passed through a 2-mm sieve to separate coarse fragments, and analyzed for particle-size distribution by the hydrometer method. Spoil pH was determined in a 1:2 soil-water mixture with a glass electrode. Soluble salts were determined by measuring the EC of a 1:5 soil-water extract (Bower and Wilcox 1965). Organic carbon content was determined by the Walkley-Black Method (Nelson and Sommers 1982); FA and HA were determined by the method of Schnitzer (1982). Cation-exchange capacity was determined by the Gillman method for acid soils (Rhoades 1982). Aluminum was determined colorimetrically (Barnhisel and Bertsch 1982), total Fe by the O-phenanthroline method (Olson and Ellis 1982), and Mn by the periodate oxidation method (Gambrell and Patrick 1982). Sulfate was measured turbidimetrically (Rhoades 1982).

All leachates were analyzed immediately for pH by the potentiometric method (EPA 150.1) and for sulfate by the Turbidimetric Method (EPA 300). Aluminum, total Fe, and Mn were determined by the colorometric procedures described for spoils.

Statistical Methods

The experimental design is completely randomized except for weekly treatment effects on leachate composition. One-way ANOVA was used to evaluate effects that were measured at the end of the study period. Treatment effects on leachate quality could not be analyzed by ANOVA because serial observations were not independent. Time series methods, though effective for describing time dependence, are poorly suited for evaluating treatment effects. Instead, we adapted Shewart control-chart methods that have long been used to study industrial processes.

A process is "in control" if the fluctuation of means of a control variable, plotted over time, is "reasonably explained" by the "inherent variablity" of the observations. Inherent variability (homogeneity of variation is assumed) is based on the variation between sample units taken from the process at the same time, which is analogous to our "between replicates" variation, for which we used the square-root transformation on all variables but pH to impose homogeneity of variance. By using the difference between the treatment and the control, we ensured that observations would fluctuate about zero in the absence of treatment effects. As with conventional hypothesis testing, Shewart control charts are chary of claiming that a process is "out of control" without substantial evidence. (A 1/20 type II error rate prevails for processes disturbed by normally distributed white noise.) However, the control-chart criteria include trend influences; a process is considered out of control when eight consecutive observations are on the same side of the center line, as well as when the observation deviates greatly (three standard errors) from the control line. The criteria for control using these X-bar Shewart control charts are found in the quality control literature (Western Electric 1958).

For an example using the data from this study, see figure 1 in which the initially large and negative influence of HA on leachate Fe is evident (the 'x' points "out of control"); this influence diminishes up to week 40, where it becomes questionable and data points '.' are "in control."

Results

Effects of Fulvic and Humic Acids on Leachate Chemistry

Additions of HA, FA, and fresh OM caused pH, Al, Fe, Mn, and SO_4 leachate concentrations to deviate from the control conditions, but the results were not uniform. Plots of FA and HA treatment means versus controls are shown in figure 2, and fresh OM treatment results are similarly shown in figure 3. Table 1 summarizes the OM treatment effect on column leachate and specifies in which time periods the treatment effects were outside the limits of the control.

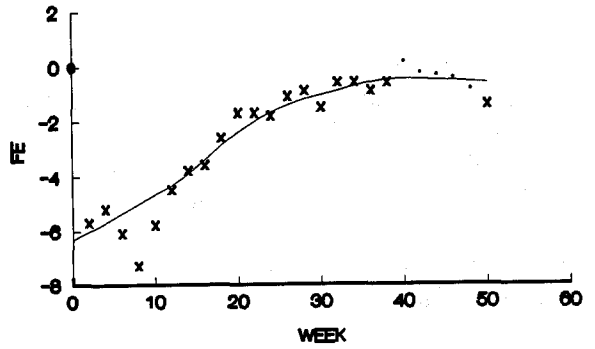

Figure 1. A Shewart control chart showing leachate Fe response to HA in spoil.

In general, HA had a greater and longer lasting beneficial effect on pH, Al, Fe, Mn, and SO_4 of mine spoil leachates than FA. Of the raw additions of organic matter, tall fescue had the greatest and longest lasting beneficial effect on leachate quality.

Changes in Spoil Organic Matter Over Time

Addition of FA resulted in FA fraction that was still significantly above that of the control spoil 50 weeks later (table 2). Addition of HA resulted in significantly higher HA and total OM at the end of the study.

Addition of raw OM resulted in an increased FA fraction in spoils in all treatments, increased HA for all but poke and white pine, and increased total OM in all but poke (table 3).

Discussion

The mine spoil used in this study was not sterilized so transformations of fresh OM, HA, and FA took place. Losses of OM indicate a significant amount of respiration loss of carbon as CO_2, transformation by soil microbial populations, and leaching. The high amount of variability within and among fresh OM addition treatments is likely due to successive transformations of fresh OM into various forms at different rates, which made leachate-chemistry responses complex.

That some fractions of OM are efficient in sequestering metal ions and some anions is well known.

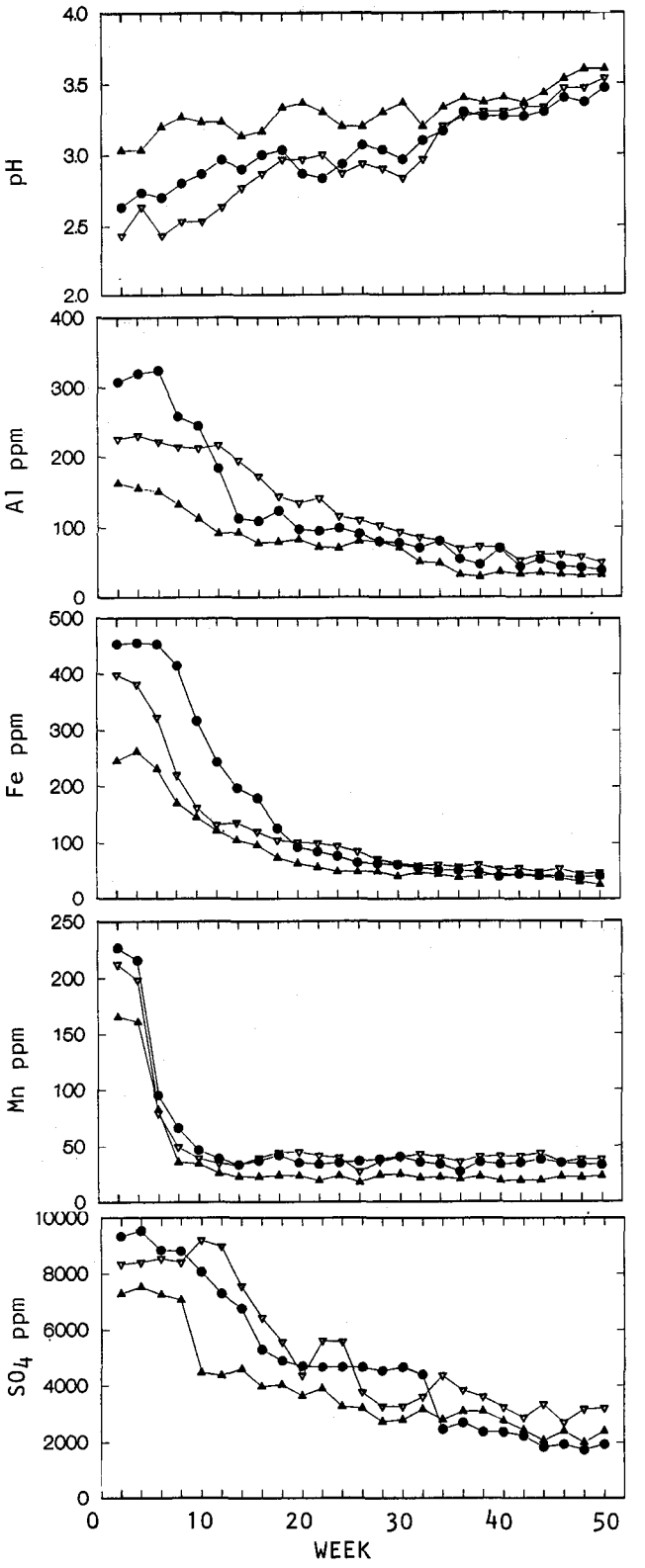

Figure 2. Effects of FA and HA on spoil leachate chemistry. ●-●-control, ▽-▽ FA, △-△ HA.

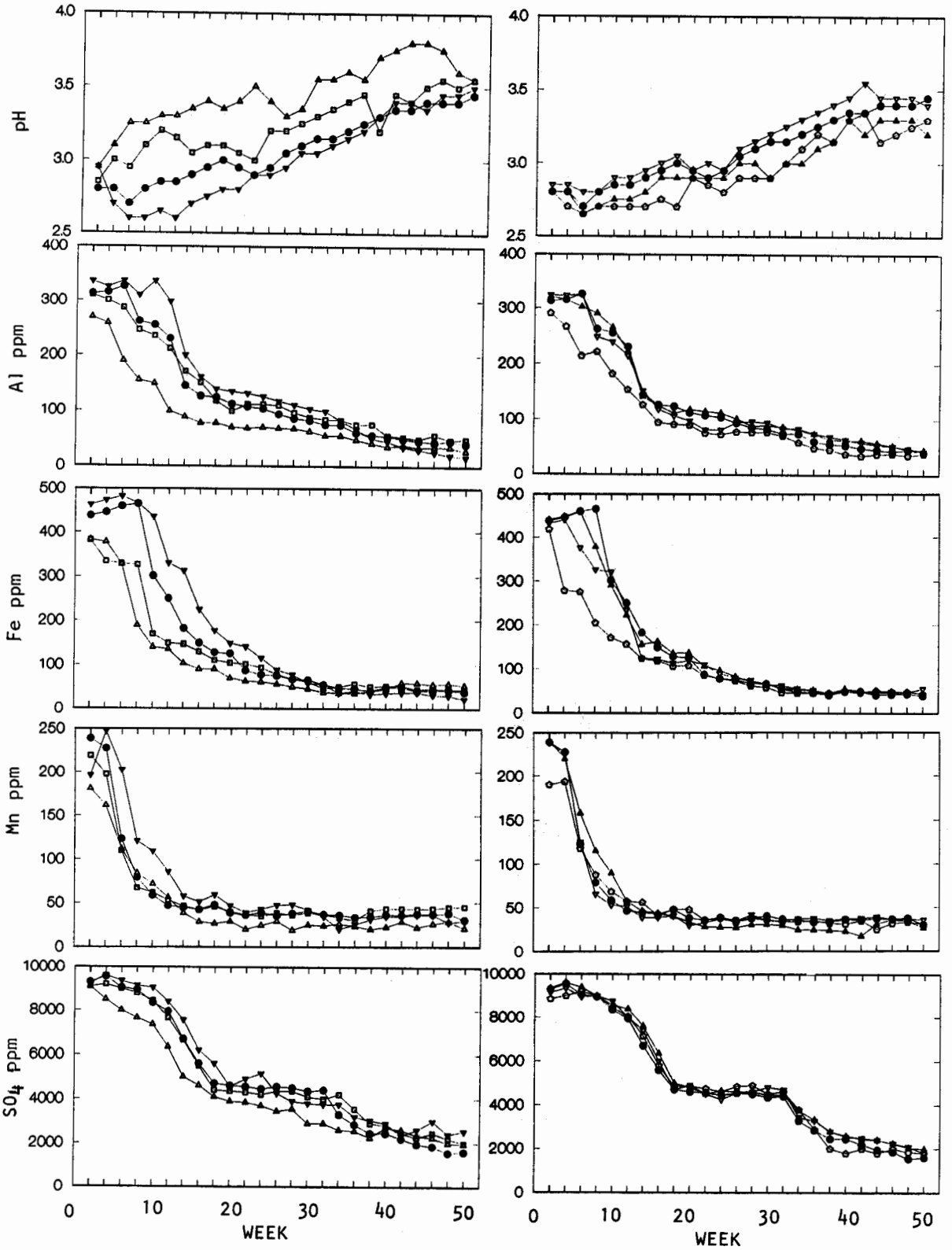

Figure 3. Effects of raw OM on spoil leachate chemistry. control, fescue, poke, black cherry, Virginia pine, red pine, white pine.

Table 1. Organic matter treatment effects on ions in column leachate. Values in this table are the weeks in which the treatment effects were outside the control limits. Underlined values were above the control; values not underlined were below the control.

Treatment	pH	Al	Fe	Mn	SO_4
Fulvic acid	2-18	2-10	2-18	4-14	2-6
	22	12-50	26-30	20-24	10-20
	28-32		34-50	26	22-24
				34-44	34-50
				46	
				48-50	
Humic acid	2-50	2-50	2-40	2-50	2-32
			52		38-50
Fescue	2-50	2-50	2-40	2-4	4-38
			44-50	12	44-50
				14-50	
Poke	2	8-36	10-28	2	10
	4-20	42-50	40-42	6-30	14
	30-36		46-50	34	16
				46-50	24
					28-32
					34-50
Black cherry	4-36	2	4-20	4-8	24-32
	38	6	24	40-50	34-50
	40-50	10-12			
		14-16			
		18-20			
		26-50			
Virginia pine	10-20	8	8	6-12	14-20
	30-50	28-50	14	24-50	36-50
			26-50		
Red pine	10-30	18-24	6	20	38-50
	36-48	32-50	8	22	
	50		18		
			20		
			24		
White pine	2	4-50	2-22	4	2
	8-40			6	22-36
	44-50			14-16	38-44

Table 2. Organic matter fractions in Lily spoil containing 0.5 percent added ferulic acid or humic acid after 50 weeks of leaching with pH 4.0 simulated acid rain. Values are means of 3 replications. Standard errors of means are in parentheses under controls.

	Fulvic Acid %	Humic Acid %	Total O.M. %
Initial condition	0.27	0.38	1.32
Treatment			
Control	0.19	0.35	1.21
	(0.02)	(0.02)	(0.04)
+ FA	0.29*	0.36	1.34
+ HA	0.22	0.48*	1.49*

*Significantly different from the control, P = 0.05, using Bonferroni's inequality applied to t-tests in the ANOVA.

Table 3. Spoil OM fractions in Lily mine spoil containing 2 percent added OM of different species after 50 weeks of leaching with pH 4.0 simulated acid rain. Values are means of 2 replications with 2 analyses per replication. Standard errors of means are in parentheses under controls.

	Fulvic Acid %	Humic Acid %	Total O.M. %
Initial condition	0.27	0.38	1.32
Treatment			
Control	0.20	0.33	1.19
	(0.02)	(0.03)	(0.06)
Fescue	0.38*	0.78*	1.71*
Poke	0.45*	0.36	1.36
Black cherry	0.39*	0.41*	1.35*
Virginia pine	0.53*	0.46*	1.86*
Red pine	0.47*	0.46*	1.66*
White pine	0.57*	0.45	1.79*

*Significanlty different from the control, P = 0.05, using Bonferroni's inequality applied to t-tests in the ANOVA.

Decreasing treatment effects over time may be due to saturation of OM exchange sites, microbial or transformation of active OM fractions and leaching of organic compounds.

Treated spoils and leachates both showed marked similarities and differences in overall results. Leachate pH generally increased in all treatments throughout both studies. Concentrations of Al, Fe, Mn, and SO_4 in soil leachates showed marked decrease over time. At the end of the study, pH and Al, Fe, Mn, and SO_4 leachate concentrations were virtually the same in all treatments in both studies. Additions of FA and HA to the mine spoil generally had immediate effects on leachate chemistry, which dissipated gradually. By contrast, effects due to addition of fresh OM took time to develop.

There were marked differences in the cumulative effects of treatments in both studies (table 4). Addition of HA and fescue leaf material significantly decreased total amounts of Al, Fe, Mn, and SO_4 leached from the spoils. The sums of ions leached from the fescue treatment are similar to those of the HA treatment, and the poststudy HA fraction of the fescue treatment was elevated well above that of other treatments. This suggests that fescue material may have been rapidly converted to the HA class of organic matter.

Among the three angiosperms (fescue, poke, and black cherry), there was a general inverse relationship of leachate pH and concentrations of the other ions. There was a less obvious difference between treatments among gymnosperms (pines). Different amounts of acidity produced during decomposition of various plant materials may reinforce or counteract the ion-complexing roles of some OM fractions in soils.

Table 4. Milligrams of Al, Fe, Mn, and SO_4 leached from 1.6 kg of Lily spoil with a variety of organic matter treatments over a 50 week period. Standard errors of the means are in parentheses.

Treatment	Al	Fe	Mn	SO_4
Raw O.M.				
None	1291	1510	593	48700
	(14)	(14)	(9)	(402)
Fescue	869*	1063*	455*	42460*
Poke	1453*	1743*	678*	52100
Black cherry	1310	1226*	585	49280
Virginia pine	1340	1499	583	51580
Red pine	1292	1411	580	50700
White pine	1023*	1113*	562	49340
Mature OM				
None	1228	1496	543	48280
	(33)	(11)	(11)	(277)
HA	750*	859*	366*	38547*
FA	1276	1214*	540	52453*

*Significantly different from the control, P = 0.05, using Bonferroni's inequality applied to t-tests in the ANOVA.

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RECLAIMING COARSE TACONITE TAILING WITH MUNICIPAL SOLID WASTE COMPOST[1]

Glenn D. Melchert, A. Paul Eger,
Zena Kassa, and Steven W. Dewar[2]

Abstract: Coarse-textured taconite tailing cover about 3,000 ha on Minnesota's Mesabi Iron Range. These tailing are deficient in nutrients, cation-exchange capacity, and moisture-retention capacity and are highly erodible. Previous reclamation efforts, in most cases, have been unable to meet the regulatory requirement for 90% vegetative cover after 3 yr. Small-scale test plots have demonstrated that municipal solid waste compost can be used to successfully reclaim tailing areas, which also provides a use for the compost. There has been a general reluctance to use municipal compost for reclamation activities, however, owing to concerns relating to the release of metals and organics. A field demonstration study was initiated to examine the feasibility of applying the compost on a larger scale and to examine its impacts on water quality and runoff from the reclaimed areas. Three 0.1-ha plots were located on the south-facing slope (15% slope) of a coarse tailing dike. Three treatments were studied: (1) a control with no treatment, (2) standard diammonium phosphate fertilizer application of 448 kg/ha, and (3) a compost treatment with 224 kg/ha fertilizer and 44.8 Mg/ha (dry weight) of municipal solid waste compost. All plots except the control were seeded with a grass-legume mixture and mulched with hay. Runoff collection plots were set up, and lysimeters were installed in each plot. Three months after planting, ground cover was 67%, 39%, and 0% for the compost, standard fertilizer, and control plots, respectively. Biomass, as kilograms per hectare, was 976, 225, and 0 for the compost, standard fertilizer, and control plots, respectively. Chloride, sulfate, calcium, magnesium, sodium, and nitrate were elevated in the soil water of the compost plot. Nitrate was the only chemical that exceeded water quality standards; however, these concentrations are not expected to significantly impact the local aquifer because the compost is only applied once and the concentrations of all parameters decreased with time.

Additional Key Words: water quality, heavy metals, nutrients, water balance, biomass, percent cover, erosion, Minnesota.

Introduction

Across the Mesabi Iron Range in northeastern Minnesota there are over 10,000 ha of land covered with taconite tailing waste from eight taconite mining operations. Portions of this tailing, especially the dikes, are comprised of relatively coarse-textured material, which covers over 3,000 ha. State mineland reclamation rules require that these areas be vegetated and achieve a vegetative cover of 90% after three growing seasons, except for slopes that face south or west, where a limit of five growing seasons applies (MNDNR 1980). To date, companies have found it particularly difficult to meet the 3- and 5-year requirements on coarse-textured tailing.

Coarse tailing is low in organic matter, nitrogen, phosphorus, cation-exchange capacity, and electrical conductivity (Noyd et al. 1992). Coarse-tailing slopes that are sparsely vegetated or bare often appear droughty and exhibit extensive gullying. The gray color of the tailing results in large temperature extremes, especially on slopes that face south or west, and the large open expanses typical of these basins subject these areas to considerable wind

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[2] Glenn D. Melchert, Hydrologist, A. Paul Eger, Principal Engineer, Zena Kassa, Research Analyst, and Steven W. Dewar, Mineland Reclamation Specialist, Minnesota Department of Natural Resources, Division of Minerals, St. Paul, Minnesota.

erosion. These factors all contribute to the difficulty in vegetating coarse tailing.

Recent plot studies have shown that organic amendments, such as municipal solid waste (MSW) compost, will improve soil structure and substantially increase the growth of grasses and legumes on coarse tailing compared with present standard reclamation practices (Norland et al. 1991, 1992). The use of compost may be a viable approach to meet reclamation standards at problem sites. Municipal solid waste compost slowly releases macro and micro nutrients, increases cation-exchange capacity and water-holding capacity, complexes heavy metals, and increases microbial populations in the tailing (Veith and Norland 1992, Wirth 1991, Heil and Barbarick 1989, Stark 1987).

The research base necessary for environmentally acceptable use of composted municipal solid waste on mineland reclamation projects in Minnesota is very limited. Many potential compost users have expressed concern about the levels of contaminants in the composts. Although landfill materials are transformed into a useful soil amendment by composting, potentially hazardous levels of trace metals may be introduced into the soil environment and food chain (He et al. 1992). MSW compost products are inherently variable (Walker and O'Donnell 1991), and potential environmental impacts must be evaluated before they are used extensively.

The objectives of this project were to determine: (1) the feasibility of applying MSW compost to a large area, (2) the effectiveness of MSW compost applications in meeting vegetative cover standards, and (3) impacts on water quality that may occur at these sites through the use of municipal solid waste compost as a soil amendment.

Site Description

The study area is about 11 km (7 miles) west of Hibbing in northeastern Minnesota (93°2' N latitude, 47° 22' W longitude). The plots are located on a coarse-tailing dike within the National Steel Pellet Co. tailing basin. Taconite tailing are the waste products from the processing of taconite rock to produce iron ore. Tailing particles range in size from fine gravel to clay and consist mostly of silica (Myette 1991). In general, at the study site, 25% of the material is coarse sand or larger (>2.0 mm), while about 8% of the material is silt and clay (<0.075 mm). The tailing is pumped as a slurry to a tailing basin, where the solids settle and the water is reused. Depositional processes occurring in the tailing basin are similar to those occurring with the formation of deltas, with the coarsest particles settling nearest the spigot while the finer materials move into the basin interior to settle out of suspension. The slurry lines are periodically lengthened as the tailing builds up. Coarse tailing is usually utilized to construct a dike, which encircles or subdivides the basin.

During installation of the lysimeters, it was observed that much of the coarse tailing consists of interbedded coarse and fine particles. These layers were typically less than 2 cm thick and sloped away from the center of the dike.

This area experiences short, mild summers and long, cold winters with monthly averages ranging from -12.3°C in January to 18.1°C in July and a mean annual temperature of 4°C. Mean annual precipitation is 688 mm, of which about 510 mm is lost through evapotranspiration (Myette 1991).

Materials and Methods

MSW Compost

The compost used was produced by the East Central Materials Recovery/Compost Facility near Mora, MN and was classified as a class 1 material. The municipal waste it receives comes from households and small industry from a 5-county area with a mostly rural population of about 100,000. According to Minnesota's guidelines (Wirth 1991) a compost is class 1 if it is processed without sewage sludge, has composted for at least 180 days, and if contaminant levels do not exceed Minnesota Pollution Control Agency limits (table 1). A class 1 rating allows for

unrestricted application rates. Further details on the composting process can be found in Dewar and Norland (1994 in press).

The compost was sampled and analyzed for moisture content, organic and inorganic constituents, nutrient content, and trace metals. The entire suite of semivolatile and pesticide compounds, PCBs, and the trace metals silver, arsenic, selenium, copper, nickel, zinc, mercury, cadmium, and chromium were analyzed according to EPA methods (SW846).

Table 1. Comparison of allowable contaminant concentration limits for class 1 MSW compost (dry weight basis) and sample results from the compost used in this experiment.

Contaminant	Concentration, mg/kg	
	Limits	Compost results
Arsenic	NAp	5.0
Barium	NAp	90
Cadmium	10	3.75
Chromium	1,000	58
Copper	500	285
Lead	500	175
Mercury	5	1.45
Nickel	100	25
Selenium	NAp	0.30
Silver	NAp	3.1
Zinc	1,000	785
PCB	1	<1

NAp indicates not applicable, no limits set.

Plot Establishment

A south-facing slope of an unvegetated coarse tailing dike was selected because this aspect is the most difficult to vegetate successfully. This dike was graded to fill the gullies and to form a uniform slope. The final slope averaged 15%.

Three 0.1-ha plots (80 m by 12.5 m) were established (fig. 1). One plot was a control that was left bare, one was vegetated using reclamation methods typical for the industry (standard reclamation), and one had compost and fertilizer applied. No treatments were applied to the control plot. This plot was disced, however, so its surface would be similar to that of the other plots.

The standard reclamation plot received an application of 448 kg/ha (400 lb/acre) of granular diammonium phosphate fertilizer (18-46-0). The fertilizer was applied with a broadcast spreader and then incorporated into the tailing to a depth of 15 cm with a disc. Then the plot was seeded at a rate of 56 kg/ha (50 lb/acre) with a grass-legume cool-season mix with a broadcast spreader. After seeding, the plot was immediately mulched with hay at the rate of 2.2 Mg/ha (2 t/acre). The mulch was then crimped to a depth of about 7.5 cm.

The MSW compost plot received an application of 44.8 Mg/ha (20 t/acre) of municipal solid waste compost and 224 kg/ha (200 lb/acre) of granular diammonium phosphate (18-46-0). The compost was applied with a standard rear-throw manure spreader. The fertilizer was applied with a broadcast spreader, and then the compost and fertilizer were disced to a depth of 15 cm. Fifty-six kilograms per hectare of the grass-legume seed mix and 2.2 Mg/ha of hay mulch were applied and incorporated in the same way as for the standard reclamation plot.

Prior to vegetating, a backhoe was used to bury three 0.26 m^2 pan lysimeters at a depth of 61 cm in each plot. Lysimeters were placed 20 m apart and 6 m from the high edge of the plots. After the plots were seeded, a 1.5- by 12.5-m (18.75 m^2) runoff plot within each plot was established to collect runoff water. Sets of suction lysimeters were installed near each pan lysimeter at depths of 15, 30, and 61 cm (fig. 1).

Water Quality

Water samples were collected from the runoff plots on a weekly and/or event basis. Total runoff and total sediment loss were measured for each plot. Samples were collected from the pan and suction lysimeters when sufficient quantities of water were available. Since the only water applied to these plots was from precipitation, the interval between sample collections varied. Water samples collected from the runoff plots were analyzed for total

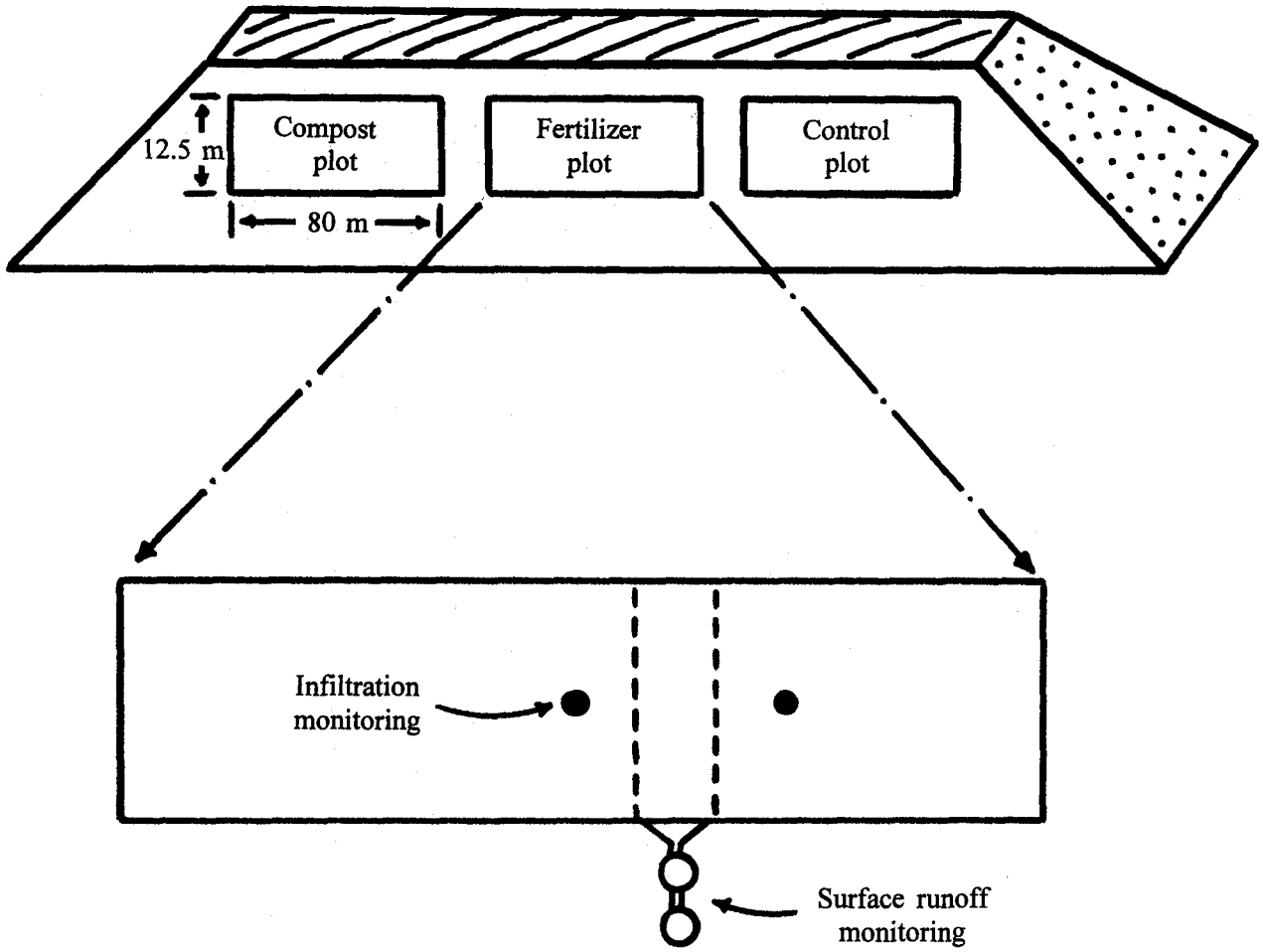

Figure 1. Test plot design and sample locations.

sediment, pH, specific conductance, major anions and cations, iron, and manganese. Selected samples were analyzed for heavy metals. Metals samples were filtered through a 0.45-μm filter prior to preservation with ultrapure nitric acid. The metals were either analyzed with flameless atomic absorption (EPA methods 206.2, 270.3, and 245.1) or ICP (EPA method 200.7). Water collected in the pan and suction lysimeters was measured and generally sampled for the same parameters as the runoff plots. Precipitation was measured on site with a weighing bucket rain gage.

Vegetation Sampling

In early August 1993, 3 months after planting, percent ground cover and biomass were measured on each plot. Percent cover was measured using a systematic point-quadrat sampling method (Raelson and McKee 1982) to estimate cover within a 95% statistical confidence. Biomass was estimated by clipping all aboveground plant material from six randomly selected 0.11-m^2 quadrats.

The two major plant species, timothy *Phleum pratense* and alfalfa *Medicago sativa*, were analyzed for nutrients and trace metals. Twenty randomly selected plants of each species for each treatment were collected and

composited. The entire above-ground portion of the timothy plants and the top 6 inches of each alfalfa plant were used. The plants were dried at 70°C and ground in a Wiley Mill to pass a 1-mm sieve. The dried plant mass was dry ashed and digested in 10% HCl, and the resulting solution was analyzed by ICP-AES (Inductively Coupled Plasma Atomic Emission Spectroscopy) according to EPA methods.

Results

Compost Contaminants and Application

Concentrations of all trace metals were below the maximum allowable limit for class 1 material. No pesticides or PCBs were detected, and the only semivolatile compounds detected were butylbenzylphthalate and bis(2-ethylhexyl)phthalate. The concentrations of these compounds were 5.1 and 41 mg/kg, respectively. Phthalates are commonly found in plastics, so these elements were most likely released from the small plastic particles present in the compost when the sample was heated during the extraction.

The use of farm equipment was reasonable for the application and incorporation of the MSW compost. The manure spreader distributed the compost rather evenly, but discing did not evenly incorporate the compost in the tailing. Most of the compost remained on or near the surface with only a small amount mixed to 15 cm. The disc, either while incorporating the compost or during crimping of the mulch, tended to concentrate the compost in narrow bands rather than uniformly distributing the material across the plot.

Vegetation

Percent cover and biomass were significantly higher in the compost plot than the fertilizer plot. Percent cover and biomass in the fertilizer plot were 39% and 225 kg/ha while these parameters were 67% and 976 kg/ha in the compost plot. The tissue analysis was set up as a screening procedure to determine if significant amounts of metals had accumulated in the plants. Table 2 compares the tissue concentrations from this study with sufficient (desired) concentrations for optimum growth (Jones et al 1991). Although zinc concentrations for both species in the compost plot were low, they were higher than the plants from the fertilizer plot. None of the trace metal concentrations were elevated above sufficient levels (Jones et al 1991).

Water Balance

Runoff was greatest in the control plot, and the bulk of it occurred during three storms in late June and early July. Rainfall events of less than about 2.5 cm and of low intensity generally did not produce runoff in any of the plots. Total runoff, as a percent of total precipitation of 58 cm, was 0.2, 1.5, and exceeded 4.5 for the compost, fertilizer, and control plots, respectively. For several large storms, the sediment load exceeded the capacity of the collection trough in the control plot and as a result some runoff and sediment was not measured.

Erosion was greatest in the control plot and minimal in the compost plot. Sediment loss for the year (May-October) was 0.7 Mg/ha, 2.5 Mg/ha, and 34.4 Mg/ha for the compost, fertilizer, and control plots, respectively.

Precipitation over the course of the study was only 7 cm above normal. For every month except July precipitation was normal or below normal. In July, precipitation was 13 cm above the normal 10 cm, with 15 cm falling over the first 11 days of July.

Evaporation from the plots was high, so there was little infiltration below the plant rooting zone and into the pan lysimeters. Of the 58 cm of rain that fell, only 1.9%, 1.8% and 1.5% infiltrated through the compost, standard reclamation and control plots, respectively. Nearly all of this occurred in early July after 10 cm of rain fell over 2 days.

Table 2. Concentrations of nutrients and metals in timothy and alfalfa from the study plots as compared to sufficient levels in the literature.

Parameters	Timothy			Alfalfa		
	Fertilizer plots	Compost plot	Sufficient[1]	Fertilizer plot	Compost plot	Sufficient[1]
Concentration, %:						
Phosphorus	0.25	0.19	0.11-0.18	0.25	0.19	0.26-0.7
Potassium	1.91	1.92	1.14-1.70	1.31	1.39	2.00-3.5
Calcium	0.22	0.24	0.09-0.35	2.38	2.68	1.80-3.0
Magnesium	0.23	0.11	0.06-0.25	1.06	0.42	0.30-1.0
Concentration, mg/kg:						
Aluminum	6.22	3.96	NA	38.5	14.0	NA
Boron	6.24	20.8	1-10	38.9	56.5	30-80
Cadmium	<0.12	<0.12	NA	0.16	0.34	NA
Chromium	<0.28	<0.28	NA	0.42	0.33	NA
Copper	3.49	4.96	7-45	3.52	6.69	7-30
Iron	168	35.8	22-54	326	152	30-250
Lead	<1.17	<1.68	NA	2.06	1.76	NA
Manganese	181	97.5	11-35	394	199	31-100
Nickel	1.22	1.03	NA	0.68	1.37	NA
Zinc	7.94	40.2	24-62	7.89	51.9	21-70

NA indicates not available.
[1]Values taken from plant analysis handbook by Jones and others (1991).

Water Quality

As a result of a low rate of infiltration, there was only sufficient water in the lysimeters for sampling on 2-3 occasions. Despite the limited data base, some differences between the treatments were observed. Water infiltrating through the treated plots had higher specific conductance than the control. While this difference was generally about 20%, water in one of the lysimeters in the compost plot had a specific conductance value that was more than three times that of any other site. This water also contained the highest concentrations of sulfate (950 mg/L), chloride (220 mg/L), calcium, magnesium, sodium, nitrate (140 mg/L), and most of the trace metals except for zinc (tables 3 and 4). Although trace metal concentrations were generally low and below water quality standards, concentrations in the compost plot were slightly higher than in the other two plots (table 4).

Discussion

Using MSW compost as a soil amendment appears to be a successful method for reclaiming coarse taconite tailing without creating any additional serious environmental impacts. The compost served as a mulch which minimized runoff, and increased moisture retention. The compost plot had a 67% ground cover after 3 months, compared with 39% for the fertilizer plot. Plants in the compost plot appeared taller and more closely spaced than those on the fertilizer plot. The larger biomass in the compost plot may be due to a combination of increased growth and increased germination. Since ground cover typically increases significantly in the 2 or 3 yr after planting, the compost plot appears to have a good chance at meeting the 90% cover requirement. Norland and Dewar (1994) have shown that flat coarse-tailing plots, amended with similar MSW compost and fertilizer

Table 3. Water quality summary of lysimeter samples from the study plots[1].

Parameters	RAL	Control plot		Fertilizer plot		Compost plot	
		Median	Range	Median	Range	Median	Range
S.C.		675	550-900	937	825-1,100	1,125	800-3,800
pH	6-8	7.72	7.46-7.84	7.54	7.49-8.07	7.44	7.16-7.80
Alk	NA	350	312-504	544	479-712	366	235-479
SO_4	250	16.8	2-51	26.2	2.0-44.5	287	122-950
Cl	250	4.6	1-18	3.0	1.0-5.2	21.0	12.0-220
Ca	NA	47.5	34-51	42.0	32.2-51.0	77.0	42.7-131
Mg	NA	74.8	56-100	110	82-140	180	94-220
Na	NA	11.5	9.3-17.7	20.2	14.8-22.4	51.0	29.6-260
K	NA	9.9	4.5-21.1	10.8	4.95-13.35	14.2	9.4-34.0
Fe	0.05	0.01	<0.01-0.02	0.02	0.01-0.07	0.02	0.01-0.29
Mn	0.6	0.01	<0.01-0.01	<0.01	<0.01-0.01	0.01	<0.01-0.06
NO_3-N	10	1.1	0.50-1.46	9.0	5.2-20.81	30.1	9.6-140
NH_3-N	NA	0.05	NAp	0.06	NAp	0.02	0.02-0.14
TKN	NA	<0.02	NAp	<0.20	NAp	1.23	0.20-1.36
Total P	NA	<0.02	NAp	0.03	NAp	0.03	0.02-0.05

NA indicates not available.
NAp indicates not applicable (only 1 sample was analyzed).
RAL indicates recommended allowable limits for drinking water contaminants in private wells (MN Dept. of Health 1991).
[1]Alkalinity and metals concentrations are in milligrams per liter, pH is in standard units, specific conductance (S.C.) is in microsiemens. The number of samples (n) represented by the ranges is between 4 and 8.

applications, achieved 90% cover after 4 growing seasons. It will be much more difficult, however, for the fertilizer plot to meet reclamation standards.

Even though compost applications have successfully increased vegetative cover in coarse tailing, there is some concern related to the uptake of the metals in the compost by vegetation. Taconite tailing generally have low trace metal concentrations and are deficient in micronutrients. The application of compost, with its higher metal content, may be beneficial to the vegetation. Our results indicate that there were no significant metal accumulations in the vegetation which suggests that the metal concentrations in the compost plot are not high.

In general, the application of MSW compost did not adversely affect the water infiltrating through the tailing. Although concentrations of major cations and anions were particularly high in one compost plot lysimeter, only sulfate and nitrate were above groundwater standards. The increase in concentrations is related to components that can be easily leached from the compost. Samples of compost which were extracted with water showed high concentrations of all major cations and anions. The difference in water quality among the lysimeters in the compost plot results from variability in the compost or uneven application of the material. Despite these elevated concentrations, the overall impact on groundwater in the area is expected to be small, since the amount of water moving down through the tailing is minimal.

Present reclamation costs for vegetating coarse tailing using standard techniques are about $500 per acre. The compost application used in this study was half the rate recommended by Norland and others (1992). Their

Table 4. Trace metal concentrations in test plot lysimeters compared to levels in MSW compost and groundwater in tailing area[1,2].

Parameters	Concentration, μg/L					
	RAL	Control plot	Fertilizer plot	Compost plot		Groundwater background[3]
				Median	Range	
Cadmium	4	<0.2	<0.2	0.2	<0.2-4	<1
Chromium	100	1.6	1.8	3.3	2.9-4.2	5
Lead	50	<2	<2	<2	<2-2	<10
Silver	20	<1	<1	5	4-16	NA
Arsenic	0.2	<4	<4	<4	<4	5
Barium	2000	<40	<40	<40	<40	60
Mercury	2	<0.2	<0.2	<0.2	<0.2	0.3
Selenium	20	<5	<5	<5	<5	<1
Copper	1000	<1	<1	10	8-18	<10
Nickel	100	<6	<6	15	13-27	3
Zinc	1000	<10	<10	20	13-30	<3

RAL indicates recommended allowable limits for drinking water contaminants (MN Dept. of Health 1991).
[1]Water analyses conducted on filtered samples.
[2]Samples represent composites of the three lysimeter samples. In the compost plot, each lysimeter was analyzed separately.
[3]Reported in Myette (1991) for tailing groundwater.

research showed, however, that this lower rate resulted in only 10% less vegetative cover than their recommended rate. By using the lower rate, the costs to transport and apply the compost are reduced by about 50%. Applying compost (once the material is on site) increases the cost by about 10% to 20% or $50 to $100 per acre. A major cost could be the shipment of compost to the site. If the compost is within 25 miles of the site, the hauling cost would be about $2.50/t. However, if the distance increased to 50 miles, the cost would double. Since the material contains about 50% moisture, the cost to apply 20 dry t/acre would be about $100 to $200 per acre, depending on the haul distance. Since mining companies have not had success in reclaiming the coarse tailing areas, they have had to revegetate these areas several times at an additional cost of up to $500 per acre each time. Despite these additional treatments, they have been unable to consistently meet reclamation standards for coarse tailing. The additional cost of a compost application may actually be the most cost-effective approach.

Conclusions

An application of 44.8 Mg/ha of MSW compost along with 224 kg/ha of diammonium phosphate fertilizer: (1) improved vegetative success as compared to the standard reclamation plot, (2) reduced erosion and runoff in the compost plot as compared to the standard reclamation plot, (3) resulted in acceptable trace metal concentrations in the vegetation, (4) resulted in water quality that was generally acceptable and improved with time, (5) is not expected to produce significant or lasting impacts on ground water quality because the compost is applied only once, and (6) may be the most cost-effective application rate for vegetating coarse taconite tailing.

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RECLAMATION OF PHOSPHOGYPSUM IN FLORIDA: EFFECTS OF AN OVERBURDEN CAP [1]

Steven G. Richardson [2]

Abstract: Phosphogypsum is a byproduct of phosphate fertilizer production, and more than 600 million metric tons have accumulated on 2,000 plus ha in Florida. Field research was initiated in late 1990 to compare the benefits of capping phosphogypsum with overburden (up to 15 cm in depth) from mined sites versus treatment of the phosphogypsum with minimal amendments. After two growing seasons, vegetation cover and production were equally good on plots either with or without an overburden cap as long as adequate N, K and Mg were supplied. However, more species became established with an overburden cap. Fluoride uptake by bermudagrass (*Cynodon dactylon*) was high when grown directly on phosphogypsum (895 mg/kg in leaf tissue) and was reduced slightly by a 15 cm overburden cap (670 mg/kg). Fluoride uptake was less in bahiagrass (*Paspalam notatum*) (215 mg/kg) and bushy beardgrass (*Andropogon glomeratus*) (460 mg/kg) than in bermudagrass (847 mg/kg) at another overburden-capped site. Unexpectedly, radium uptake in bermudagrass grown directly on phosphogypsum (0.6 pCi/g) was less than when grown on the overburden cap (1.8 pCi/g). The presence of grass cut the radon efflux from phosphogypsum in half (from 24 pCi/m^2-s to 11 pCi/m^2-s), while 15 cm of overburden, in addition to grass cover, halved it again (down to 5 pCi/m^2-s).

Additional Key Words: radium, radon, fluoride, bermudagrass, bahiagrass

Introduction

Phosphogypsum is a by-product of phosphoric acid production. For each kg of P_2O_5 produced, approximately five kg of phosphogypsum must be stockpiled or disposed. To date, more than 600 million metric tons have accumulated in Florida on 2000 plus ha, and phosphogypsum is continuing to accumulate at the rate of about 30 million metric tons per year. After phosphate rock has been reacted with sulfuric acid to produce phosphoric acid, the precipitated gypsum is removed on filters and is pumped in slurry form to an impoundment where it is allowed to settle. As the gypsum accumulates, a small dragline removes some of the gypsum for raising the height of the dikes. By this process the gypsum settling impoundment, or "stack" as it is often called, increases in elevation. As a stack grows in height (up to 60 m) the area of the settling impoundment decreases, until a point is reached where the pond capacity becomes too small and the pumping height requires too much energy. When phosphogypsum stacks reach the end of their useful lives, they must be closed and reclaimed. Reclamation will include a cover of vegetation to improve aesthetic qualities, to reduce erosion and improve the quality of surface runoff water, and to allow beneficial use of the land such as for wildlife habitat.

There are several problems associated with establishing vegetative cover on phosphogypsum, including (1) the residual acidity, (2) nutrient deficiencies and low nutrient-holding capacity, and (3) the tendency for caking and crust formation. In addition, there are other potential environmental problems. Phosphogypsum has elevated levels of fluoride and radionuclides, such as ^{226}Ra.

The primary problem for vegetation establishment is the acidity of the phosphogypsum. May and Sweeney (1980) observed pH values usually in the range of 2 to 3 in numerous samples, although samples from older, more weathered stacks had slightly higher pH values near 4. Bromwell and Carrier, Inc. (1989) reported a pH of 5.1 in

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[2] Dr. Steven. G. Richardson, Director of Reclamation Research, Florida Institute of Phosphate Research, Bartow, FL, USA.

a sample of phosphogypsum from an older, inactive stack. Phosphogypsum is comprised primarily of the slightly soluble neutral salt, calcium sulfate dihydrate. The acidity is due mainly to residual phosphoric and sulfuric acids, although hydrofluoric and fluosilicic acids are also formed during processing (Smith and Wrench 1984, Wrench 1987).

Laboratory, greenhouse and field studies, plus field observations, have indicated that plants could grow on phosphogypsum if the pH was raised to 4.0 or greater and if N, K, and Mg nutrients were supplied (Richardson 1993). Dolomitic limestone (also an Mg source) and phosphatic clay (also a K and Mg source) were effective neutralizing agents, while leaching with water was also an effective mechanism for reducing acidity. Leaching with rainwater is probably responsible for the higher pH values (4.7 to 5.3) found at the surface of older, inactive phosphogypsum stacks compared to pH values near 2 or 3 at active, acid-laced phosphogypsum stacks.

Fluoride concentrations in phosphogypsum produced from Florida phosphate rock often range from 4 to 7 g/kg (Rechcigl et al. 1992, Carter and Scheiner 1992, Oates and Caldwell 1985). Radium-226 commonly ranges from 20 to 30 pCi/g in central Florida phosphogypsum and from 9 to 15 in northern Florida phosphogypsum (Roessler et al. 1979, Mays and Mortvedt 1986, May and Sweeney 1980). The concern with fluoride is the possible development of fluorosis (a bone and tooth disorder) in grazing animals from consuming plants with high concentrations of fluoride. Radium-226 may also be absorbed by plants and be ingested by animals. In addition, ^{226}Ra gives rise to ^{222}Rn gas, which may diffuse out of the phosphogypsum into the atmosphere. There is also concern about the quality of surface runoff waters and leachate from phosphogypsum stacks.

This study is part of a research program with the primary objective of developing effective, but low cost, methods of establishing vegetation on phosphogypsum stacks when they are closed. One approach to coping with the chemical and physical properties of phosphogypsum is to cover it with overburden or soil. However, the question then arises as to how deep the soil or overburden cover must be to support an adequate vegetative cover. The deeper the soil cover, the more costly are the earthmoving operations and the greater are the impacts on soil borrow areas. This paper compares vegetation establishment (1) directly on phosphogypsum and (2) with modest depths (5 to 15 cm) of overburden cover. It also examines the effects of overburden capping on plant uptake of fluoride and ^{226}Ra and on the efflux of ^{222}Rn gas from phosphogypsum.

Methods

The effects of depth of overburden cover on vegetation establishment on phosphogypsum stacks were studied at two sites. The first site was on a nearly level (1% to 2% slope) area on top of the Estech Silver City stack, which had been inactive for about 25 years. The second site was on a 3:1 (horizontal:vertical) slope on the eastern side of the active IMC New Wales stack. Surface pH values of phosphogypsum at the Estech site were near 4.8, while surface pH values at the IMC site, which was underlain by a toe drain, were near 4.5.

Estech

On September 17, 1990, overburden cover was applied at the Estech site at depths of approximately 0, 5, 10 and 15 cm with the aid of a scraper, a dozer and a grader. To facilitate use of the scraper, the various overburden depths were applied in adjacent 6 m wide strips running east and west, with the shallowest depth strip to the north and progressively deeper covered strips to the south. Two treatments were superimposed: (1) one st dolomite per acre and (2) no dolomite. All plots were rototilled and disked. Individual plots were 3.6 m x 6 m, and the treatments were replicated four times. Fertilizer was broadcast at the per ha rates of 54 kg N, 13 kg P_2O_5, 27 kg K_2O, 5.6 kg Fe, 0.37 kg Mn, 0.34 kg Zn, 0.13 kg Cu, and 0.13 kg B on October 4, 1990. No Mg was applied in 1990 except as dolomite. Seed was broadcast on October 4, 1990 at the per ha rates of 11 kg weeping lovegrass (*Eragrostis curvula*), 22 kg hulled common bermudagrass, 22 kg Pensacola bahiagrass, 22 kg ryegrass (*Lolium perenne*), and 22 kg browntop millet (*Brachiaria ramosa*). Bahiagrass hay mulch was blown on the site at the rate of 4 Mg/ha, and the mulch was crimped-in with a cultipacker roller. On July 18, 1991, the plots were

fertilized at the per ha rates of 54 kg N, 13 kg P_2O_5, 27 kg K_2O, 4 kg Mg, 4 kg Fe, 1.5 kg Mn, 0.3 kg Cu, 0.27 kg Zn, and 0.1 kg B. It became apparent later in 1991 that the Mg rate was insufficient for good plant growth on phosphogypsum without dolomite or overburden (Richardson 1993). Therefore all plots were fertilized with 56 kg N, 28 kg K_2O and 14 kg Mg per ha in June 1992.

Percent cover was determined by the line-point method on the overburden depth plots at the Estech site on December 6, 1991, and September 21, 1992. Plant tissue of bermudagrass, bahiagrass and bushy beardgrass (a natural invader) were sampled from plots without overburden cover and with 15 cm of overburden cover in 1992 and analyzed for fluoride. Samples of overburden from the 15 cm depth plots, phosphogypsum from the plots without overburden, and phosphogypsum from a barren, untreated area were analyzed for pH, electrical conductivity, Mg, K, P, Mn, and Zn. Overburden and phosphogypsum samples with and without 2.2 metric tons/ha of dolomitic limestone amendment were also compared.

IMC

A study was designed to compare the effects of 0, 7.5 and 15 cm of overburden cover on vegetation establishment on a 3:1 slope at the IMC New Wales phosphogypsum stack. The three overburden depths were replicated three times on plots measuring 30 m downslope and 7.6 m along the contour. A site was cleared on June 20, 1991, on the east side of the IMC New Wales phosphogypsum stack. An existing bench, located immediately above the experimental site was widened to allow truck access and sloped to intercept surface runoff and route it away from the study plots. Dolomitic limestone was applied at the rate of 1.5 Mg/ha on plots that did not receive any overburden cover ("0" overburden depth) and the site was disked. Overburden application began June 24 and was completed on June 25, 1991. Overburden from a mined site south of the gypsum stack (site of the new stack expansion) was trucked and dumped on the bench and upper slope of the appropriate study plots. A D-3 dozer spread the overburden down-slope over the phosphogypsum, guided by a series of survey stakes marked with the proper depths (0, 7.5 or 15 cm). On the afternoon of June 25, 1991, the site was lightly scarified with a disk harrow and the overburden depth study plots were broadcast seeded with 29 kg common bermudagrass and 58 kg browntop millet per ha. Fertilizer was broadcast at the per ha rates of 78 kg N, 20 kg P_20_5, 39 kg K_20, 10 kg Mg, 2.4 kg Fe, 1.5 kg Mn, 0.5 kg Zn, 0.25 kg Cu, and 0.15 kg B. Bahiagrass hay mulch was blown on the overburden depth plots at 4 Mg/ha, and the hay was crimped in with a cultipacker roller and the dozer tracks. The plots were fertilized again on June 5, 1992 with 56 kg N, 28 kg K_20 and 14 kg Mg per ha.

Percent plant cover was determined on the overburden depth plots at the IMC site by the line-point method on December 5, 1991 and September 23, 1992. Samples of bermudagrass tissue were collected from the overburden depth plots at the IMC site on October 24, 1991, and were analyzed for ^{226}Ra activity, and for F concentration. Overburden from the 15 cm depth treatment, phosphogypsum from the plots without overburden, and phosphogypsum from a barren, untreated area were analyzed for pH, electrical conductivity, Mg, K, P, Mn, and Zn.

Radon-222 fluxes from the ground surface were measured in February, March and May 1992, with the aid of large area activated charcoal canisters placed for 24 hours on the grass-covered overburden depth plots and a barren (no vegetation or overburden cover) phosphogypsum "control" plot. In February the charcoal canisters on the grass-covered plots were simply pressed and twisted into the ground to achieve good contact with the ground, but the grass was not clipped. In an attempt to assure a better canister to soil seal in March and May, the grass was clipped to a height of approximately 4 cm, a canister was placed on the ground, and bermudagrass stolons (runners) and "soil" were cut with a knife around the perimeter of the canister. The canister was then pressed downward and twisted to attain good ground contact. On all sampling dates, canisters were pressed onto the surface of the barren phosphogypsum control plot, but because the ground surface was both hard and uneven, phosphogypsum was packed around the outside perimeter of the canisters to create a better seal. Additionally, in March and May a knife was used to scratch the ground surface around the perimeter of each canister to aid our efforts to press them into the ground on the barren plots. The ground surface beneath the charcoal-laden portion of each canister was not

disturbed. Following the 24-h period in the field, charcoal from each canister was removed, placed in a sealed container, and sent to the Florida Health and Rehabilitative Services laboratory in Orlando for radon flux determination.

Results

Tables 1 and 2 show the chemical analyses of phosphogypsum and overburden used as capping material at the Estech and IMC sites. At Estech the overburden was higher in pH, lower in electrical conductivity, and higher in Mg, K, P and Mn than was the phosphogypsum. Application of dolomitic limestone raised the pH of the overburden by 0.2 unit and of the phosphogypsum by 0.3 unit. Available Mg was also increased by dolomite application. At the IMC site the overburden was higher in pH, Mg, K, Mn, and Zn than the bare phosphogypsum, but the dolomite amended phosphogypsum was higher in pH and Mg than the overburden. Phosphogypsum at both sites was chemically similar, except the pH was a little lower at IMC. The pH of the IMC overburden was lower and its electrical conductivity was higher than that of the Estech overburden. The overburden caps at both the Estech and IMC sites were loamy sands (Estech: 84% sand, 4% silt, and 12% clay. IMC: 85% sand, 4% silt and 11% clay).

Vegetation

Estech. During the winter and early spring of 1990-91, an excellent cover of ryegrass became established on all the overburden-covered plots, but the phosphogypsum without overburden cover remained nearly bare. This is in contrast to a fertilizer study in which ryegrass did become established on phosphogypsum without overburden cover (Richardson 1992). In late spring the annual ryegrass died back and the warm season perennial grasses began to grow. In June, 1991, green grass (mainly bermudagrass) and weed cover appeared greater on the phosphogypsum without overburden cover than with overburden cover. By mid-July 1991, green grass cover on the dolomite-treated plots was fairly similar, regardless of overburden depth. On the plots without dolomite, grass cover and vigor appeared poorer on those without overburden. This was shown to be due to a magnesium deficiency in an experiment at another site (Richardson 1993). Because of the apparent Mg deficiency in the non-overburden covered plots without dolomite, cover data was only collected from the dolomite treated plots in December 1991. Total living plant cover ranged from 82% to 89% and was not statistically different (0.05 level) with or without overburden cover, and their were no statistically significant differences in litter cover. However, bare ground was statistically greater (0.05 level) without overburden (11%) than with overburden (4% to 5%).

A greater rate of Mg was applied in June 1992 to overcome the Mg deficiency observed in 1991 on phosphogypsum without dolomite. In September 1992, no differences were found between the dolomite or nondolomite treatments, so these data were pooled when analyzing the effects of overburden depth. Table 3 shows that ground covered by plants was slightly, but significantly, less without overburden cover than with overburden cover, but bare ground was only 3% without overburden cover. Even without overburden, plant cover was excellent and exceeded 90 percent. The most striking effect of overburden cover was on plant species diversity. The establishment of weeping lovegrass and bahiagrass was enhanced by overburden cover, and some species such as the invading annual legume, hairy indigo (*Indigofera hirsuta*), failed to grow on phosphogypsum without overburden cover. Bushy beardgrass, another invader, was actually more abundant without overburden cover.

Plant dry weights were measured in October 1992, but neither overburden depth nor dolomitic limestone application had any significant effects on plant yields (data not shown). Apparently, as long as adequate nitrogen, potassium and magnesium nutrients are supplied, and pH exceeds 4.5, excellent vegetation cover and production can be achieved directly on phosphogypsum without an overburden cap (see also Richardson 1993).

IMC. Only bermudagrass and browntop millet were planted at the IMC site. By mid-July, 1991, the height and vigor of browntop millet was clearly better on the overburden-covered than on the non-covered phosphogypsum plots

Table 1. Chemical analysis of overburden and phosphogypsum at the Estech site.[1]

			Mineral analyses (mg/kg)				
	pH	EC [2]	Mg	K	P	Mn	Zn
Overburden (grassed)							
Without dolomite	5.43	0.27	76.2	49.9	2185	3.37	1.72
With dolomite [3]	5.63	0.35	113.0	52.6	2197	3.69	1.37
Phosphogypsum (grassed)							
Without dolomite	4.77	2.35	16.3	14.7	196	0.34	1.57
With dolomite [3]	5.11	2.36	50.8	16.5	252	0.31	1.03
Bare phosphogypsum	4.81	2.36	7.1	10.5	212	0.25	1.16

[1] Mean values of four replicates
[2] Electrical conductivity, dS/m
[3] 2.2 metric tons/ha of dolomitic limestone applied

Table 2. Chemical analyses of overburden and phosphogypsum at the IMC site.[1]

			Mineral analyses (mg/kg)				
	pH	EC [2]	Mg	K	P	Mn	Zn
Overburden (grassed)	5.02	2.01	45.9	32.8	2193	2.43	2.37
Phosphogypsum (grassed) [3]	5.30	2.36	97.5	7.3	NA	0.33	0.64
Phosphogypsum (bare)	4.53	2.35	4.0	4.8	NA	0.15	0.89

[1] Mean values of three replicates
[2] Electrical conductivity, dS/m
[3] Dolomitic limestone applied at rate of 1.5 metric tons/ha
NA Not available

Table 3. Percent cover as affected by overburden depth at the Estech phosphogypsum stack in late September, 1992.[1]

	Overburden depth (cm)			
	0	5	10	15
Bermudagrass	91.0 a	86.6 a	78.0 b	89.4 a
Bahiagrass	9.7 b	28.8 a	33.1 a	32.7 a
Weeping lovegrass	0.0 b	5.1 b	18.2 a	4.0 b
Bushy beardgrass	13.9 a	3.5 b	2.4 b	2.6 b
Other grasses	0.0 c	0.9 c	2.6 b	5.8 a
Hairy indigo	0.0 c	0.0 c	9.7 a	5.2 b
Other broadleaves	0.0 a	0.2 a	0.0 a	0.2 a
Ground covered by plants	93.4 b	97.2 a	97.2 a	98.2 a
Litter	3.3 a	2.1 a	2.1 a	1.8 a
Bare ground	3.3 a	0.7 b	0.7 b	0.0 b

[1] Mean values of 8 replicates. Values within rows followed by the same letter are not significantly different at the 0.05 level. Because of plant overlap, sums of cover values within columns may exceed 100 %.

at IMC. However, plant cover was good on the phosphogypsum without overburden cover, and bermudagrass was actually doing better directly on phosphogypsum where competition from browntop millet was less. The purpose of the browntop millet was to provide a quick cover for erosion control. As expected, the browntop millet, which is an annual, died back in the fall. By December, 1991, bermudagrass cover was greater on the plots with zero or 7.5 cm of overburden than on the plots with 15 cm overburden. Bare ground ranged from 0 to 8 percent, but there were no statistically significant (0.05 level) differences.

In September 1992, ground covered by plants ranged between 96 and 100 percent and did not differ significantly regardless of overburden depth (Table 4). Bare ground was one percent or less. The primary effect of overburden was increased invasion of weeds, such as hairy indigo. The overburden cap had little or no effect on two other invaders, dog fennel (*Eupatorium capillifolium*) and saltbush (*Baccharis halimifolia*).

Plant yields were not significantly affected by overburden depth (data not shown).

Fluoride Analyses

At the Estech site, the fluoride content of bermudagrass was very high (Table 5). The 15 cm overburden cap only reduced the F content of bermudagrass by about one-third. The F concentrations in bushy beardgrass and bahiagrass were somewhat lower, but the 15 cm overburden cap had no significant effect on F concentrations in these two species. The F concentration in bahiagrass grown on phosphogypsum was many times greater than reported for bahiagrass grown on a native sandy soil (Rechcigl et al. 1992). Fluoride concentrations in bermudagrass at the
IMC site were also high, with mean values of 895 mg/kg on phosphogypsum and 670 mg/kg on the 15 cm thick overburden capped phosphogypsum (difference significant at 0.05 level).

Table 4. Percent cover as affected by overburden depth at the IMC phosphogypsum stack in late September, 1992.[1]

	Overburden depth (cm)		
	0	7.5	15
Bermudagrass	96.5 a	99.3 a	99.7 a
Other grasses	0.0 a	0.7 a	1.1 a
Hairy indigo	0.0 c	17.7 b	30.9 a
Dog fennel	8.3 a	9.4 a	10.1 a
Saltbush	1.0 a	0.0 a	2.1 a
Other broadleaves	0.0 a	0.4 a	3.8 a
Ground covered by plants	96.5 a	99.3 a	100.0 a
Litter	2.4 a	0.7 a	0.0 a
Bare ground	1.0 a	0.0 a	0.0 a

[1] Mean Values of three replicates. Values within rows followed by the same letter are not significantly different at the 0.05 level. Because of plant overlap, sums of cover values within columns may exceed 100%.

Table 5. Fluoride concentration (mg/kg) in leaves of bermudagrass, bushy beardgrass, and bahiagrass growing on phosphogypsum with or without a 15 cm overburden cap at the Estech site.[1]

	Overburden Depth (cm)	
	0	15
Bermudagrass	1256 a	847 a*
Bushy Beardgrass	505 b	460 b
Bahiagrass	255 c	215 c

[1] Mean values of three replicates. Values within columns followed by different letters are significantly different at the 0.05 level. An asterix (*) indicates a significant difference within a row at the 0.05 level.

Radionuclides

The activity of Ra-226 in bermudagrass tissue actually increased with increasing depth of overburden cover over phosphogypsum (Table 6), and linear regression analysis produced an r^2 value of 0.92. Apparently, Ra-226 uptake by bermudagrass is not necessarily closely correlated with Ra-226 levels in the growth medium, because Ra-226 concentrations are expected to be much higher in phosphogypsum than in overburden. It may be that the Ra-226 in the overburden is in a more soluble form.

The presence of bermudagrass drastically reduced the radon flux from phosphogypsum, based on the charcoal canister technique employed in this study (Table 6). Radon flux appeared to be reduced about in half by bermudagrass, while 15 cm of overburden cover appeared to reduce it in half again. Although the mean values of radon flux with the various depths of overburden cover differred significantly only in May, there did appear to be a trend for reduced radon flux with increasing overburden depth. The correlation coefficients (r) for radon flux versus overburden depth were -0.74 in February (r^2 = 0.54), -0.66 in March (r^2 = 0.44) and -0.82 in May (r^2 = 0.66). The correlations were significant at the 0.05 level in February and May, and nearly significant (0.052 level) in March.

Table 6. Radium levels in bermudagrass tissue and radon flux from the ground surface at the IMC phosphogypsum stack as affected by overburden cover and by plant cover.[1]

	Control (no plants)	Overburden Depth (cm) (with plant cover) 0	7.5	15
^{226}Ra in plant tissue (pCi/g)	--	0.6 c	1.1 b	1.8 a
^{222}Rn flux (pCi/m_2-s)				
Feb. 1992	18.3 a	5.6 b	3.5 b	1.1 b
Mar. 1992	32.4 a	15.1 b	12.7 b	8.3 b
May 1992	22.8 a	18.5 b	8.1 bc	6.9 c
Average	24.5	11.4	8.1	5.4

[1] Mean values of three replicates. Values within rows followed by the same letter are not significantly different at the 0.05 level.

Discussion

An overburden cap of up to 15 cm in depth was not necessary for the establishment of excellent plant cover on phosphogypsum when the pH was 4.8 to 5.3, when adequate nitrogen, potassium and magnesium were supplied, and when adapted plant species such as bermudagrass were planted. However, the overburden did enhance the establishment of plant species such as weeping lovegrass and bahiagrass, and thus may improve species diversity. The overburden also appears to be a source of nutrients such as magnesium, potassium, phosphorus, manganese and zinc.

Fluoride uptake was high in bermudagrass grown directly on phosphogypsum or on the six inch overburden cap, although the overburden cap did somewhat reduce the fluoride concentration of bermudagrass. Fluoride concentrations were much less in bahiagrass leaf tissue, but they were still nearly ten times the levels considered safe for grazing animals (Rechcigl et al. 1992, Church 1979). The simplest solution to this potential problem may be to prevent animals from consuming a large proportion of their diets from plants grown on phosphogypsum. Alternatives are to provide a thicker cap of low fluoride soil material and to use plant species that are less prone to accumulate fluoride. This study shows that plants may differ in their uptake of fluoride. It may also be possible to reduce the solubility of fluoride by lime application.

Although ^{226}Ra in phosphogypsum or overburden was not measured at the IMC study site, typically phosphate overburden in central Florida has much lower ^{226}Ra levels (average about 5 pCi/g) than phosphogypsum (average about 26 pCi/g) (Roessler et al. 1979 and 1980). Therefore it was surprising that the levels of ^{226}Ra in bermudagrass increased, rather than decreased, with increasing thickness of the overburden cap. It is possible that the ^{226}Ra in the overburden may have been in a more soluble form. Soil pH, soil calcium, and the chemical form of the radium compounds in the soil may all affect plant uptake of radium (Simon and Ibrahim 1988, Mortvedt 1992). The radium levels in bermudagrass grown on phosphogypsum in this study are similar to the radium levels in stargrass (*Cynodon nlemfuensis*), a relative of bermudagrass, grown in central Florida on waste phosphatic clay containing 22 pCi/g of ^{226}Ra (Mislevy et al. 1990). Bahiagrass grown on phosphatic clay contained less than one-third the radium level of the stargrass. This suggests that bahiagrass might take up less ^{226}Ra than bermudagrass when planted on overburden-capped phosphogypsum (bahiagrass establishment was poor on uncapped phosphogypsum at the Estech site).

Plants growing in Ra-containing materials may have an effect on the release of radon to the atmosphere. Lewis and MacDonell (1990), for example, provided evidence that ^{222}Rn dissolved in soil water may be absorbed by plants and released to the atmosphere via cuticular transpiration. This mechanism of ^{222}Rn release would be most significant when a Ra-containing material was covered by a low radioactivity soil cap and plant roots came into contact with the buried material. The data presented here indicate that the presence of plants growing directly on phosphogypsum will reduce the efflux of radon, and a thin (e.g. 15 cm thick) soil or overburden cap will further reduce the radon efflux.

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FRACTIONATION OF HEAVY METALS IN ORGANICALLY AMENDED MINE LANDS[1]

Michael R. Norland[2]

Abstract: Sequential extraction was used in this U.S. Bureau of Mines study to determine the fractions of Cd, Pb, and Zn in abandoned Pb-Zn mill tailing amended with four types of organic residues, applied at three levels, with and without inorganic fertilizer. Trace element fractions selected for sequential chemical extraction were: (1) exchangeable, (2) dilute-acid extractable (bound to carbonates), (3) iron-manganese oxide bound, (4) organically bound, and (5) residual. Metal fractions in lead-zinc mill tailing change dependent upon organic residue type and rate of application. For all organic residue types, concentrations of Cd, Pb, and Zn in the Fe-Mn oxide and organically bound fractions increased as the rate of organic residue application increased, while concentrations of Cd, Pb, and Zn in the exchangeable and dilute-acid extractable fractions decreased as the rate of residue application increased. The results suggest that Cd, Pb, and Zn in abandoned Pb-Zn mill tailing will be prevalent in the weakly mobile, nonbioavailable forms as the rate of organic residue application is increased.

Additional Key Words: sequential extraction, trace elements, bioavailability, compost.

Introduction

The surface waste materials or tailing discarded from mineral processing activities results in a unique environment which potentially can severely inhibit vegetation stabilization. Mineral tailing materials contain residues from metal ore processing operations and are often characterized by high concentrations of heavy metals, alkaline or acid conditions, or toxins from the chemicals used to process the ore (Chambers and Sidle 1991).

Tailing materials associated with mineral processing activities are of environmental concern because of the potential for heavy-metal contamination in the mining-related wastes, and in both surface and shallow ground water flowing under, through, and into a tailing disposal area. Elevated trace element concentrations in tailing materials can also lead to human and animal health risks owing to exposure through ingestion of wind-blown material, through ingestion of the tailing directly, through uptake and accumulation by vegetation, and through various food chains. There are also concerns of trace element contamination due to sedimentation of river channels and reservoirs and the impact of contaminated sediment on floodplain soils (Moore and Luoma 1990).

Reclamation of mineral-related processing wastes contaminated with trace elements has not been successful using conventional agricultrual techniques owing to high concentrations of metal elements and other edaphic factors. Treatment technologies available for contaminated soils have been discussed by Gilbertson (1992); however, mineral mine wastes, particularly tailing materials, differ from soil in terms of chemical, physical, and biological properties. As a result, there is a need for development of treatment technologies applicable to mineral-related waste sites contaminated by trace elements. Abandoned Pb-Zn chat tailing, associated with the Tri-State Mining District consisting of portions of southwest Missouri, southeast Kansas, and northeast Oklahoma, offers an opportunity to study various environmental problems associated with tailing contaminated with trace elements and methods of physical, chemical, and vegetative stabilization.

In a 1989 record of decision (ROD), the U.S. EPA selected a remedial action plan for southeastern Kansas based on the following considerations: (1) surface mine wastes contain high levels of trace elements, particularly Cd, Pb, and Zn; (2) human ingestion through exposure to these elements and leaching of these elements into the shallow ground water and surface waters are of major concern; and (3) the shallow ground water system and surface waters are contaminated with trace elements.

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[2]Michael R. Norland, Soil Scientist, U.S. Department of Interior, Bureau of Mines, Minneapolis, MN, USA.

The goal of this U.S. Bureau of Mines study was to evaluate the concentration and chemical fractionation of Cd, Pb, and Zn in Pb-Zn chat tailing that had been amended with organic residues. A sequential extraction procedure based on work by Tessier et al. (1979) was used to partition these trace elements into five fractions to determine occurrence and bioavailability.

Materials and Methods

The study was conducted within the Galena subsite of the Cherokee County Superfund Site in southeastern Kansas. In April 1990, a 4x3x3 unbalanced factorial experiment arranged in a randomized complete block design with three replications was initiated. The design included four organic waste materials, three levels of application, and three types of fertilizer application. Control plots were included in the design. Treatment combinations were assigned to 2.5- by 4-m test plots at random within each replication.

Four locally available or potentially available organic residues were applied at the surface to experimental plots. The organic residues used were: (1) composted cattle manure, (2) composted yard waste, (3) spent mushroom compost, and (4) turkey litter. Each of the organic amendments were applied at rates of 22.4, 44.8, and 89.6 Mg/ha on a dry weight basis. Fertilizer applications were based on recommendations by the Cherokee County Extension Agriculture Agent for new plantings of native grass and a fescue-legume mixture. The applications of fertilizer consisted of none, 22.4 kg/ha N + 29.4 kg/ha P + 74.4 kg/ha K (recommended rate for the establishment of native grasses), and 44.8 kg/ha N + 48.9 kg/ha P + 130.1 kg/ha K (recommended rate for the establishment of a fescue-legume mixture). The control plots consisted of no organic amendment or fertilizer applications; no organic amendment but fertilizer applied at the recommended rate for native grass establishment; and no organic amendment but fertilizer applied at the recommended rate for fescue-legume establishment.

Table 1 contains a complete list of the grass and forb species selected for study and the rates of each within the mix. The species selected for the seed mix, its percentage in the seed mix, and the rate of application were based on the results of a species trial experiment conducted on chat tailing adjacent to the current study area (Norland and Veith 1990). Recommendations by the Soil Conservation Service (1987) as to the minimum and maximum percentage for each species in a mixture for critical area planting were used as a guideline. The seed mix was formulated based on the rate of pure live seed (PLS) in kilograms per hectare.

Table 1. Species selected for use in the seed mix.

Species	Percent in mixture[1]	Origin[2]	Amount of seed (PLS kg/ha)
Cool-season grasses			
Smooth brome (*Bromus inermis*).....	5	I	1.4
Tall fescue (*Festuca arundinacea*).....	5	I	1.7
Warm-season grasses			
Big bluestem (*Andropogon gerardi*).....	20	N	3.4
Little bluestem (*Schizachyrium scoparium*)...	20	N	3.4
Switch grass (*Panicum virgatum*).....	20	N	2.7
Sand bluestem (*Andropogon hallii*).....	10	N	1.3
Leguminous forbs			
Partridge pea (*Cassia fasciculata*).....	5	N	1.7
Crownvetch (*Coronilla varia*).....	5	I	1.1
Alfalfa (*Medicago sativa*).....	5	I	1.0
Yellow sweetclover (*Melilotus officinalis*)..	5	I	0.8

[1]Precent of each species in bulk mixture, prior to PLS correction.
[2]I=Introduced, N=Native.

During the 1992 growing season, tailing samples were collected in each organic residue type and rate of application treatment plot from nonrhizosphere locations. Three randomly located subsamples were taken to a depth of 15 cm in each plot using a soil recovery probe. Subsamples were composited, based on treatment combination, into 13 samples for analysis.

A five-step sequential fractionation scheme was used to partition the trace elements into exchangeable; dilute, acid-extractable; Fe-Mn oxide bound; organically bound; and residual fractions. The scheme (table 2) was based on work by Tessier et al. (1979), Pickering (1986), and Elliot et al. (1990). All samples were submitted to MVLT Laboratories, Inc., Bismarck, ND, for sequential extraction and analysis. The procedures used by MVLT Laboratories followed those established by Elliot et al. (1990). Reagents were added to 1.0-g samples of oven-dried chat tailing, which were then agitated on a shaker for the specified time and centrifuged at 1,500 rpm for 20 min. The centrifugate was decanted into a plastic bottle and set aside for analysis.

Table 2. Chemical extraction scheme for trace element fractionation.

Fraction	Reagent conditions[1]	Shaking time, h
Exchangeable.....	16 mL 1M $MgCl_2$ (pH 7.0).....	1
Dilute acid-extractable...	16 mL 1M NaOAc adjusted to pH 5 with HOAc.....	5
Fe-Mn oxide bound.....	40 mL of 0.175M $(NH_4)_2C_2O_4$ and 0.1M $H_2C_2O_4$.....	4
Organically bound.....	40 mL of 0.1M $Na_4P_2O_7$.....	24
Residual.....	Dry (in oven) 0.1 g of material remaining after previous step. Add 4 ml HNO_3 1.0 mL $HClO_4$, and 6.0 mL HF and heat at 140 °C for 3.5 h. Add 5.0 g boric acid and dilute to 100 ml volume.	

[1]These volumes are for 1.0 g oven-dried chat tailing.

The next extracting solution was added, and the extraction process was repeated. After the first four extractions, 100 mg (dry wt.) of the remaining material was transferred to a Teflon decomposition bomb, and a hot triple-acid digestion was performed (Elliott and Shields 1988). Cadmium, Pb, and Zn analysis was performed using flame atomic absorption spectroscopy.

Results and Discussion

The total concentrations of Cd, Pb, and Zn in the Galena subsite tailing materials are extremely high (table 3). For comparative purposes, the trace element concentration action levels established by the U.S. EPA for the Galena subsite of the Cherokee County Superfund Site are included. Trace element concentration action levels are used to identify surface-deposited mine wastes (waste rock, chat tailing, and slag) for selective placement in disposal sites such as open subsidences, pits, and shafts within the Galena subsite. This action will through proper disposal essentially eliminate human exposure via ingestion of contaminated surface-deposited mine wastes (dust or direct) and will reduce long-term shallow ground water and surface water trace element loading resulting from mine wastes (Black and Veatch 1990).

Trace element concentration action levels established are: (1) Pb - 1,000 mg/kg, (2) Zn - 10,000 mg/kg, and (3) Cd - 25 mg/kg. According to the U.S. EPA, chat tailing characterized above the action level for Pb and below the action level for Cd and Zn may be mixed with plus 5-cm waste rock and buried in wet voids. Chat tailing containing Cd and/or Zn above the action level must be buried in dry voids. This chat tailing may be mixed with waste rock that is also placed in dry voids to minimize settlement after backfilling. Chat tailing below the action level for Cd, Pb, and Zn may be used as cover material or material for recontouring mine waste areas.

Table 3. Trace element concentrations of chat tailing and uncontaminated soils, mg/kg, dry wt.

Trace element	Chat tailing Range	Chat tailing Mean[1]	Soils Range[2]	Soils Typical[3]	Action levels
Cd.....	6- 86	27	0.01- 7	0.4	25
Pb.....	191- 2,432	800	2 -200	15	1,000
Zn.....	898-31,940	11,939	10 -300	40	10,000

[1]Means are given for 14 chat tailing samples.
[2]Lindsay, 1979.
[3]Berrow and Reaves, 1984.

The mean total concentration of Cd and Zn exceed established action levels, while the mean value for Pb was below the action level. All three trace elements have ranges that fall above and below their specific action levels, but are above levels typically found in uncontaminated soils. This indicates that before the chat tailing is used as cover material or material for recontouring mine waste areas, x-ray fluorescence spectrometry or other U.S. EPA approved field detection methods must be used to determine the concentration of each trace element.

Without proper classification, the material used for cover may continue to be a human and animal health risk and lead to revegetation failures due to increased plant concentration of trace elements (phytotoxicity) or trace element chlorosis, which limits plant productivity. Trace element concentrations exceeding 8 mg/kg Cd, 100 to 400 mg/kg Pb, and 70 to 400 mg/kg zinc (Kabata-Pendias and Pendias 1984) can render soils nonproductive due to trace element phytotoxicity. Concentrations of Cd, Pb, and Zn are far above the apparently safe soil concentrations of 7, 200, and 300 mg/kg, respectively, and should be considered phytotoxic (Bowen 1979, Lindsay 1979, Bresler et al. 1982, Bohn et al. 1985). In light of these data, the toxic tailing concentrations of Cd, Pb, and Zn may be playing a large role in limiting vegetation establishment on the site. When taken up by the plant, these trace elements compete with and displace iron and other ions in enzyme systems, resulting in the alteration of the sterostructure and operation of the enzyme in plant metabolism (Lagerwerff 1967, Lagerwerff and Specht 1971).

Concurrent with surface mine waste classification and removal, all mine waste areas on the Galena subsite will be recontoured to eliminate closed basins (ponded areas) and vegetated to control erosion. The revegetation plan of the Galena subsite includes the native grasses listed in table 1 and the use of organic residues as the primary tailing amendment. Indiangrass (Sorghastrum nutans (L.) Nash) has been added to the native grass seed mix at a rate of 2.7 kg/ha PLS.

Treatment effects on the distribution and concentration of trace elements determined by chemical fractionation are shown in tables 4 and 5. For convenience, Elliot et al. (1990) placed the fractions evaluated into two broad categories which were distinguished by whether or not solid-phase dissolution is required to release the trace elements. Trace elements displaced without substrate dissolution are nondetrital (labile), while solid-phase dissolution is essential to solubilize detrital (nonlabile) trace elements. Nondetrital trace elements are those that are water soluble, exchangeable, or adsorbed onto solids phases and would correspond to the exchangeable and dilute acid-extractable fractions in table 4. Exchangeable trace elements would likely represent those most readily mobilized in tailing-water systems, whereas low pH (<5) would be required to release the dilute acid-extractable trace elements (Elliot et al. 1990). The detrital fraction (moderately labile to nonlabile) includes trace elements present as precipitated compounds (hydroxides, sulfides), fixed within the silicate matrix, or bound to hydrous oxides through occlusion, chemisorption, or coprecipitation (Elliot et al. 1990). The lability of organically bound trace elements is questioned, but because this fraction is evaluated after oxide-bound metals (table 2), it will be considered as part of the nonlabile trace elements (Elliot et al. 1990).

Table 4. Organic residue treatment effects on Cd, Pb, and Zn chemical fractionation, % of total trace element in each fraction.

Treatment[1]	Trace element	Fraction: Exchangeable	Acid soluble	Fe-Mn oxide bound	Organically bound	Residual
YW - 1	Cd	17.1	14.5	15.1	19.8	33.4
	Pb	2.2	21.3	66.6	1.0	8.9
	Zn	1.3	43.3	18.3	15.9	21.1
YW - 2	Cd	14.9	13.4	11.6	27.0	33.0
	Pb	1.5	20.6	64.3	1.4	12.1
	Zn	1.2	32.9	14.8	24.1	26.9
YW - 3	Cd	9.0	15.2	20.5	37.0	18.2
	Pb	0.9	14.0	75.4	1.3	8.4
	Zn	0.7	26.2	27.5	32.5	13.2
CM - 1	Cd	16.8	19.5	14.6	16.3	32.8
	Pb	1.5	22.5	65.2	1.5	9.3
	Zn	0.8	41.6	14.5	15.2	27.9
CM - 2	Cd	17.9	17.4	14.4	21.2	29.1
	Pb	1.1	25.3	63.1	1.3	9.2
	Zn	1.3	43.3	15.7	17.7	22.0
CM - 3	Cd	13.9	15.1	25.1	33.4	12.4
	Pb	1.6	18.0	71.5	1.7	7.2
	Zn	2.0	32.0	29.2	27.5	9.4
MC - 1	Cd	16.3	22.1	16.3	17.7	27.5
	Pb	1.6	20.2	54.7	4.4	19.1
	Zn	1.3	41.3	13.7	18.8	24.9
MC - 2	Cd	9.3	13.9	17.7	40.6	18.5
	Pb	0.9	19.3	68.0	1.1	10.7
	Zn	0.7	31.1	24.7	30.0	13.5
MC - 3	Cd	7.2	13.1	14.1	56.1	9.5
	Pb	0.9	15.5	74.2	1.0	8.4
	Zn	0.8	22.2	15.5	53.3	8.1
TL - 1	Cd	15.1	23.6	23.4	15.1	22.9
	Pb	2.1	25.6	63.5	1.4	7.4
	Zn	1.1	52.4	20.2	11.6	14.7
TL - 2	Cd	8.6	22.1	23.5	24.6	21.3
	Pb	1.4	23.4	63.7	1.7	10.0
	Zn	1.0	38.3	16.6	25.0	19.1
TL - 3	Cd	6.2	18.9	26.6	40.7	7.7
	Pb	0.4	9.3	55.1	3.7	31.5
	Zn	0.8	24.2	19.5	34.8	20.7
Control	Cd	21.4	17.2	14.2	23.9	23.4
	Pb	2.3	25.8	62.5	1.3	8.2
	Zn	1.1	41.8	20.8	21.8	14.6

[1]YW=composted yard waste, CM=composted cattle manure, MC=spent mushroom compost, TL=turkey litter; 1=22.4 Mg/ha, 2=44.8 Mg/ha, 3=89.6 Mg/ha.

Table 5. Organic residue treatment effects on Cd, Pb, and Zn chemical fractionation, concentration of trace elements in each fraction.

Treatment[1]	Trace element	Fraction: Exchangeable	Acid soluble	Fe-Mn oxide bound	Organically bound	Residual	Total
		mg/kg					
YW - 1	Cd	8.7	7.4	7.7	10.1	17.0	50.9
	Pb	19.5	193.0	602.3	9.2	80.6	904.6
	Zn	214.0	6,930.0	2,930.0	2,540.0	3,380.0	15,994.0
YW - 2	Cd	9.1	8.2	7.1	16.5	20.1	61.0
	Pb	10.8	147.6	460.1	9.9	86.8	715.2
	Zn	206.0	5,470.0	2,460.0	4,010.0	4,460.0	16,606.0
YW - 3	Cd	4.8	8.1	10.9	19.7	9.7	53.2
	Pb	6.8	102.9	554.3	9.5	61.5	735.0
	Zn	94.6	3,720.0	3,900.0	4,620.0	1,870.0	14,204.6
CM - 1	Cd	10.5	12.3	9.1	10.2	20.5	62.5
	Pb	9.2	136.9	396.8	9.4	56.4	608.7
	Zn	143.0	7,220.0	2,520.0	2,630.0	4,840.0	17,353.0
CM - 2	Cd	12.7	12.3	10.2	15.0	20.6	70.8
	Pb	8.2	191.1	475.7	9.6	69.5	754.1
	Zn	283.0	9,360.0	3,400.0	3,820.0	4,750.0	21,613.0
CM - 3	Cd	6.7	7.3	12.1	16.1	6.0	48.2
	Pb	9.0	101.0	401.8	9.6	40.7	562.1
	Zn	298.0	4,700.0	4,290.0	4,040.0	1,380.0	14,708.0
MC - 1	Cd	12.0	16.2	12.0	13.0	20.2	73.4
	Pb	18.3	237.4	641.3	52.2	224.1	1,173.3
	Zn	221.0	6,930.0	2,290.0	3,150.0	4,180.0	16,771.0
MC - 2	Cd	5.9	8.8	11.2	25.7	11.7	63.3
	Pb	8.4	170.4	601.0	10.0	94.5	884.3
	Zn	127.0	5,430.0	4,320.0	5,250.0	2,360.0	17,487.0
MC - 3	Cd	4.2	7.6	8.2	32.6	5.5	58.1
	Pb	10.0	175.5	839.2	11.2	95.4	1,131.3
	Zn	144.0	4,230.0	2,960.0	10,200.0	1,540.0	19,074.0
TL - 1	Cd	7.1	11.1	11.0	7.1	10.8	47.1
	Pb	14.6	177.5	440.0	9.8	51.3	693.2
	Zn	168.0	8,370.0	3,230.0	1,860.0	2,350.0	15,978.0
TL - 2	Cd	6.4	16.5	17.6	18.4	15.9	74.8
	Pb	8.0	137.9	376.0	9.8	58.8	590.5
	Zn	191.0	7,360.0	3,190.0	4,800.0	3,670.0	19,211.0
TL - 3	Cd	3.6	11.0	15.5	23.7	4.5	58.3
	Pb	3.8	81.8	484.4	32.2	276.8	879.0
	Zn	145.0	4,560.0	3,690.0	6,570.0	3,910.0	18,875.0
Control	Cd	10.4	8.2	6.9	11.6	11.2	48.3
	Pb	17.9	198.2	474.0	9.5	61.0	760.6
	Zn	149.0	5,423.0	2,777.0	2,917.0	1,950.0	13,216.0

[1]YW=composted yard waste, CM=composted cattle manure, MC=spent mushroom compost, TL=turkey litter; 1=22.4 Mg/ha, 2=44.8 Mg/ha, 3=89.6 Mg/ha.

The easily mobile, exchangeable Pb and Zn fractions were 2.3% and 1.1%, respectively, of the total Pb and Zn concentration in unamended chat tailing. Additionally, exchangeable Pb and Zn were not affected by organic residue type or rate of application. Exchangeable Cd was reduced compared to the control when organic residues were applied and the rate of application was increased (table 4). The unamended tailing exchangeable Cd fraction of 21.4% of the total was similar to what Sims and Kline (1991) found for unamended Atlantic coastal plain soils, 21.8% to 35.1% of the total. For each organic residue there was a progressive decrease in the exchangeable Cd fraction as the rate of application was increased. Applications of 89.6 Mg/ha of composted cattle manure, composted yard waste, spent mushroom compost, and turkey litter decreased the exchangeable Cd fraction to 13.9%, 9.0%, 7.2%, and 6.2% of the total, respectively. This agrees with Sims and Kline (1991) and Sposito et al (1983), who found that additions of 45 Mg/ha of composted sewage sludge will decrease the exchangeable Cd fraction.

The acid-soluble Cd and Pb fraction was unaffected by the application of organic residues. The relative amounts of both Cd and Pb did not show much change with increasing rates of organic amendment application, with the exception of Pb, where the acid soluble fraction decreased at the 89.6 Mg/ha rate of application for all organic residues. In unamended tailing materials, acid-soluble Zn accounted for a substantial portion (42%) of the total Zn. The Zn fraction, however, exhibited a progressive decrease as the rate of organic residue application increased, regardless of type. This suggests that if there is a sudden change in the tailing pH from the present value of 6.8 to a more acidic reaction, less Zn would be available in this fraction to be affected by change in tailing pH.

The sum of the first two fractions (exchangeable and acid soluble) represents the most labile, bioavailable fractions in the tailing system. On the basis of relative amounts of each element released from control plots in the first two extraction steps, Zn would have the greatest mobility in the tailing, followed by Cd and Pb. With the application of organic residues and increased rates of application, there was a decrease in labile (mobile), bioavailable fractions in the tailing system.

In unamended tailing, 28% of the Pb was in the labile fraction. With increasing rates of organic residue application, to 89.6 Mg/ha, there was a progressive decrease in labile Pb to 20% with composted cattle manure, 16% with spent mushroom compost, 15% with composted yard waste, and 9% with turkey litter. Most of the Pb (63%) was in the nonbioavailable Fe-Mn oxide bound fraction. With the exception of turkey litter, applications of organic residues at a rate of 89.6 Mg/ha increased the relative amount of Pb in the Fe-Mn oxide bound fraction to 72% with composted cattle manure, 74% with spent mushroom compost, and 75% with composted yard waste. This agrees with the findings of Sims and Kline (1991), who found that applications of co-composted sewage sludge (44 Mg/ha) to soil increased the relative amounts of precipitated Pb and that the majority of soil Pb was in the precipitated fraction.

The application of organic residues increased organically bound Cd and Zn. In unamended tailing, organically bound Cd and Zn accounted for 24% and 22% of the total Cd and Zn in the tailing system. At an application rate of 89.6 Mg/ha, the amount of organically bound Cd increased from 24% in unamended tailing to 33% when composted cattle manure was applied, 37% when composted yard waste was applied, 41% when turkey litter was applied, and 56% when spent mushroom compost was applied. This retention of Cd by organic matter has been documented for soils by Elliot et al. (1986). At the same rate of application, organically bound Zn increased from 22% of the total in unamended plots to 28% when composted cattle manure was applied, 33% when composted yard waste was applied, 35% when turkey litter was applied, and 53% when spent mushroom compost was applied. Organically bound Pb was not affected by applications of organic residues.

On the basis of the relative amounts of each element in the Fe-Mn oxide bound, organically bound, and residual fractions, the application of organic residues to abandoned Pb-Zn chat tailing increases the nonlabile (detrital) fractions in the tailing system. At an application rate of 89.6 Mg/ha, the relative amount of nonlabile Cd increased from 62% of the total in unamended tailing to 71% of the total when composted cattle manure was applied, 75% of the total when turkey litter was applied, 76 % of the total when composted yard waste was applied, and 80% of the total when spent mushroom compost was applied; for Pb, the nonlabile fraction increased from 72% of the

total to 77% when composted yard waste was applied, 80% when composted cattle manure was applied, 84% when spent mushroom compost was applied, and 90% when turkey litter was applied at a rate of 89.6 Mg/ha; for Zn, the nonlabile faction increased from 57.2% of the total in unamended tailing to 66%, 73%, 75%, and 77% of the total in plots amended with composted cattle manure, composted yard waste, turkey litter, and spent mushroom compost, respectively, at a rate of 89.6 Mg/ha.

Conclusions

The application of organic residues to abandoned Pb-Zn mine tailing not only benefits plant establishment, growth, and community development (Norland 1993), but also changes the distribution of trace elements among the tailing chemical fractions. A five-step fractionation procedure identified the main trace element retention forms based on bioavailability.

Several organic residues were evaluated to determine their effect in reducing the bioavailability of Cd, Pb, and Zn in contaminated mine wastes. In unamended plots, the bioavailability (mobility) of Zn was the greatest, followed by Cd and Pb. In amended plots, the type of organic residue used had no effect on tailing chemical fractions, but the rate of application did affect the relative amount of each element in a chemical fraction. Additions of composted yard waste, composted cattle manure, spent mushroom compost, or turkey litter at a rate of 89.6 Mg/ha all reduced the labile (biovailable) Cd, Pb, and Zn fractions, while increasing the nonlabile (nonbioavailable) Cd, Pb, and Zn fractions.

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GROWTH AND TRACE METAL CONTENT OF POPLARS PLANTED ON TAILING TREATED WITH MUNICIPAL SOLID WASTE COMPOST[1]

Steven W. Dewar and Michael R. Norland[2]

Abstract: Taconite tailing wastes in Minnesota are infertile and difficult to vegetate to meet mineland reclamation standards. Taconite tailing reclamation is a potential use for a municipal solid waste (MSW) compost product. MSW compost is rich in organic compounds, and its application to tailing is likely to improve fertility even though it may increase the level of potentially harmful trace metals. This study assesses tree growth and elemental tissue concentrations of Souixland poplar (Populus deltoides var. Souixland) planted in four 420-m^2 field plots. Plots were randomly treated with 0, 22.4, 44.8, or 89.6 Mg/ha of MSW compost. After compost application, all plots were fertilized, disced, seeded with a grass-legume mixture, and planted with rooted cuttings of poplar at a 2 by 2.5 m spacing. After two growing seasons, tree growth was significantly greater where compost was applied. The average growth per live tree was highest in the 89.6 Mg/ha plot. Average tree height increased from 68.2 to 84.1 cm, a 23% increase over the control for the highest rate. Survival of trees; however, ranged from 84 to 98% for the high rate and control, respectively. Ground cover and biomass increased with increasing compost application rates. Analyses of the compost indicated that it met regulations for field application, including PCB's and trace metals. Soils analyses of the tailing after compost application showed higher levels of N, P, S, Zn, and Cu with increasing application rates. Analyses of poplar leaf tissue revealed slightly higher concentrations of P, Zn, Cu, and Cd for treatments receiving compost. Results suggest that applying compost to coarse tailing will increase the growth of ground cover and poplar trees without adversely affecting the environment.

Additional Key Words: mineland reclamation, taconite, Minnesota, Souixland poplar, Populus deltoides, survival, ground cover, biomass

Introduction

In northeastern Minnesota there are over 10,000 ha of land covered with taconite tailing waste from eight taconite mining operators. Minnesota's Rules Relating to Mineland Reclamation are administered by the Department of Natural Resources (MDNR) and require that these areas be vegetated to meet a 90% cover standard after 3 years (MDNR 1980). Coarse-textured tailing areas are particularly difficult to vegetate to meet reclamation requirements. Dependent upon the type of beneficiation process employed, taconite facilities produce tailing either as distinct coarse and fine tailing or as unseparated composite tailing. Coarse tailing are usually utilized to construct a dike which encircles the basin, while a slurry of fine materials is released into the basin interior to settle out of suspension. There are now over 3,000 ha of coarse tailing disposal areas in Minnesota.

The physical and chemical characteristics for taconite tailing do not differ widely among the mine operators (Jordan and Dewar 1985). Taconite tailing do not contain unusual amounts of metals that might inhibit plant growth (Blake 1975). Major plant nutrients of nitrogen and phosphorus are low and have to be applied. In addition, coarse tailing are droughty and lack organic matter essential to maintain soil productivity.

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[2]Steven W. Dewar is Field Supervisor, Mineland Reclamation, Minnesota Department of Natural Resources - Division of Minerals, Hibbing, MN 55746; Michael R. Norland is a Soil Scientist, Environmental Technology, Twin Cities Research Center, Bureau of Mines, U.S. Department of the Interior, Minneapolis, MN 55417.

Adding organic matter materials, such as composted municipal garbage, can improve the physical, chemical, and biological properties of soil.

Landfilling solid waste has become less environmentally acceptable and more highly regulated. Amendments to the Minnesota Waste Management Act 1991, list waste reduction, reuse, recycling, and composting as high priorities (Wirth 1991). Composting is a way to transform waste materials into a useful soil amendment. Composting destroys pathogens and converts putrescible organic wastes into a stabilized material, although potentially hazardous levels of trace metals may be introduced into the soil environment and food chain (He et al. 1992).

Composts must reach an acceptable degree of maturity before being marketed as a soil amendment for unrestricted use (MPCA 1991). The degree of stabilization reached by the organic matter in MSW compost appears to be important in determining potential mobility of heavy metals (Leita and De Nobili 1991). Bioavailable trace metals can have strong toxic affects on plants. Incorporating MSW compost into the soil may limit the level of heavy metal solubility. Other notable effects of immature compost on plants is the inhibition of plant seed germination by the production of phytotoxic substances (Wong 1985) and the biological blockage of soil-available N (Jimenez and Garcia 1989).

Marketing a garbage compost product is difficult. The contaminant level, odor, the presence of glass and plastic, and cost can be problems and are of concern to potential users of compost (Stark 1987). Adverse compost characteristics can be controlled by proper upfront planning in the area of quality control. The quality of the compost will determine a particular market for the product. MSW compost products are inherently variable (Walker and O'Donnell 1991, Johnson and Crawford 1993), and the potentially harmful components must be considered before use. Composted MSW is now being used in agriculture, landscaping, and horticulture. Opportunities now exist for solid waste compost use in mineland reclamation (Hart 1968, Stark 1987, Jokela et al. 1990).

Minnesota has over one-third of the operating MSW compost systems in the U.S. (Johnson and Crawford 1993). The research base necessary for composted MSW to enter markets for environmentally acceptable use on mineland reclamation projects in Minnesota is limited. In a companion study (Melchert et al. 1994), MSW compost was applied to a tailing site to examine impacts on water quality and runoff. In recent years other studies have been initiated by the MDNR and the U.S. Bureau of Mines, studying the usefulness of MSW compost as a soil amendment for revegetating coarse taconite tailing (Norland et al. 1991, 1992, 1993). The objective of this paper is to assess tree growth and trace metal concentrations in leaf tissue for trees planted in coarse tailing plots treated with four rates of MSW compost.

Materials and Methods

The experimental plots are located 10 km south of Eveleth, MN. Eveleth Mines made a portion of their coarse tailing stockpile available for this research. Eveleth Mines separates coarse materials from the tailing and truck-hauls it to the disposal site. The study area experiences a typical continental climate with wide extremes in temperature between summer and winter, with a mean annual temperature of 4° C and precipitation of 688 mm (Aakre 1975).

The texture of coarse taconite tailing is mostly sand and fine gravel. Analyses of tailing material show it to be alkaline in reaction and low in organic matter, cation-exchange capacity (CEC), soluble salts, and concentrations of nitrate-nitrogen and phosphorus. The chemical and physical properties of the coarse tailing at Eveleth Mines are described in detail elsewhere (Norland et al. 1991).

The research site was prepared by leveling end-dumped coarse tailing piles. Four plots, each measuring 17.5 by 24 m were randomly assigned a compost treatment. A 5-m buffer zone separated adjacent plots. In

late April 1992, composted MSW was uniformly applied at the rates of 0, 22.4, 44.8, and 89.6 Mg/ha. Fertilizer was applied to all four plots at the rate of 224 kg/ha granular diammonium phosphate. The area was then disced so that amendments were incorporated to a depth of 15 cm. Treatment plots were then seeded with a grass-legume cool season mixture that included smooth brome (Bromus inermis Leyss.), red fescue (Festuca rubra L.), perennial ryegrass (Lolium perenne L.), alfalfa (Medicago sativa L.), and buckwheat (Fagopyrum sagittatum Gilib.). Plots were then mulched with hay at the rate of 2 Mg/ha and crimped in with a disc. Immediately after seeding, the area was hand planted with rooted cuttings of Souixland poplars (Populus deltoides var. Souixland). Individual trees were selected for uniformity prior to planting. Each plot was planted with 84 poplars at a 2 by 2.5 m spacing.

The compost was obtained from a MSW compost plant located at Mora, MN. The municipal waste it receives comes from a five-county area with a mostly rural population of about 100,000. More than 50% of the raw household trash consists of organic waste such as paper, food waste, and plant trimmings and is compostable. At the time the compost used in this experiment was obtained, the processing procedure (East Central Solid Waste Commission 1992) was as follows: corrugated cardboard, newsprint, and plastics were removed, and ferrous metals were separated out by magnet. Remaining materials moved through a shredding, screening, and wet separation process. The wet separators removed glass and other substances. The material biodegraded under controlled moisture and aeration conditions for 2 weeks. The organic material went through a screening process to remove the remaining inorganic material. It then proceeded to the final static aerated pile composting phase. The resulting compost product was ready after 180 d. It was then shredded and screened to separate high-grade from low-grade compost.

The Minnesota Pollution Control Agency (MPCA) administers rules related to solid waste management (MPCA 1991). This includes operating a compost facility and marketing a compost product. Compost distributed as a soil amendment must be registered with the Minnesota Department of Agriculture. Compost produced at a solid waste compost facility must be classified as class I or class II. Class I compost may be distributed for unrestricted use and class II may be distributed on a restricted basis. Class I compost may not be processed with sewage sludge and may contain contaminant levels no greater than shown in table 1. Cd, Cu, and Zn came closest to approaching the allowable limit for a class I compost. Class I compost must be stored until the compost is mature and may contain inert materials in quantities (based on particle size) no greater than shown on table 2. A class II compost consists of any compost generated from sewage sludge or fails to meet the class I standards.

The compost facility reported that the compost used on these experimental plots was a class I compost based on the results from analyses completed on the compost just prior to delivery to

Table 1. Comparison of the allowable average contaminant concentrations (mg/kg) for a class I compost (dry weight basis) and sample results from compost used in this experiment.

	Concentration	
Contaminant	Allowable	Compost result
PCB.....	1	Not detected
Cadmium.....	10	8.4
Copper.....	500	370
Chromium....	1,000	89
Lead.....	500	158
Mercury.....	5	1.1
Nickel.....	100	32
Zinc.....	1,000	750

Table 2. Comparison of the allowable quantities of inert materials for a class I compost and sample results from compost used in this experiment.

	Dry weight percent	
Particle size	Allowable	Compost result
Up to 10 mm.....	1.0	0.59
Up to 16 mm.....	2.0	1.28
Up to 25 mm.....	4.0	1.30

the site (tables 1 and 2). Compost material was considered to be mature since it had cured for 180 d and was not processed with sewage sludge.

Tree survival and heights were measured at the end of the growing season in August 1993. Only the inner 50 trees in each treatment plot were measured. Trees planted on the outside rows and row ends were considered buffer trees and were not measured. Trees were measured from ground level to the terminal bud of the dominant shoot of each living tree. Tree survival and heights were calculated and averaged by treatment.

Elemental tissue concentrations were determined for trees by treatment. Two mature poplar leaves were collected from the upper crown of 20 randomly selected live trees for each treatment and composited. Leaves were dried at 70° C and ground in a Wiley mill to pass a 1-mm sieve. Ground tissues were sampled for chemical analysis using ICP (inductively coupled plasma atomic emission spectroscopy) methods.

Ground cover biomass and percent cover were measured in August 1993 for each treatment. Percent ground cover was measured using a systematic point-quadrat sampling method (Raelson and McKee 1982) to estimate cover within a 95% statistical confidence. Ground cover biomass was estimated by clipping all aboveground plant material from 6 randomly selected 0.11-m^2 quadrats for each plot. Plant material was dried for 48 h at 70° C and weighed.

Treatment rate effects of composted MSW on tree heights were tested using a one-way analysis of variance (ANOVA). Multiple comparisons of treatment means for tree heights were made using the least significant difference (LSD) test (Steel and Torrie 1980).

Results

Ground Cover

A seeded grass-legume ground cover consisting mostly of smooth brome and alfalfa dominated all treatment plots. A minor component of the ground cover included a few scattered weeds found throughout all plots, these included: Indian mustard (Brassica juncea L.), lamb's-quarters (Chenopodium album L.), and ox-eye daisy (Chrysanthemum leucanthemum L.)

Percent ground cover and biomass production differed substantially between MSW compost treatments 15 months after seeding (2 growing seasons). Biomass and percent cover were higher than the control for all compost treatments (table 3). Cover responses increased as MSW compost application rates increased. The highest rate (89.6 Mg/ha) had significantly greater percent cover and biomass than any of the other treatments. Percent cover and biomass compare closely after 2 growing seasons with results from a study (Norland et al. 1992, 1993) completed earlier using similar application rates of MSW compost. Subsequent findings (Norland et al. 1994) showed that the 90% cover standard was achieved after 4 seasons.

Table 3. Percent ground cover and biomass production (dry weight) by compost treatment 15 months after application.

Compost Rate, Mg/ha	Percent Cover	Biomass, kg/ha
Control...	22	165
22.4.....	46	1,066
44.8.....	53	1,201
89.6.....	80	2,553

Table 4. Survival rates by compost treatment for Souixland poplars 15 months after planting.

Compost rate, Mg/ha	Percent survival[1]
Control...	98
22.4.....	86
44.8.....	96
89.6.....	84

[1]Average of 50 trees.

Tree Survival and Growth

Trees suffered little mortality on all treatments 15 months after planting (table 4). Tree survival ranged from 84 to 98% for the high and control rates, respectively. A slightly higher mortality rate for the compost treatments may be attributable to greater amounts of ground cover competing with trees. Competing vegetation can lower the survival and growth of newly planted trees (Smith 1962). Survival rates above 80% are considered good for new tree plantings.

Applications of MSW compost significantly increased the growth of poplars for all treatment rates (table 5). Mean tree heights ranged from 68.2 to 84.1 cm for the control and highest compost rate, respectively. Increases in tree growth from compost application ranged from 16% to 23% over the control (fig. 1). Tree growth increased with increasing rates of compost.

A leaf blight developed on most of the trees during July 1993 and may have reduced the growth of infected trees. The fungi for this disease (Marssonia populi) develop in wet weather. The spring and summer of 1993 had an above average amount of rainfall. Although trees are seldom killed by the fungi involved, their growth rate can be reduced (French and Cowling 1975).

Table 5. Multiple paired comparisons between mean tree heights for all compost treatments using the LSD test.

Compost rate, Mg/ha	Mean tree height, cm[1]
Control...	68.2 a
22.4.....	79.2 b
44.8.....	80.6 bc
89.6.....	84.1 c

[1]Average of 50 trees. Means followed by the same letter are not significantly different, $LSD_{.05}$ = 4.2 cm.

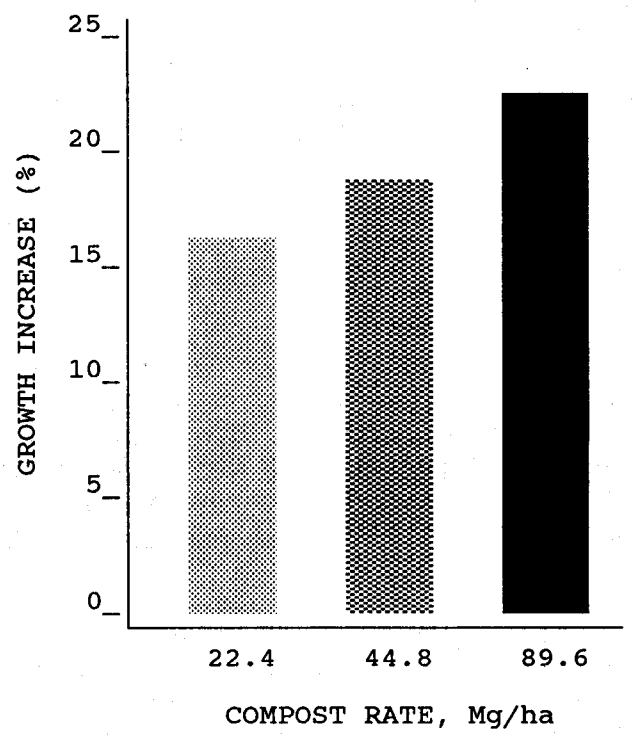

Figure 1. Percent increase in growth of Souixland poplars for each compost treatment as compared to the control treatment.

Soil and Tissue Chemistry

Analyses of the soils and leaf tissue indicated there were changes in soil and leaf tissue chemistry with increasing rates of MSW compost applications (tables 6 and 7). Metals and PCB's can have toxic impacts on plant or animal life. Elevated levels of trace metal contaminants of Zn and Cu were detected in both the soils and tissue analyses with increasing rates of compost. Higher levels of Cd were detected in leaf tissue. Cd was not tested in the soils analyses. Since PCB's were not detected in the compost it was not analyzed in the soils or tissue. Concentration levels of Pb and Cr in the leaf tissue was at or near detection limits. Compost analyses from table 1, showed that Cd, Cu, and Zn come closest to approaching the allowable limit for a class I compost. Elevated levels of Zn, Cu, and Cd may be of some concern; however, they are of less concern when incorporated into alkaline tailing. These metals move less readily in an alkaline soil and tend to revert to more unavailable forms over time (Crawford and Johnson 1988).

Potentially harmful trace metals did not limit plant growth. For all treatment rates, concentrations of Zn

and Cu in the tailing soil were within the normal range found in soils (Allaway 1968). Plant tissue values for Zn, Cu, and Cd were below toxic levels (Melsted 1973) for all treatments. This is substantiated by the higher amounts of ground cover and tree growth with increasing compost rates reported earlier.

Compost applications tended to increase soil fertility, as indicated by a considerable increase in CEC and extractable P with increasing rates (table 7). Levels of extractable P above 15 mg/kg are sufficient for grass-legume stands (Jokela et al. 1981). Higher levels of organic matter were detected at the high rate. There was no substantial change in the pH, soluble salts, and nitrate-nitrogen. It is unlikely that increases in extractable Zn and Cu contributed significantly to increases in growth.

Table 6. Elemental tissue concentrations (mg/kg) of poplar leaves for selected elements by compost treatment 15 months after application.

	Compost rate, Mg/ha			
Element	0	22.4	44.8	89.6
P.....	1,531	2,466	2,281	2,493
Zn.....	30	164	111	197
Cu.....	3.2	6.4	5.7	6.2
Pb.....	1.8	<1.7	<1.7	<1.7
Cr.....	<.28	.30	.31	.29
Cd.....	.19	.66	.38	.84

Table 7. Selected soil characteristics by compost treatment 15 months after application.

	Compost rate, Mg/ha			
	0	22.4	44.8	89.6
pH.....	8.4	8.5	8.2	8.0
Organic matter, %.....	0.5	0.4	0.4	1.0
Nitrate-nitrogen, kg/ha.....	1	2	2	1
Extractable P, mg/kg.....	6	4	15	21
Extractable Zn, mg/kg.....	0.2	0.8	2.7	10.9
Extractable Cu, mg/kg.....	0.4	0.7	1.0	4.2
Cation exchange cap. (CEC)....	8.3	8.3	14.5	17.9
Soluble salts, mmhos/cm.....	0.1	0.1	0.1	0.1

Discussion

Compost incorporated into coarse-textured soils may improve the physical and chemical properties of soil (Pritchett and Fisher 1987). Previous research (Norland et al. 1994) near this site reported increases in percent ground cover and biomass using similar rates of MSW compost. The 90% cover standard was achieved on comparable compost treatments after 4 growing seasons, a similar result can be expected for this study. Garbage compost incorporated into sandy soil sustained the growth of pine trees for 16 yr without long-term deleterious effects to the ecosystem (Jokela et al. 1990). Trace metals contained in MSW composts will revert to less soluble forms upon incorporation, thus reducing the likelihood of plant uptake or leaching losses. In a companion study (Melchert et al. 1994), on a tailing site treated similarly with MSW compost, initial sampling indicated that nitrates were the only chemical that exceeded water quality standards. Nitrates decreased after subsequent sampling and are expected to diminish further.

Mining companies have had considerable difficulty meeting vegetation standards on coarse tailing sites. The addition of organic matter from compost may help sustain long-term plant growth on coarse tailing. Higher levels of organic matter can improve the water-holding capacity and maintain plant growth during dry periods. Taconite tailing are deficient in plant-available P. MSW compost applications can supply sufficient levels of P. The nutrient-supplying characteristic of compost may be similar to slow-release fertilizers.

The variability of MSW compost products is a problem (Walker and O'Donnell 1991). Levels of nutrients and contaminants can vary significantly among compost operations and between batches (Johnson and Crawford 1993). MSW compost must meet an acceptable degree of maturity. The use of immature compost can cause

N deficiencies, inhibition of seed germination, and an increase in the solubility of heavy metals. The compost used in this study was placed on the storage pad for 180 d, which is considered ample time for decomposition to occur to produce a mature and stable compost product. However, there was some odor associated with the compost which can be an indication of anaerobic conditions. Anaerobic conditions will inhibit the composting process. A conversation with the compost operator indicated that the compost piles may not have been "turned" enough to provide sufficient aeration for favorable composting conditions. Even though the compost may not have fully cured, no toxic effects on plant growth were noted from compost applications. MSW compost applied to tailing increased plant growth at increasing application rates. Compost quality is of primary concern for agricultural, landscaping, and horticultural uses. It may not be as important for use in reclamation. A lesser grade of MSW compost may be acceptable for reclamation use, particularly when mining sites are located away from populated areas. Developing standards of acceptance for MSW compost products should receive careful attention.

There is widespread agreement that municipal wastes should be recycled and reused. The manufacturing of compost from the biodegradable portion of solid waste is gaining attention as an alternative to landfilling. Interest in composting is growing in Minnesota, largely because of increasing land disposal costs (Wirth 1991). Composting is becoming cost competitive with other solid waste management alternatives. Opportunities exist in Minnesota for MSW compost use on mineland reclamation sites, although with a haul distance of over 150 km from the nearest compost facility, it may not be feasible.

The potential for using MSW compost on coarse taconite tailing sites is significant. There are over 3,000 ha of coarse tailing that could be reclaimed to meet reclamation standards using a MSW compost product. This could utilize over 270,000 Mg of composted MSW. The largest compost facility in Minnesota produces about 75 Mg/day of class I compost. This facility would have to produce compost for 10 y to satisfy the potential demand for all existing coarse tailing reclamation sites.

Conclusions

Using composted garbage on coarse tailing reclamation sites should be more environmentally acceptable and economically feasible than landfilling or other methods of disposal. The results of present and other studies, reveal that MSW compost can improve plant growth on infertile soils considerably.

MSW compost applications improved the soil properties of taconite tailing and increased the growth of ground cover and Souixland poplar trees. Trace metals introduced into the soil environment were not toxic or limiting to plant growth at application rates up to 89.6 Mg/ha.

Revegetation work on coarse tailing sites is not likely to succeed to meet reclamation standards without some organic amendment addition. MSW compost is potentially an inexpensive source of organic material for reclamation if a compost facility is located nearby. The use of composted MSW as a soil amendment is an appealing method for improving the growth of ground cover and trees on coarse taconite tailing reclamation sites without adversely affecting the environment.

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RECLAIMING ABANDONED MINING SITES: REURBANIZATION CONCEPTS AND EXAMPLES[1]

Dieter D. Genske[2] and Peter Noll[3]

Abstract: The Ruhr District in Germany, one of the oldest and largest hard coal mining regions in Europe, has been in a process of constant change over the last 40 years. Since the end of the World War II, the coal industry has been declining. There have been a number of successful attempts to establish new industries in the Ruhr District. However, the initiation of new industries depends strongly on the availability of space. Since the Ruhr District is a rather populated region, the only chance to gain space is to reclaim the areas formerly used by now obsolete industries. The remediation of former mining sites in the Ruhr District and the reestablishment of alternative industries have now become both a challenge for city planners and a prestigious attribute for ambitious politicians. It has become the declared goal of the German government to convert the Ruhr District into the greenest industrial region in the world. This paper discusses geotechnical techniques, as well as economical risks involved, in the recycling of abandoned mining sites. To demonstrate remediation techniques used in Germany, a recent remediation project is described and analyzed.

Additional Key Words: Recycling of derelict land, contaminated sites, remediation techniques.

Reactivation of Abandoned Sites

In North Rhine-Westphalia, Germany's most densely populated industrial province, a growing need for land for commercial and industrial use has become apparent. To meet the increasing consumption of open space, and faced with the need to protect open space, to assist cities and communities in their development, and at the same time to reactivate now-abandoned industrial sites, the province of North Rhine-Westphalia has resolved to make land reactivation a central goal of its land policy.

Strategies for the renewal of old industrial areas are being developed internationally and discussed intensively. The Ruhr area represents a particular challenge in this field.

On average, in North Rhine-Westphalia more than every other usable piece of land with a size of more than 5 ha is an abandoned industrial site. The areas of the abandoned sites range from 0.3 ha to more than 70 ha, with about 47% of the sites having an area of more than 10 ha. Corresponding to the industrial history and changes in the infrastructure in North Rhine-Westphalia and in the Ruhr area in particular, most sites were formerly mines, coke plants, or steel works. So, for example, in the eastern Ruhr area about 80% of the abandoned sites are old mines, and in the coming years further similar areas will be released.

The rehabilitation plans for the areas currently under development entaila future return to commercial or industrial use for 51% of the areas. 45% is allocated for parkland and recreation. The remaining 4% is to be used for future housing. The wide range of prior uses of the sites, their usually large area, and their often very long history of use are usually reflected in the extent of the problems of dealing with contaminated sites, which will be examined below.

Characteristics of Abandoned Sites

As stated above, the commercial, industrial, and transportation sites described have a number of features in common, even though they differ with regard to prior use, which creates highly specific problems regarding the handling of contaminated sites and demands specific procedures for so doing. They are, after all, not the typical degraded sites (such as gas stations) with one or a few contaminants, or large storage facilities or dumps with a more or less broad spectrum of contaminants but limited extent. Rather, they are mostly extensive, fully structured, and contaminated former industrial or commercial sites. The history of industrial or commercial use, often extending back for

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[2]Prof. Dr. Dieter D. Genske, Engineering Geology, Delft University of Technology, Delft, The Netherlands.

[3]Dr. Peter Noll, Managing Director, Entwicklungsagentur Östliches Ruhrgebiet, Bergkamen, Germany

more than 100 years, has left numerous traces on these abandoned sites, which are characterized by increased concentrations of contaminants in ground and surface water, sometimes considerable contamination of the soil, contaminated remnants of structural material, disorderly accumulations of production waste, extensive heaps of builder's rubble, mining material, slag, and ash. In addition, there are underground tanks, shafts, and basements of cooling towers filled with trash, production conduits that have not been removed and are still filled, as well as massive foundations and remains of former buildings. Not infrequently, one must reckon with surface contamination due to uncontrolled demolition.

The large-scale sites of the coal and steel industries are characterized by their extensive area and their usually vast heaps and dumps with enormous quantities of contaminated material. With the small-scale industrial sites (e.g. metalworking), on the other hand, the problem lies mainly in the complex spectrum of contaminants.

Abandoned sites like those described here must in every case be treated as areas of suspected old contamination. Such suspicion has proved to be justified in the overwhelming majority of the projects handled up to now.

Guidelines for Reclamation of Contaminated Sites

The contamination of soil and ground water has become a peripheral restriction on recycling. The process of recycling abandoned sites is hindered and delayed to a considerable extent by soil and ground water contaminations. Legal problems, the unavailability of suitable reclamation technology, inadequacies in assessing potential hazards, and, above all, the bottleneck in financing lead at least to time delays in the planning and execution processes.

However, the land recycling process cannot stop at merely removing hazards. Reclamation, and hence all strategic considerations for reclamation, must be oriented toward a demanding usage profile. Aspects of dealing with contaminations must be harmonized with marketing. There must not remain any incalculable risks for users or investors. Precautionary measures play a considerable role. The aim is a broad spectrum of uses, on the basis of city planning considerations and the site profile. Restrictions to this are set by limited finance and time and by technical insufficiencies.

Reclamation of contaminated sites, therefore, cannot result in the production of a piece of land uncontaminated and original as if untouched by humankind. After more than 10 years' experience in dealing with contaminated sites, we have realized that it is frequently impossible to achieve anything more than a reduction in contamination.

The authors' understanding is that reclamation measures are measures to ensure that a reclaimed contaminated site poses no danger to human life or health and no threat to the living or nonliving environment in connection with the existing or planned use of the site. The authors' understanding is also that the reclamation of contaminated sites must be designed to be as environmentally and socially acceptable as possible. As criteria for the environmentally acceptable reclamation of contaminated sites, the following apply especially:

- not transferring the problem to other environmental media or other locations,
- the reduction in environmental hazards at the site due to reclamation must exceed the environmental hazards caused by the reclamation,
- upon conclusion of the reclamation measures, the land should be suitable for a broad spectrum of uses.

Important aspects of a socially acceptable reclamation strategy are the intelligibility, practicability, and justification of evaluations and decisions, the openness of data, decisions, and plans, in short the "glass-clear reclamation". For socially acceptable reclamation, acceptance should be gained in particular by the people in the area.

Reclamation Management and Procedures

Besides the complexity of the task and the manifold interlocking of the different fields of activity in the process of recycling, it is frequently the shortage of time, the limited availability of funds, and the large number of projects needing to be worked on that make effective reclamation management -- integrated into the overall management of the project -- a necessity.

Experience in the Land Management Funds with the reactivation of contaminated abandoned sites shows that the successful completion of a project is possible only if:

- the reclamation process is completely and purposefully structured,
- the management is centralized in one person's hands,

- demolition and reclamation measures are implemented on the basis of comprehensive demolition and reclamation plans,
- the execution of measures is monitored and controlled by a reclamation supervisor.

Complete and purposeful structuring is the basis of any project planning. The reclamation process must be divided into sections that result in sensible interfaces and correspond to the planning process, e.g., in the working steps of other working areas. The structure should be chosen in such a way that the sections can be delimited from one another as to content; the units are small enough to be readily surveyed, planned, and controlled; and the intelligibility and reconstructability of the reclamation process are promoted.

In the Land Management Funds, it has proved expedient to prescribe a standard structure and to lay down binding rules for the content and sequence of decisionmaking and communication processes to facilitate internal and external coordination.

The procedure for the handling of contaminated abandoned commercial and industrial sites can be described in terms of the following working steps:

1. Determination of potential hazards
   1.1 Site investigation / present condition / prior use
   1.2 Assessment of hazard

2. Reclamation
   2.1 Investigation of demolition
   2.2 Planning of demolition and reclamation, environmental acceptability, safety at work
   2.3 Controlled demolition
   2.4 Decontaminating and improving soil and ground water
   2.5 Confirming success of reclamation / documentation / monitoring

3. New use and preparations
   3.1. Preparing the site
   3.2 New use

The actual order of procedure must, however, be worked out for each site from the specific circumstances; the strategy for investigating and reclaiming contaminated sites does not exist. Neither does the technology for carrying out reclamation, even though advertisements often try to persuade us otherwise. Each contaminated site is unique. Contaminated sites are "individuals" and thus need to be handled individually. They conceal surprises that we have not anticipated or prepared for and that necessitate measures specific to the site in question.

Reclamation Strategies

For the reclamation of a contaminated site, it is generally necessary to distinguish between the strategies of:

* use-dependent reclamation and
* contaminant-dependent use.

In the case of use-dependent reclamation, the reclamation requirements for the site are determined by the planners in light of the intended future use, and reclamation occurs in accordance with these goals. In the case of contaminant-dependent or contaminant-related use, the contamination of the ground is accepted as a restriction and the use is determined accordingly. Both strategies have in the past been employed by the Land Management Funds. Measures for contaminant-dependent use, i.e., adaptation of use to the contaminant situation, have probably composed the vast majority of cases. Thus, zoning as parkland, i.e., about 45% of the planned usage of the sites bought, generally involves highly or very highly contaminated lots.

This orientation of the Land Management Funds toward the strategy of contaminant-dependent use can be understood against the background of the general, original task and aim of the Funds, the limited availability of resources, and the almost complete absence of reclamation technology that prevailed up until the second half of the 1980's. It is necessary to examine this passive, reactive strategy from various viewpoints:

On the one hand, there is the development of awareness of the problem and of standards and requirements. On the other hand, there are new technical capabilities, improvements in methodological know-how, and experience from a multitude of projects that allow new possibilities to be contemplated. Above all, however, it is the increased demands on the sites, especially in the "Work in the Park" projects, which were partly implemented within the framework of the Emscher Park International Building Exhibition.

In the Land Management Funds, the development of reclamation goals and reclamation strategies -- in short, the reclamation process -- is closely linked to the development of city planning goals. Integrated reclamation and use plans, which come into being in an iterative, interdisciplinary planning process, have proved to be an effective instrument for the reactivation of abandoned sites.

Integrated Use and Reclamation Plans

Reclamation plans for land areas that are currently in use differ from those that are developed for abandoned sites in the framework of recycling. In the former, the existing uses and the safeguarding of these play a central role, and in the majority of cases preventive measures are the ones employed, i.e., hazard prevention by interrupting contamination pathways; but reclamation planning for abandoned sites or unused industrial or commercial areas can lead to more far-reaching solutions. To this end, the development of integrated use and reclamation plans is desirable. Such integrated use and reclamation plans are characterized by:

- Guaranteeing minimum legal standards and standards of environmental technology (interruption of contamination pathways etc.).
- Integration of decontamination/remediation and future use by dealing creatively with contamination situations and usage plans. As far as this is compatible with the usage plan, use sensitivities and contamination situations are matched up. As far as possible, remediation features, e.g., sealing, are harmonized with uses that also have a sealing character. Site sections that have been made safe should be used in such a way that if reclamation is later found to be necessary, they will not impede reclamation measures.
- Former abandoned sites should be used for residential purposes only if decontamination is carried out that involves complete removal of the contamination potential. Old sites that have not been merely made safe should not be zoned for housing.
- Besides the integration of future use, it is also necessary to coordinate the procedure for preparation and production, the location and removal of equipment, and finally the demolition or (preferably) dismantling of the industrial or commercial buildings, with the planning and execution of reclamation measures. Thus, the production of a load-bearing construction base over the entire surface (including the removal of cavities and foundations) results in different reclamation needs, a different reclamation strategy, and possibly a different investigation strategy compared with site-centred preparation.

Reclamation plans should allow for different solutions for different parts of the site. Different remediation measures may be envisaged for different parts of the site depending on the contamination situation found.

Reclamation Methods

It is possible to react in different ways to hazards and environmental effects emanating from contaminated sites:

- Protection measures and limitation of use of the site as a temporary solution until measures for remediation are carried out.
- Making safe as a measure to reduce or prevent the issuance of contaminants for a limited time by interrupting the contamination pathways.
- Decontamination aims at the final removal of hazards at the source and in the contaminated area by destroying the contaminants or permanently separating the contaminants phase by various technical methods.

The choice of the most suitable reclamation method depends basically on three factors:

1. Location, i.e., topography, environment, geology/hydrology, soil physics (substrate, permeability, pH value, etc.), planned use,

2. Contaminants, i.e., spectrum, concentrations, mobility, toxicity, medium, etc.,

3. Goods to be protected.

The use of reclamation technologies must also be preceded by careful and detailed planning and feasibility studies, which enable project-related decisions to be made in individual cases. In the comparative evaluation of reclamation methods, a variety of criteria are used, such as acceptability, effectiveness of the method, condition of the ground after decontamination, ease of monitoring and control, flexibility, persistence, quality, and quantity of residues, primary and secondary emissions (transfer of the problem), energy requirement, technical safety risks, stage of development and availability, operational experience, costs (process, associated, and consequential costs), complexity and operational safety, efficiency (duration of reclamation), space requirements, suitability for mobile equipment, requirement for authorization procedure, etc.

In connection with making-safe, the following criteria in particular should be investigated and compared: spatial and material effectiveness of barriers and hydraulic, pneumatic, and immobilization measures, information that allows assessment of the long-term behavior of potential contaminants, findings concerning the long-term effectiveness of the method, nature and extent of necessary auxiliary and monitoring measures, reparability of barrier systems, keeping open a wide range of reclamation options, influence on later changes of use, objective (by scientific criteria) and subjective (taking account of psychosocial factors) compatibility with the intended use, costs (including monitoring measures), requirement for authorization procedure.

The making-safe of abandoned sites can be considered when suitable material and (hydro)geological conditions exist, if the making-safe procedure is both objectively and subjectively compatible with the intended use. The interruption of relevant hazard pathways must be guaranteed with high probability. Measures for making safe have the character of temporary, intermediate solutions. This results on the one hand from the long-term behavior of potential contaminants and safety devices being often impossible to predict, and on the other hand from the provisional nature of the criteria that can be invoked for the decision to make safe. Procedures for making safe require continuous monitoring of their effectiveness. Further, it is necessary to check at regular intervals whether the state of technology and the hazard assessment findings permit continuing with the strategy of making safe. Changes of use may make it necessary to rethink. In the long term, therefore, reclamation of contaminated sites that have been made safe may prove necessary.

The advantages of making-safe procedures are that they are rapidly available, are frequently inexpensive compared with decontamination methods, are often implemented using simple and well-tried methods of building technology, and that authorization is usually unnecessary or easily obtained. Disadvantages are the limited availability of the sites and the consideration that further taxes may be imposed and reclamation measures may possibly become necessary at some future date. Making-safe procedures make it essential that knowledge of the contamination potentials and restrictions be guaranteed for the long term.

On the basis of the selection criteria for the individual reclamation technologies and considering the infrastructure of the abandoned sites, it is easy to conclude that making-safe procedures will find frequent application within the framework of recycling of land. Especially on the extensive sites of the coal and steel industries, most of which have widespread and sometimes deep-reaching soil contamination, a great variety of making-safe procedures are frequently employed. The use of decontamination procedures fails on these sites, mostly because of the insufficient availability of suitable techniques for the range of contaminants and the types of ground involved. In addition, these sites are usually characterized by enormous quantities of contaminated soil and other materials. The use of decontamination techniques, which are usually cost-intensive, would here exceed the bounds of any financial framework.

Abandoned small-scale industrial sites often have a broad spectrum of contaminants, but usually small quantities. In the framework of integrated use and reclamation plans, they demand a high level of creativity because the smallness of the area and the number of contaminated sections often prevent restorage of material on-site, decontamination methods are often ruled out by the types of contaminants, and adaptations of use are usually impossible. Here, different methods are frequently used for different parts of the site -- a combination of decontamination and making-safe procedures and restorage or dumping.

Case Study

The former mine and coking plant "Graf Moltke 3/4" is located in the northwestern part of the Ruhr District close to the city of Essen and covers an area of about 230,000 m^2. In 1873, the first shaft was excavated, with three more following within the next 30 years. A coking plant was built in 1903-04 followed by benzol and ammonia factories. Over the next 50 years, additional industries were established, turning the site into a multiuse industrial complex. Coal production went up to 1 million tons annually in the 1960's until the mine was closed in 1971.

In the following years, the Graf Moltke Mine became a typical industrial wasteland with all its negative attributes. The positive point, however, was the existing and still intact infrastructure of the surroundings and the

immediate neighborhood of the site to the A2 Autobahn, one of the most frequented German highways and an important east-west traffic connection, especially in light of German reunification. It was therefore decided to remediate the site as a project of the "Internationale Bauaustellung" and establish a high-quality industrial park. The European Community provided the project with appropriate funding (European Fund for Regional Development EFRE) so that only about 50% of the remediation costs had to be supplied by the owners of the former mining site.

Figure 1 illustrates the geology of the site. The subground of the Brauck site can be divided into three parts: an upper filling with foundation fragments (2 to 9 m), a middle stratum of quarternary sediments (about 10 m below surface), and a fractured cretaceous marl as bedrock. The ground water table is located at a depth of about 5 m.

A serious hydrocarbon contamination has been detected in the vicinity of the former coking plant (figure 2, upper graph). During the past decades, the pollutants have migrated through the porous sediments into the fractured rock, where they have been percolating through the joint network.

For the remediation of the "Graf Moltke" site, it was required by the environmental protection agency to minimize all excavation. Therefore, the sealing and reinforcing technique was chosen for the heavily contaminated sectors (figure 2, lower graph).

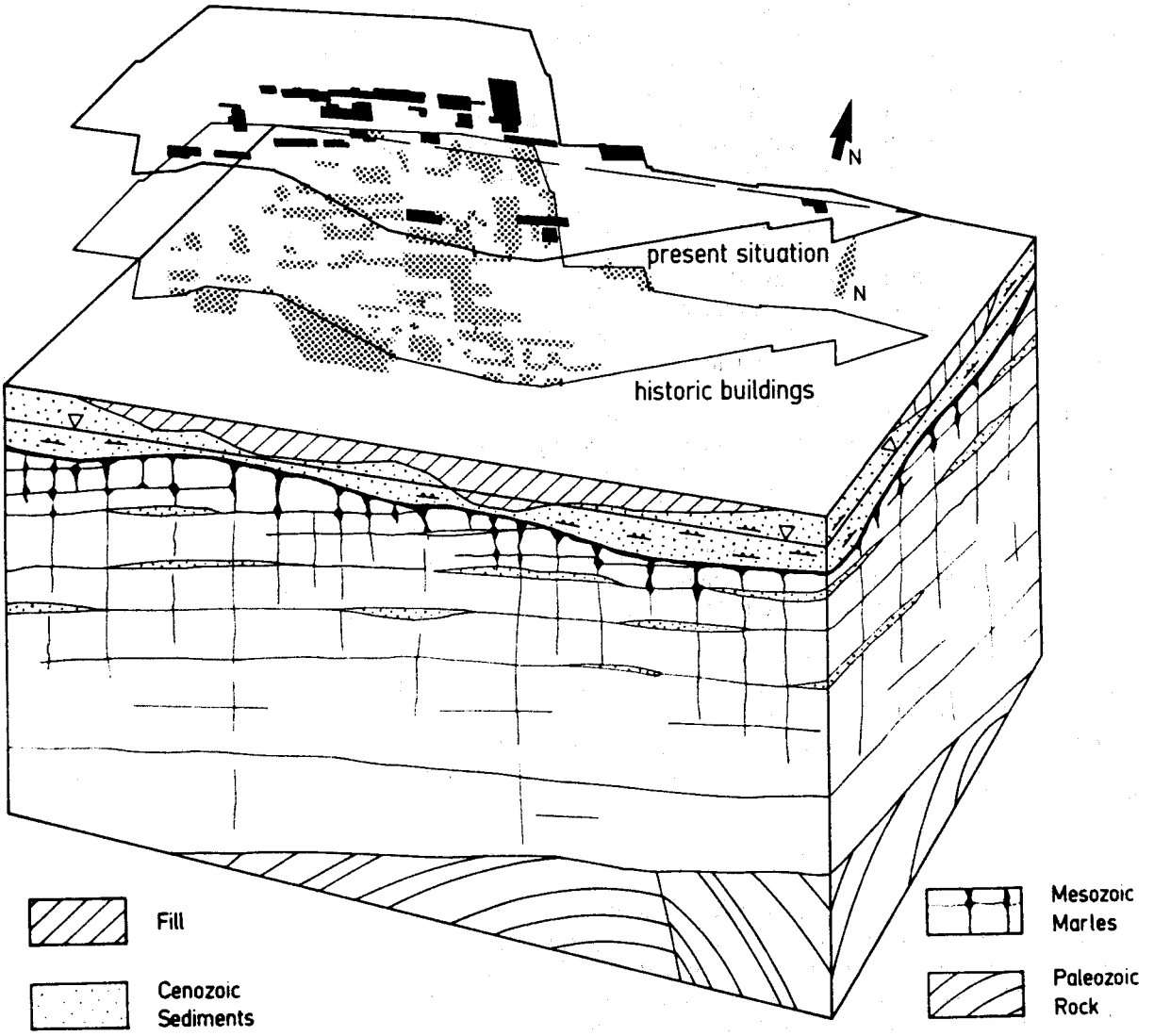

Fig. 1: Geology of the Brauck site.

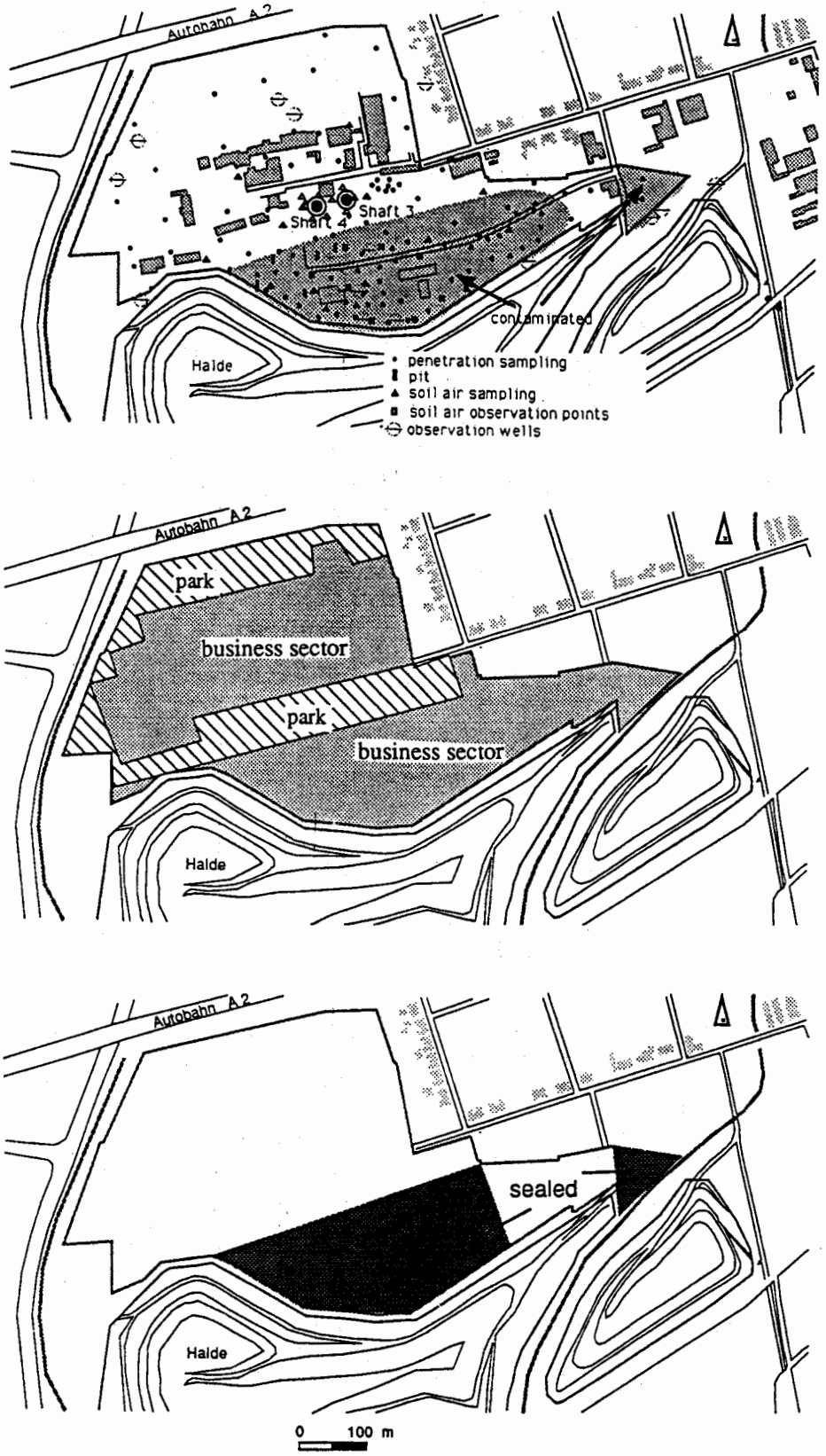

Fig. 2: Contamination, future land use, and remediation strategy for the "Graf Moltke" site.

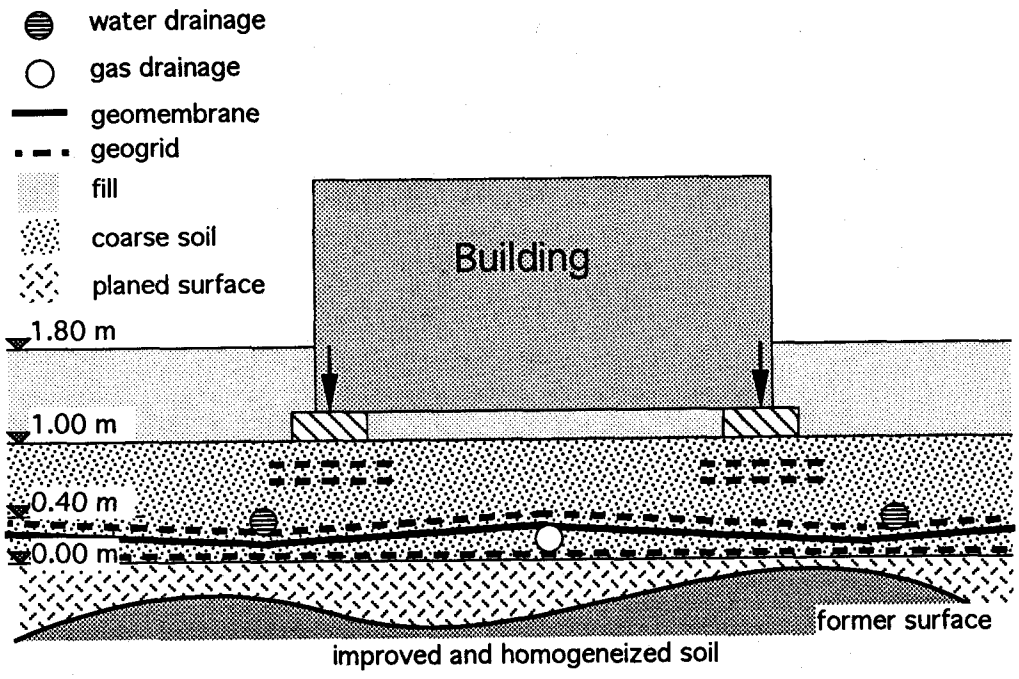

Fig. 3 A reinforced geotextile sandwich system. Besides the reinforcing geogrids, a geomembrane is integrated.

Figure 3 depicts a composite soil - geosynthetic - system that was designed to satisfy all three objectives mentioned above. Basically, this "sandwich" system is composed of three elements:

1. a lower supporting layer reinforced with geogrids,
2. a drainage and sealing system using geotextiles and geomembranes,
3. an upper reinforced foundation layer to account for structural loads.

Because the chosen technique is a rather innovative one, it was decided to design and construct a 500-m^2 full-scale test field on the site. The aims of the field test were to:

1. study the feasibility of using geosynthetics,
2. establish quality control tests and measures,
3. investigate possible damages during construction,
4. create new design criteria for such structures.

The test field location was chosen at an area within the derelict mining site where the ground conditions were representative for the whole site. To monitor the stress-strain behavior of the two reinforced layers using geogrids as reinforcing elements and the performance of the waterproofing geomembrane, several measurement devices were placed within the test field. A loading test was conducted to observe the stress-strain behavior within the sandwich system with special regard to the deformation of the sealing geomembrane. Also, the absolute and relative settlement at the surface were measured. The results from this test field appear to be promising. They are commented in Genske et al. (1993a, 1993b).

To monitor the quality of the ground water, a number of observation wells were installed, some of which may become recovery wells if a certain degree of contamination of the ground water is exceeded. The tolerable concentrations were negotiated with the environmental protection agency.

Conclusion

Recycling of derelict sites is one of the most challenging engineering tasks of the future. There are not many geotechnical problems that can be compared with the complex task of remediating contaminated sites. An interdisciplinary approach is needed to account for all the different aspects. Geotechnical engineers, environmentalists, politicians, financial experts, and the local citizens involved all must cooperate to ensure that the remediation project will be successful.

Each recycling project is unique. Too many aspects and boundary conditions give every site to be recycled a unique profile. This calls for individual concepts. In this paper, different aspects of land recycling are indicated. Aspects and guidelines as to land recycling are discussed, and several questions to be resolved are raised. A case study of a German derelict coal mine is presented, and the remediation strategy chosen is demonstrated.

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PROJECT D.I.R.T.: A LANDSCAPE ARCHITECT'S EXCAVATION[1]

Julie Bargmann[2]

Abstract: To a landscape architect interested in the creative manipulation of the land, mineral extraction presents a massive load for design exploration. The primary goal of Project D.I.R.T., Design Investigations Reclaiming Terrain, was to unearth the possibilities for renewing the perception of mining as creatively integral with the cultural and natural evolution of the landscape. The potential collaboration and planning inherent in the multi-disciplinary task of reclamation became evident on a journey to document selected mining sites around the country. Various coal, gold, copper, and taconite operations visited represent a diversity of regional landscapes with specific mining and reclamation practices. Both active and abandoned mined sites serve as case studies presented from a landscape architect's perspective. Examining the legal, social, and economic factors uncovered the frustrations with the legislation governing reclamation and the financial and ethical dilemmas of future land use for dependent communities. Reviewing the technical and ecological aspects revealed the innovations and progress of reclamation technology, but also limited application of ecological principles. These current practices of mining and reclamation illustrate a need to revise the legislation and coordinate an interdisciplinary effort to form truly productive and sustainable landscapes. Moreover, from this landscape architect's point of view, a synthetic approach guided by a broader vision of mining must lead to the creation of meaningful places that objectively reveal and celebrate the industrial heritage of the landscape. Mined lands can be an expression of a reciprocal connection between communities and individuals with the land one cultivates and all nurture. These monumental landscapes may come to represent a dynamic relationship between culture and nature for the next century.

Introduction to Project D.I.R.T.

The rapid development of mining and reclamation practices points towards an optimistic view of extracting important resources while constructing new viable landscapes. Although there are many shortcomings of the legislation and difficulties with the technology, mined lands are being restored to support wildlife and human activity. Yet these efforts rarely acknowledge the meaning inherent in transforming these modern industrialized landscapes.

"This brings us to the possibilities for the present and the future....we must stand at the moment in which humans may learn to be present to the Earth in a mutually enhancing manner" (Berry 1992). To reconcile and even embrace landscapes of production such as mineral extraction, our culture must learn to see itself as a part of nature. This emerging perception of interdependence between nature and culture (Gablik 1992) will guide our actions beyond "fixing" mined lands toward a visionary practice of constructing meaningful places for the present and next century.

With this vision for a renewed perception of the industrial landscape, Project D.I.R.T. was initiated. Focusing on mined lands, the goal of the study was to provide some insight into the limitations and possibilities observed in current practices of reclamation. This investigation attempts to combine practical considerations with idealistic projections for these landscapes of the future.

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[1]Paper presented at the International Land Reclamation and Mine Drainage Conference and the Third International Conference on the Abatement of Acidic Drainage, Pittsburgh, PA, April 24-29, 1994.

[2]Julie Bargmann, Assistant Professor of Landscape Architecture, University of Minnesota, Minneapolis, MN, USA.

A Landscape Architect's Perspective

The fundamental aspects of mining as massive landscape change represent a forum for all the concerns of a landscape architect. The cultural and ecological components of reclaiming land speak to the interdisciplinary nature of landscape architecture. The exploration of Project D.I.R.T. incorporated various viewpoints through interaction with individuals that affect, and are affected by, mining.

The perspective of a designer situates the activity of mining and resultant places into a history of landscape traditions. While addressing the complex realities of implementing significant changes in the landscape, a landscape architect strives to envision all the cultural and ecological implications. A designer synthesizes those realities and gives form to an overall vision for this new landscape. Project D.I.R.T. marks an initial effort to speculate how the evolving practices of reclamation and misconceptions of mining could be converted to a progressive means of building beautiful and inspiring landscapes.

The issues common to both landscape architecture and reclamation form a basis for the observations presented here. First, the legal, economic, and social factors include frustrations with mining and reclamation legislation, financial pressures of reclamation costs and the burdens of the bond, and conflicts with communities. Second, the technical and ecological practices comprise some progress and innovations, but reveal the need for revised regulations to allow for more appropriate and creative solutions. Finally, the potential cultural expression of these mined lands lies dormant, ripe for new interpretations.

Journey to Unearth the Issues

The primary component of Project D.I.R.T. was a series of site visits to mined land in several regions of the United States. Locations included the Iron Ranges of Minnesota, South Dakota, Wyoming, Utah, Arizona, Virginia, Kentucky, Indiana, and Illinois. Sites were selected for representation of different mining operations (coal, gold, copper, and taconite) in particular cultural and ecological contexts. Varying perspectives were presented by the individuals ranging from personnel of mining companies and government agencies to citizens and professionals involved with community groups and university research.

Rather than presenting a chronological documentation of the site visits, this paper draws from the key examples to illustrate my observations. This exploration of the issues is not intended to judge the practices of mining and reclamation witnessed. Rather, the inquiry raises questions of how the collective effort of reclaiming land may progress and how the renewed perception of mining resources might unfold.

Legal, social, and economic factors of the mining industry were felt most acutely in the southeastern area of Appalachia. the mountain top removal activities of companies like Martiki Coal Corp. in Lovely, KY had obvious implications for the economics of the communities with planning of secondary land uses. The successful farming operations demonstrated on the Martiki mine site were impressive and admirable. Questions came to mind though of the development capacity of the region. It is difficult to speculate if a community dependent on the mining industry for many decades could embrace a new livelihood. Land uses in this case cannot simply be applied, but need to emerge from some keen understanding of the economic and social constructs of the associated community. Assessing a genuine need (Zipper[a] et.al. 1989) would enable members of a community to both understand and support activities like those witnessed at the Martiki mine.

A visit with a member of the Kentuckians for the Commonwealth revealed a lack of understanding and/or communication between the industry and the community. Adamant about protecting his home of many family generations, surrounded by the beauty of the "holler" of his native landscape, this gentleman was perplexed by politics that allowed for exploitative mining practices. Meeting this concerned citizen painfully pointed to the need for cooperative interaction between the industry, government agencies, and the community.

A great example for the common ground established for all those involved in and affected by mining was seen at the Powell River Project in Wise County, VA. Here the research on the technology of coal mining and reclamation is conducted and the results are disseminated. The mining industry and government agencies have a forum to evaluate regulations. Most importantly, the community is educated about mining as a resource for the future of their homeland (Zipper et.al. 1992).

Research at the Powell River Project also addressed another common dilemma of the limitations of the legislation, in this case the Surface Mining and Reclamation Control Act. A proposed and tested alternative to returning land to the "approximate original contour" (AOC) has proved to have both economic and environmental benefits (Zipper[b] et.al. 1989). Many other instances such as erosion problems on AOC slopes illustrate shortcomings of the regulations. These Federal laws mandate the creation of uniform and generic "natural" landscapes that are often dysfunctional land with green veneers. The regulations seem to discourage the advancement and application of the technology that could produce ecologically rich landscapes.

Technical and ecological issues of the quality of the reconstructed mined land were illustrated by the many efforts to revegetate and recontour. The predominant feeling from several site visits was the pressure to stabilize slopes and make them "green." The regulations for a minimum percent of vegetative cover seems to favor quantity versus quality.

Reclaimed fields at a gold mine in Lead, SD that were a rich mixture of native species were not considered by company officials as lush and acceptable as fields of ephemeral crops such as alfalfa. Test plots and reclaimed tailing ponds in northern Minnesota were planted with non-native species that could invade surrounding disturbed or developed land. These invasive species also do not allow for processes of succession and colonization to establish fields of diverse plant habitats (Winterhalder 1993). Regulations often prevent the sculpting of habitats which create unique micro climates for multiple, and sometimes rare, species (Wade and Thompson 1993).
This skin-deep, quick-cover approach ignores the flesh and blood of the landscape. Lack of creative consideration for landform structure and water systems seems to result from the limited ecological bases of the regulations. Often pressure from a company unwilling to challenge the rules prevented any exploration of the integration of ecological principles with reclamation practices. A reclamation ecologist at AMAX Coal Co. in Gilette, WY expressed her frustration at the inability to incorporate landforms and water bodies that would enrich the ecosystem of the reclaimed land. Even though scoria outcrops and playas are indigenous characteristics of that natural landscape, the regulations and fear of prolonged bond release erased the hope for any shaping of a site specific, ecologically rich reconstruction. Mining and reclamation must not be practiced and viewed solely as extracting valuable resources, but also as constructing integrated or new ecosystems (Wade 1988).

Mining as a Creative Process

The act of mining must be regarded as a resource. Laws must allow flexibility for testing new technologies with ecological bases while also holding mining companies accountable for the quality of the reclaimed land. The understanding of specific landscape structures must allow building upon or contributing to existing systems. Mining processes unique to each location must be allowed to enter the equation.
Often the dynamic forms produced by mining are sacrificed to regulations that mandate uniformity. Federal legislation is directly counter to the idea of the "genius of the place." Despite the fact that mining produces extremely dramatic forms, the reclaimed result is often a generic, romanticized notion of the "natural" landscape. "The boldness of the operation, the ruggedness of the rock formation, the interplay of water, rock, and land are sterilized by the fear and image of mining and by the regulations designed to 'protect' the environment. The art, that is to say, the texture, form, pattern, contrast, focal point, accent, and uniqueness of a place is buried by the 'rules'." (Bauer 1993).
In many cases, when the mined landscape does not exhibit any structural, environmental, nor safety hazards, consciously and responsibly choosing to <u>not</u> reclaim the land could be a viable alternative. Revised regulations may someday acknowledge where forms created by mining provide wholly new landforms or add to the diversity and economy of a region. On some unreclaimed coal fields of Indiana, housing developments are benefiting from the new water bodies and rock formations of endcuts and highwalls. Combined with the natural lakes of the Iron Ranges of northern Minnesota, numerous pit lakes provide recreation areas and wildlife habitat. Development of these unique mined landforms must be allowed to contribute to the cultural heritage of their contexts.

Mining as Cultural Expression

The opportunity to explore mined lands as expressive representations of our culture remains barely unearthed. The Project D.I.R.T. site visits along with studies of natural, historical, and contemporary landscapes lead to some speculation of how we can proactively engage with monumental landscapes and realistically and progressively redefine our relationship with Nature.

Drawing analogies to monuments of the natural world reframes the way we may perceive the dramatic processes of mining. The mine pits and lakes of northern Minnesota can appear as powerful as places like Canyon de Chelley in the Southwestern United States. The terraced stone formations of Monument Valley strike a strong silhouette similar to waste rock piles of taconite mines.

Many ancient precedents inform a new perception of mined lands. Analagous because they embody the ambitious and willful human act of land shaping, prehistoric iron age hillforts in Great Britain bear an uncanny resemblance to the lifts and benches of tailings impoundments. As evidenced by an excavated stone amphitheater dating to about 500 BC, the Etruscans were practicing "progressive reclamation" when they mined the tufa for their building while carving an important civic gathering place. Seen in this light, cannot the Berkeley Pit be seen as the heart, not the hole, of Butte, MT? These historical precedents are revered as emblems of their time and beg the question whether mined landscapes can also be considered mystical landmarks of our culture.

Contemporary experiments with reclaimed land represent a range of design responses. Many artists have attempted to address the complex issues of mining and proposed alternatives for building monuments of the industrial landscape. Some of their work has revealed the difficulties and dilemmas of reclamation.

Michael Heizer sculpted Effigy Mounds on abandoned coal spoils at Buffalo Rock State Park in Ottawa, IL. Although conceptually compelling, the giant earthforms are now disintegrating from poor reclamation techniques. It is also questionable how these insect-shaped mounds, visible primarily from the air, contribute to the community's park experience and the understanding of the vision of this new type of landscape.

In a symposium on land reclamation as sculpture, Robert Morris issued a powerful warning that reclamation can often amount to a clean up service wiping out "technological guilt" (Morris 1979). His executed piece portrays both the destructive results and the creative forms of a mined pit, a juxtaposition that is a powerful reminder of the dual forces at work in the act of excavation.

Before his early death, the sculptor Robert Smithson was posing the most prophetic questions about the relationship of nature and culture. Delving into the geological and industrial underpinnings of mineral extraction, Smithson postulated that industrial landscapes were the place where "remote pasts meet remote futures" (Smithson 1979). His proposals for the Bingham Copper Pit in Utah and tailing ponds in Colorado re-presented these landscapes as monuments of our time and culture.

Current projects dealing with industrial landscapes are reconsidering these places as an integral part of the public realm. In this context they can act as forums of community education and interaction. Seen as civic monuments, these places can embody the celebration of our industrial heritage. An architecture critic, speaking of a recently built example, wrote: "If we're willing to troop up to the American Museum of Natural History to look at the shattered pots of ancient civilizations, why should we profess revulsion at the sight of our own archeology in the making? At a time when people recognize the critical need to establish more harmonious relations with the earth, why shouldn't there be a place we can go to see how that goal might be attained?" (Muschamp 1993).

New Practices, New Perception

The issues of reclamation are difficult and complicated. The perception of mining continues to be misguided. Yet these factors should not discourage the pursuit of an interdisciplinary effort toward a progressive practice and renewed vision of mining. Mining companies need incentives to do more than just stabilize lands with green veneer. Reclamation laws must allow for flexibility to build ecologically rich new landscapes. Interaction with the community must allow for an understanding of how the landscape change can positively reshape their lives and the lives of the next generation. All efforts must build toward visionary landscapes that represent a reciprocal relationship of nature and culture.

There are many years ahead of reclaiming thousands of acres of abandoned mines. Lessons from reckless mining practices are restructuring processes that incorporate precautions to protect the environment while harvesting resources. We should not be ashamed of mining as a landscape of production, which can be viewed as similar to the necessary and beloved activity of farming. If the practice of reclamation is associated with cleaning up after irresponsible mining processes, I look forward to the day when we can retire the term reclamation. Instead, we O hope we will see a time when the word mining will evoke images in everyone's mind of productive and beautiful landscapes. The pride in the act of mining can unearth new possibilities for how we shape our landscapes of the future.

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INFLUENCE OF GRADING INTENSITY ON GROUND COVER ESTABLISHMENT, EROSION, AND TREE ESTABLISHMENT ON STEEP SLOPES [1]

John L. Torbert and James A.Burger [2]

Abstract: In 1991, a 5-year study was established in eastern Kentucky to evaluate the effect of surface grading intensity on ground cover development, erosion, and tree establishment. The study included three grading treatments. The "moderately" graded treatment consisted of backfilling a 40% slope and back-blading with a Caterpillar D-9 bulldozer to create a smooth slope. The "intensively" graded treatment involved additional back-blading passes, followed by "tracking-in" with D-9 bulldozers. The "roughly" graded treatment was created by ripping plots with a 36-in ripping bar pulled directly downslope with a D-9 bulldozer. Each plot (150 by 150 ft) was installed on the slope, with a level area at the base of each plot. Five species of trees were planted in each plot (white pine, loblolly pine, sycamore, sweetgum, and yellow-poplar), and the entire area was hydroseeded with a "tree-compatible" ground cover. Erosion was measured monthly using erosion rods. Ground cover was measured along a transect after the first growing season. Total ground cover averaged 82% after 3 years and was not affected by grading treatment, although there was a significantly higher amount of legume cover on the roughly graded plots. Erosion was highest on intensively graded plots. Tree survival varied by species. Average survival was best for sycamore (83%) and poorest for white pine (11%). Growth of sycamore and loblolly pine was significantly reduced by the intensive grading treatment.

Additional Key Words: reclamation, compaction, productivity.

Introduction

Our reclamation research in the central Appalachian region of Virginia, West Virginia, and Kentucky has caused us to conclude that the single most important factor limiting forestry land use opportunities on mined land is mine soil compaction, specifically, compaction caused by grading the final surface and "tracking-in" with bulldozer treads to create a seedbed. These conclusions are based partially on a compaction study that was maintained for 2 years before the study site was lost to remining (Torbert and Burger 1990). We find this compaction to be particularly disturbing when forest land is the specified post- mining land use. Smooth surfaces (free of boulders, depressions, and gullies) are desirable for hayland or pasture, but are not typical of natural Appalachian forest land, and are not beneficial to forest productivity or forest management activities. We believe coal companies could save money and increase land use productivity by reducing their grading activities on reclaimed forest land. We also believe that creating a productive forest ecosystem is the best way to provide long-term environmental benefits.

The reasons that intense grading practices persist on land reclaimed with trees are understandable. Coal companies usually do not own the land being mined, and consequently, they have no financial interest in the long-term implications of grading impacts on forest productivity. Coal company reclamationists have a

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[2] John L. Torbert, Research Associate, and James A. Burger, Associate Professor of Forest Soils, Virginia Polytechnic Institute and State University, Blacksburg, VA, USA.
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corporate mandate to reclaim the land and achieve bond release as quickly and efficiently as possible. As long as reclamation inspectors expect coal companies to construct smooth surfaces and establish a vigorous ground cover dominated by Kentucky-31 tall fescue, there will be no incentive for coal companies to modify their grading practices, despite pleas from landowners to maximize the productivity of reclaimed forest land. It is also understandable that regulators will have little reason to discourage intensive grading until its adverse effects on tree growth are clearly demonstrated by research. Regulators also need to be convinced that less grading to improve forest productivity will not compromise environmental protection.

To address the issues of grading effects on forest productivity and environmental protection, a 5-year study was established in 1990 to evaluate the effect of reclamation grading practices on ground cover development, erosion, and tree growth. The purpose of this paper is to provide third-year results of the study.

Methods

Site Selection and Treatments

The study is located on land mined by Martiki Coal Corp. near Lovely, KY. A slope, approximately 150 ft (upslope) by 1,500 ft (along the contour), had been backfilled and moderately graded to its final contour (40% slope). 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 (150 by 150 ft) which were used to accommodate three replications of three grading treatments. The three grading treatments, installed on March 26, 1991, were

1. "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 small boulders and small gullies were present. Since 6 months elapsed between the times these sites were graded and seeded, a hard-crusted surface had developed.

2. "Intensive" grading. This treatment is typical of operational reclamation practices in the region. The purpose of intensive grading is to eliminate small boulders and gullies such as those that existed in the moderately graded treatment, and to break the surface crust and create a better seedbed for ground cover germination. Bulldozers (D-9 Caterpillars) smoothed the slopes by dragging their blades as they backed downhill, after which they tracked-in the surface by running up and down the slope until the entire surface was covered with grouser tracks from the bulldozer treads.

3. "Rough" grading (or "ripped"). The initial study design called for a rough-graded treatment where the slope had been returned to its approximate original contour by backfilling with a minimal amount of grading. Such a site would have an uneven surface with boulders and depressions, and a loose surface soil. Unfortunately, when selected, the study site was already beyond a roughly graded level to the moderately graded treatment. To create a rough surface with loose soil, the mine soil was ripped with a 36-in subsoiler pulled down the slope with a D-9 bulldozer. Rips were created at 10-ft intervals. On the level area at the base of the slope, rips were installed perpendicular to the slope. Ripping created a very rough surface by pulling large boulders to the surface and creating gullies and ridges of loose soil aligned up and down the slope.

Tree Planting

Trees were planted on April 1 and 2, less than one week after installation of the grading treatments. Five species of trees were planted: white pine (Pinus strobus), loblolly pine (Pinus taeda), yellow-poplar (Liriodendron tulipifera), sweetgum (Liquidambar styraciflua), and sycamore (Platanus occidentalis). Approximately 40 of each of the pines and 20 of each of the hardwood species were planted in each plot. All species were 1-year-old seedlings except white pines, which were 2-year-old seedlings. Trees in the roughly graded treatment were planted in the gullies created by ripping. After the third growing season, tree heights

were measured and survival rates were determined.

Ground Cover Establishment

On April 16, 2 weeks after tree planting, a "tree-compatible" ground cover was established by hydroseeding (table 1). After the first and third growing seasons, three 100-ft transects were established in each plot to measure ground cover. These 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 2-ft intervals along the transect, a 1-in diameter sighting tube was used to assess ground cover. If more than half the area observed through the tube consisted of bare spoil, the point was tallied as such. If more than half the area was covered with vegetation, the existing species were tallied. This was done at 150 points per plot.

Table 1. Ground cover species and seeding rates.

Species	Application rate lb/acre
Winter rye [1/]	10
Perennial ryegrass	5
Orchard grass	5
Kobe lespedeza	5
Appalow lespedeza	5
Birdsfoot trefoil	5
Redtop	3
Weeping lovegrass	3
Ladino clover	3
Crown vetch	1

[1/] Foxtail millet (5 lb/acre) would have been used instead of winter rye, but was unavailable at time of seeding.

Erosion Measurements

Soil movement from the slope and deposition at the base 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 ten metal rods (spaced 10 ft apart) 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. Another row was placed at the base of each slope to measure soil deposition. Rods were measured in October 1991 and 1993, after the first and third growing seasons.

Results

Site Description

The study was installed on a steep slope where the likelihood for erosion effects would be greater than on a level or slight slope. The spoil on the site was derived mostly from gray siltstone with a minor component of gray and brown sandstone. The site was not topsoiled. Spoil samples collected from each plot prior to hydroseeding had pH values ranging from 7.7 to 8.8. All plots had a south to southeast aspect, which, combined with dark spoil, resulted in high surface temperatures during the summer. The climate at this site was very dry during 1991. This site received virtually no rainfall from July through September during the first year. Rainfall that did occur in the early summer came as short, intense storms.

Ground Cover Establishment

The species hydroseeded in this study were mostly acid-tolerant, short- statured species (Vogel 1981) which have provided good cover without overtopping trees in other studies (Torbert et al 1991). Grading treatment had no effect on ground cover during the first 3 years (table 2). Average ground cover after the first year was 44% and increased to 82% after 3 years. Most of the cover in the study resulted from weeping lovegrass, which

was virtually the only species surviving on some of the moderately and intensively graded plots after the first year. The average amount of legume cover in the study was 12% after the first year and 34% after 3 years.

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 vigorous cover that existed within the "troughs" of the ripped gullies and the relatively sparse cover that existed between rips.

Table 2. Ground cover (%) after three growing seasons as affected by grading treatment.

Grading treatment	Total cover,%		Legume cover,%	
	average	range	average	range
Rough	82a	75-89	45a	28-79
Moderate	83a	54-99	29a	5-47
Intensive	82a	64-92	28a	15-42

Values within a column followed by the same letter are statistically similar according to Duncan's multiple range test ($p<0.10$).

Mine Soil Erosion

Compaction on intensively graded plots reduced water infiltration and increased surface runoff. During a heavy storm in the early summer of the first year, surface runoff was observed flowing from the intensively graded plots. On ripped plots, water flowed across the surface of the compacted area between rips, but within the area disturbed by the rips, high infiltration rates prevented any overland flow. At the bottom of the slope, no water flowed from the gullies created by ripping.

Erosion was expressed as the average depth of spoil lost at the 30 erosion rods located on each plot and as the amount of deposition at the base of the slope and 20 ft from the base (table 3). There was a large variation in erosion measurements among treatment replicates that precluded any statistically significant effects. After the first year, the average soil loss from the intensively graded plots was 18 times greater than the loss from the roughly graded plots. Statistically there was a significant increase in soil deposition at the base of the intensively graded plots during the first year.

Erosion measurements after the third year produced some confusing results. Based on the length of metal rod above the soil surface, there would appear to be a net deposition of soil on the rough-graded and moderate-graded treatments. By comparing third year measurements with first year measurements, there appears to be an average net deposition of 0.65 cm of soil in this study. This of course is not realistic, although some deposition may have occurred at the base of some individual rods (especially in the ripped treatment where some rods were located in the trough of a rip). Some soil swelling may have occurred during the winter as a result of freeze thaw, which decreased the length of the rod remaining above the surface.

A comparison of rod measurements at the base of the slope revealed that the intensive treatment had resulted in the most deposition, but again a meaningful comparison with first-year results is not possible because of changes in surface soil density that may have resulted from freezing.

Table 3. Average soil loss from study plots and deposition at bottom of plots after the first and third year as affected by grading treatment.

Grading treatment	Soil Loss from slope, cm		Soil Deposition at toe of slope, cm	
	Year 1	Year 3	Year 1	Year 3
Rough	0.04	-0.38	1.51ab	1.84
Moderate	0.44	-0.39	0.50b	0.85
Intensive	0.72	0.0	3.48a	2.21

Values within a column followed by the same letter are statistically similar according to Duncan's multiple range test ($p<0.10$).

Tree Survival and Growth

The original study design involved only loblolly pine and white pine. These are two commercial species commonly planted on mined land in the region, and they are species that the landowner (Pocahontas Land Corp) intends to manage for sawtimber production. Unfortunately, the average pH of this site was greater than 8, which is much higher than the acidic conditions to which pines are best adapted. Consequently, the hardwood species were added to the study.

As a group, the hardwoods survived better than the pines (table 4). White pine survival was very low. Only 11% of the seedlings survived, and very few surviving trees appeared to be healthy. Loblolly pine did better than white pine, and survival was 51%. Loblolly pine growth was not good, however. Loblolly pine seemed to suffer from a high soil pH and/or high soluble salt concentrations. Most loblolly pines had yellow foliage with orange coloration at the tips. Loblolly pine growth was significantly reduced by the intensive grading treatment.

Survival was highest for sycamore, and survival was significantly reduced by intensive grading. Survival in the intensive-graded treatment was 64%, whereas survival in the moderate and rough treatments was 91% and 94%, respectively. Average height for sycamore and yellow-poplar was signifcantly lower in the intensive-graded treatment.

Summary

Reclaimed surface mines in the central Appalachians are routinely graded and tracked-in to create smooth surfaces. Some people have argued that intensive grading operations are needed to establish uniform ground covers and reduce erosion. Others have argued that such an approach is short-sighted and that the greatest erosion control in the long term will result from efforts that promote the establishment of a healthy forest. This study was established to provide some information about the short- and longer term impacts of grading on environmental protection and forest productivity. Third-year results show that intensive grading did not result in better ground cover establishment or erosion control. In fact, erosion was highest on the intensively graded plots.

The conditions created by ripping the sites were completely opposite to conditions generally strived for during reclamation. The ripped treatment did not result in an aesthetically desirable smooth surface. There were many rocks and depressions created by ripping. To the surprise of many people, however, the gullies created

by ripping did not cause problems with slope stability or erosion. Instead, the loose surface soil created by ripping enhanced water infiltration, facilitated ground cover establishment, and promoted better tree growth.

Table 4. Third year survival and height as affected by species and surface grading treatment.

	Rough graded		Moderate graded		Intensive graded	
Tree species	Survival, %	Height, in	Survival, %	Height, in	Survival, %	Height, in
Sycamore	94a	88a	91a	57b	64b	48b
Sweetgum	75	35	76	37	75	29
Yellow-poplar	69	50a	64	45ab	67	35b
Loblolly pine	58	58ab	40	67a	55	46b
White pine	8	26	17	29	8	26

Values for survival or height within a species/treatment combination followed by the same letter are statistically similar according to Duncan's multiple range test ($p<0.10$).

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PHYSICAL CRITERIA FOR THE DESIGN AND ASSESSMENT OF RESTORATION SCHEMES IN THE UNITED KINGDOM[1]

R. Neil Humphries[2] and Gordon E. McQuire[3]

Abstract: The restoration of colliery wastes and open pit coal sites in the United Kingdom (UK) is undertaken according to a land use strategy plan and detailed specifications that have been agreed upon with the planning authorities.

For two of the major land uses in the UK, agriculture and forestry, data on physical criteria (climate, site features and soils) are available to assist in the planning and design of land use strategies and specification of restoration treatments. Similar criteria could also be developed for the restoration of semi natural vegetation and habitats for landscape, wildlife, and amenity uses.

Three examples are described illustrating the use of the physical criteria in the design of schemes, the specification of treatments, and the assessment of achievements.

Additional Key Words: restoration strategy, restoration treatments, physical criteria, climate, site features, soils.

Why Criteria?

The restoration of colliery lagoons and spoil heaps, and of open pit coal sites in the UK is undertaken according to a land use strategy plan and detailed specifications. These are usually prepared by the industry in conjunction with consultants. The restoration plan shows the location and types of land use and is accompanied by detailed specifications for all aspects of the treatment and management work to be used to achieve the strategy. Both the strategy and the specifications have to be agreed upon with the local government planning authorities. Usually the strategy is agreed upon during early consultations and the application stage of the process. In long-term developments, the principles of the strategy may be outlined and, by agreement, modified according to changes in local and national land use policies. The details of the specifications may be agreed upon at the same time or delayed until just before restoration commences.

In the recent past, the strategies and specifications formulated have not been based on specific criteria for the land uses and vegetation types, but have been governed largely by land ownership, land use aspirations, and availability and allocation of soil material. While this approach has often been relatively successful for agricultural uses, it has been largely by default, and associated with those sites where the original soil types and thicknesses have been replaced. It has been less successful in the case of other land uses, and woodland in particular, where there has often been an insufficient cover of soil materials. This often occurs at older redundant collieries where, for historic reasons, soils were not recovered prior to development and waste disposal (Humphries 1984). In the past the available soil materials have preferentially been allocated to agricultural uses, with little or no cover material for the other land uses. The reasons for this have been multifarious, but have mainly been based on the misguided assumption that non agricultural land uses and vegetation types are less demanding in terms of "fertility" etc.

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[1]Paper presented at the International Land Reclamation and Mine Drainage Conference and the Third International Conference on the Abatement of Acidic Drainage, Pittsburgh, PA, April 24-29, 1994.

[2]R. Neil Humphries, Technical Director, Humphries Rowell Associates, Loughborough, UK.

[3]Gordon E. McQuire, Chief Civil Engineer, British Coal Corporation, Eastwood, Nottinghamshire, UK.

From an operational viewpoint the past approach has generally specified universal treatments irrespective of site constraints, proposed land use type and quality, and the vegetation's requirements. This blanket approach to restoration has tended to result in unnecessary and costly work such as more stone removal, greater thicknesses of soil cover, and additional cultivation, which are not economically justified owing to physical limitations on the potential quality and ability to utilize the land.

Both of the above criticisms have arisen because of the absence of accepted criteria against which strategies can be formulated and tested, and restoration treatments ultimately specified.

The lack of criteria has also made it difficult for others to objectively assess restoration achievements in the past (Humphries et al 1984). This evaluation aspect of restoration has become increasingly significant now that most coal projects in the UK are subject to environmental impact assessments under recent legislation (Department of the Environment 1989a). The ability to restore to specific land uses and qualities has now become a key issue in the planning process in the UK. Hence, there is a need for accepted criteria against which proposals and achievements can be evaluated. Furthermore, the Town and Country Planning (Minerals) Act 1981 (Department of the Environment 1982) requires land to be restored to physical characteristics that are suited to the proposed use, whether it be agriculture, forestry, amenity, or wildlife. However, no guidance on the characteristics to be used or the standards to be achieved is given in the Act, and such guidance is only cursorily referred to in respect of agriculture in more recent literature (Department of the Environment 1989b).

The purpose of this paper is to draw attention to the fact that for two major land uses, agriculture and forestry, physical criteria are available and these can be adapted to other uses. For example, the criteria for forestry are also applicable to semi natural broadleaved woodland for landscape, wildlife, and amenity purposes. Other vegetation types could also be based on the same framework, although there will need to be some development work.

Available Criteria

Published Criteria

There are published physical criteria whereby land in the UK can be assessed for its suitability for agriculture and forestry use (Bibby 1982, Bibby and Futty 1988, MAFF 1988). These methodologies have their roots in the system used in the USA in the 1960's (Klingebiel and Montgomery 1961). According to the criteria, land in the UK can be grouped into broad categories of potential land use and quality for both agriculture and forestry (table 1). The groups represent a gradation of productivity, crops and/or species, and flexibility of utilization from category 1 (no or minor limitations and a wide species range) to 5 (severe limitations and a very restricted species range).

Physical Criteria

Agricultural and forestry productivity and utilization in the UK are largely determined by climate, relief, soils, and their interactions. Hence, climate, site features, and soil characteristics need to be taken into account when designing restoration schemes and specifying treatments.

Table 1. Categories of potential land use and quality.

Category	Suitability for agriculture	Suitability for woodland
1.....	No limitations, very wide range of agricultural and horticultural crops, high yields and low variability.	Very wide range broadleaved + conifers.
2.....	Minor limitations, wide range of agricultural and horticultural crops, high yields but may be variable.	Wide range broadleaved + very wide range conifers.
3(a).....	Moderate limitations, narrow range of arable crops and less demanding horticultural crops, moderate to high yields.	Moderate range broadleaved + wide range conifers.
3(b).....	Moderate limitations, narrow range of arable crops, moderate yields, but high yields of grass.	Moderate range broadleaved + wide range conifers.
4.....	Severe limitations, suited mainly to grass, yields moderate to high.	Restricted range broadleaved + moderate range conifers.
5.....	Severe limitations, suited only to grass.	Very restricted range broadleaved + restricted range conifers.

Climate. Climate has a major influence on the range of agricultural uses (grassland, arable, and horticultural), costs, and levels of production. In many places in the UK, it can be the overriding factor; that is, climatic factors can determine land use irrespective of the nature of the soil profile. Climate is a major consideration for afforestation with broadleaved and conifer species; including determining the species planted and their productivity and form. It is also a major factor in determining other wildlife habitats such as wetland and heathland by influencing species composition and competitive ability. Climate is also of major landscape significance, influencing the type and form of vegetation and rates of development. For amenity use, it is of influence through resilience and recovery from damage to vegetation and soil. The effects of climate are partly direct through rainfall, temperature, and exposure to wind and frost, and partly indirect through soil wetness and droughtiness. In this respect, the duration of winter soil wetness is of particular importance.

The physical criteria for differentiating between categories of land use and quality in the UK on account of climate are given in table 2.

Site. Site features, such as gradient of slopes and general topography, which affects the incidence of flooding and risk of soil erosion, also influence land use and vegetation types. These too can be overriding factors but are usually secondary to climatic limitations.

The incidence of flooding and the risk of soil erosion will decide the choice of agricultural and forestry crops, while slope gradients will determine utilization through the efficient and safe operation of machinery. Similarly, these site features can influence landscape development, wildlife, and amenity use. The risk of windthrow is peculiar to forestry and woodland and is considered to be largely a site factor incorporating wind zones, elevation, relief, and soils.

Some of the physical criteria for differentiating land use and quality in the UK on account of site are given in table 3.

Table 2. Climatic requirements for potential land uses and qualities.

	Agriculture			Woodland	
Category	AT0[1]	AAR[2]	ZSMD[3]	AT0	AAR
1.....	>1,250	<1,350	>225	>1,240	<1,350
2.....	>1,150	<1,600	Not limiting	>1,100	<1,600
3(a).....	>1,100	<1,750	Not limiting	>900	<1,950
3(b).....	>1,000	<1,950	Not limiting	>900	<1,950
4.....	>800	<3,200	Not limiting	>700	<3,200
5.....	Not limiting	Not limiting	Not limiting	Not limiting	Not limiting

[1]AT0 = Accumulated temperature between January and June (°C days).
[2]AAR = Average annual rainfall (mm).
[3]ZSMD = Zero soil moisture deficit (days).

Table 3. Site requirements for potential land uses and qualities.

	Agriculture-	Woodland		
Category	Gradient	Gradient	Terrain	Windthrow class
1.....	<1:8	<1:5	Even + firm	<2
2.....	<1:8	<1:3	Uneven + slightly soft	<3
3(a).....	<1:8	Not limiting	Moderately rough + moderately soft	<4
3(b).....	<1:5	Not limiting	Moderately rough + moderately soft	<4
4.....	<1:3	Not limiting	Rough + soft	<5
5.....	Not limiting	Not limiting	Very rough + very soft	Not limiting

Soil. Soil factors affect the potential and management of agricultural and forestry soils, but may be secondary or tertiary to climate and site limitations. The principal limiting soil characteristics are texture (particle size distribution), structure (particle aggregation and stability when subjected to wetting and drying), thickness of soil layers, stoniness, fertility (nutrients), salinity, soil reaction, and toxicity. The same soil characteristics affect the potential and management of wildlife habitats, the development of landscape features, and utilization for amenity use.

The wetness of the soil profile influences agricultural use (i.e., range and utilization of crops) through ease of cultivation and trafficking and risk of damage through poaching by livestock. The droughtiness of the soil profile influences agricultural use through range of crops and level of productivity. Texture and structure are the most significant soil characteristics in determining wetness and droughtiness within the context of local climate and site features. Soil depth is also of importance, affecting degree of droughtiness. The chemical characteristics of soils do not usually influence agricultural use where fertility content and soil reaction (pH) can be maintained or corrected by standard practice of applying fertilizers, manures, and neutralizing agents (e.g., lime and limestone). Chemical characteristics only affect land use quality where there are detrimental long-term effects on soil physical conditions (e.g., the effects of salinity on soil structure) and/or direct effects on the range of crops grown, and their productivity and management (e.g., control of salinity, toxicity, soil reaction).

Conditions of soil wetness and/or droughtiness influence forestry use and woodland type and the quality of the timber product. The influence of wetness is largely through choice of species, productivity, timber quality,

susceptibility to windthrow, and ground conditions that affect the ease of trafficking for maintenance and harvesting. Soil droughtiness influences forestry use through the range of species, productivity, and quality of timber. Unlike agricultural use, it is not usual to amend inherent chemical characteristics of soil to any degree and/or on a routine basis. Soil reaction (pH) and fertility are therefore of significance for the choice of species, productivity, and quality of the timber crop. Both also influence forestry use and woodland types. Other important chemical characteristics include salinity and toxicity (e.g., heavy metals).

Some of the physical criteria for differentiating land use and quality on account of soil characteristics are given in tables 4 and 5. For simplicity the interactions between these and climate are not presented.

Examples of the Use of the Physical Criteria

Some of the ways the above criteria have been used in the design of restoration schemes, in the specification of treatments, and in the assessment of the standard of achievement are illustrated below.

Design of Schemes

The physical criteria can be used to identify the type and potential of land uses. The criteria can be used in two main ways - to define the minimum requirements for the proposed land uses or, conversely, to suggest the most appropriate land uses for the proposed site and soil conditions. The following example concerns the matching of climate and the characteristic of soil droughtiness to woodland types that may be suitably planted. It also illustrates the use of the criteria in deciding the allocation of soil resources.

Table 4. Soil texture and stoniness requirements for potential land uses and qualities[1]

	Agriculture				Woodland-	
	Soil Textures[2]		Stoniness (vol. %)[3]		Soil Textures	
Category	Mineral	Organic	>20 mm	>60 mm	Mineral	Organic
1.....	SL, SZL	LS, SL, SZL	5	5	SL, SZL, ZL, SCL, CL, ZCL	LS, SL, SZL, ZL, SCL, CL, ZCL
2.....	LS, SL, SZL, SCL, CL, ZCL	S, LS, SL, SZL, SCL, CL, ZCL	10	5	SL, SZL, ZL, SCL, CL, ZCL, SC, ZC	LS, SL, SZL, ZL, SCL, CL, ZCL, SC, ZC
3(a).....	LS, SL, SZL, SCL, CL, ZCL	S, LS, SL, SZL, SCL, CL, ZCL	15	10	LS, SL, SZL, ZL, SCL, CL, ZCL, SC, ZC	Not limiting
3(b).....	Not limiting		35	20	LS, SL, SZL, ZL, SCL, CL, ZCL, SC, ZC	Not limiting
4.....	Not limiting		50	35	Not limiting	
5.....	Not limiting		Not limiting		Not limiting	

[1]For climate regime with zero soil moisture deficits of 176-225 days.

[2]Key to soil textures:

S	= sand.	CL	= clay loam.
LS	= loamy sand.	ZCL	= silty clay loam
SL	= sandy loam.	SC	= sandy clay.
SZL	= sandy silt loam.	ZC	= silty clay.
SCL	= sandy clay loam.	C	= clay.

[3]In upper 250 mm of soil profile.

Table 5. Soil wetness and droughtiness requirements for potential land uses and qualities.

	Agriculture				Woodland		
	SWC[1]	DSPL[2]	SMB[3]		SWC	DSPL	SMB
Category			Winter wheat	Potatoes			
1.....	<II	>800	+30	+10	<II	>800	>+50
2.....	<III	>620	+5	-10	<III	>600	0 - 50
3(a).....	<IV	>350	-20	-30	<IV	>450	-50 - 0
3(b).....	<IV	>350	-50	-55	<IV	>450	-50 - 0
4.....	<V	Not limiting	<-50	<-55	<V	Not limiting	
5.....	Not limiting				Not limiting		

[1]SWC = soil wetness class.
[2]DSPL = depth to slowly permeable layer (mm).
[3]SMB = soil moisture balance (mm).

At Gedling Colliery in the Midlands it is proposed to recover coal by washing the older tips and some lagoon material prior to final restoration. The former colliery (140 ha in extent) is to be restored to a mixture of agricultural land and public open space, incorporating woodland (65 ha), pasture (26 ha), grassland for amenity and wildlife (46 ha), and a sports turf area (3 ha). Because of its geographic location within the UK, climate is not a limiting factor for either agriculture or forestry. The only potential site limitation is gradient (table 3); whether or not gradient will be limiting will depend on the contouring of the washed and reshaped tips and lagoons. For woodland the restriction of gradient is less than for agricultural and sports amenity use. The soil physical characteristics are likely to be the principal limiting factor, largely through the degree of wetness and droughtiness of the soil profile. The former is largely a function of soil texture, structural development, and profile thickness, while droughtiness is influenced by soil texture, structural development, horizon thickness, and stoniness. The degree of wetness and droughtiness is also partly determined by the local climatic factors (MAFF 1988).

As is the case in many of the older collieries, not all of the soil materials on the site were stripped and stored for restoration before tipping the colliery wastes. Hence, there is a potential shortfall of soil material, and a need to allocate the available material to the best effect. A field survey of the soil resources on the ground and in store was undertaken, and the physical and chemical characteristics were recorded. This inventory of soil types and volumes was used as a basis for the allocation of soil material according to the preferred and required land use types and quality, and vegetation types.

Using the physical criteria, it was calculated that a soil profile with a soil moisture balance of >0 mm and a soil wetness class of I to III would be required to establish a productive, species-diverse, oak woodland typical of the locality. To achieve the above moisture balance with the minimum of soil cover over the colliery waste, and using the soil resources on site, it was calculated that 300 mm of topsoil and 400 mm of subsoil cover over a "rootable" colliery waste layer of 500 mm would be required. For soil covers of 450 and 200 mm, and for no cover, the moisture balances were calculated to be at best -34, -43, and -55 mm respectively (MAFF 1988, Meteorological Office 1989). These levels of moisture balance are typical of soils that support the relatively species-poor and unproductive birch-oak woodlands characteristic of excessively drained soils in the Midlands.

Because of the limited quantity of soil resources available (less than 80,000 m^3) and higher priority being given to a soil cover on the sports turf area and the agricultural grassland, it was decided that the woodland should be allocated little of the soil materials. Hence, it is proposed to plant a birch-dominated woodland type, which requires little or no soil cover. Locally, where a more productive and diverse woodland type is to be planted for landscape reasons, a thicker soil cover at the recommended depth will be provided.

Specifications of Treatments

The physical criteria can be used to specify treatments at restoration, particularly those relating to soils (thickness of soil layer, removal of stone, depth of decompaction). The following example concerns the effect of soil replacement and decompaction practices on soil wetness and hence, potential land quality for agriculture on the side slopes of colliery tips.

Soil wetness is a very important determinant of agricultural productivity and land quality in the UK (Humphries and Whittington 1988). It is partly determined by climate and hence location, soil texture, and depth to the uppermost slowly permeable layer (SPL). The latter can be briefly defined as a layer at least 150 mm in thickness with the upper boundary within 800 mm of the surface (MAFF 1988). Structureless soil horizons with firm strength are likely to be slowly permeable. This is also true of soils with weakly developed structure with granular, blocky, prismatic, or platy shapes and arrangements of peds or fragments and >18% clay.

Where soils are replaced by earth scrapers, they are likely to be compact and conform to the above criteria for semi permeable layers. Decompaction of the soil layer will be necessary (Bacon and Humphries 1987), and the depth of decompaction (i.e., depth to the SPL) needs to be sufficient to achieve the necessary drainage and hence degree of soil wetness, but also deep enough to provide the necessary available water capacity and hence, degree of droughtiness. For example, at Bevercotes Colliery in the Midlands, the gradient (1:12) limits the agricultural potential to category 3b (table 3). Here, decompaction to a depth of about 350 mm below the soil surface will meet the soil wetness class required by category 3b (table 5). In this case, ripping deeper would have been judged to be unnecessary.

Assessment of Restoration Achievement

The physical criteria can also be used as independent standards to determine whether restoration treatments have been successfully and fully implemented. Their use is considered to be a more sound and useful approach than vegetation-based parameters such as growth, productivity, or diversity, etc., that simply reflect the type and level of management practiced rather than the inherent structure and functioning of the restored site (Humphries et al 1984, Rowell and Humphries 1985, Humphries and Whittington 1988).

In 1984 the Land Capability Classification (Bibby 1982) was used to assess the restoration of a number of tips and lagoon banks to grassland at 26 collieries in the English Midland coalfields (Rowell and Humphries 1985). The Land Capability Classifications (LCC) is the forerunner of the MAFF system and uses similar physical criteria and categories to those listed in tables 1-5; LCC category 3 being equivalent to 3a in table 1, 4 to 3b, 5 to 4, and 6 to 5.

The objective in most of the cases was to restore site and soil conditions suitable for permanent pasture and grazing, and a grass cover to ameliorate the visual impact of the tips in the others. The LCC categories to be achieved would be 5 and 6 respectively. The categories of land use qualities identified in the field assessment are given in table 6. The limiting factors were generally a combination of gradient and/or erosion risk and stoniness and/or droughtiness. Droughtiness was generally the most constant factor owing to root growth being limited to a depth of 500 mm or less by the compaction of the colliery waste, and the sandy texture of the soils. Also, the gradient at a number of sites, and the moderate erosion risk at most, restricted the safe use of agricultural equipment.

Hence, the assessment demonstrated that the restored land at all of the collieries was shown to be LCC category 4 or 5, which was adequate for the land use quality required for the permanent pasture and more than adequate for supporting a grass cover for visual purposes. However, the restored sites are only capable of supporting stock-based farming enterprises. The land was not physically suited to arable or mixed enterprises, except for an occasional cereal (barley) break crop between resowing at some sites.

Table 6. Assessment of restoration achievement using LCC category according to climate, site, and soil limitations.

Colliery	Climate limitation	Site limitations Gradient	Site limitations Erosion	Soil limitations Stoniness	Soil limitations Droughtiness	Overall LCC category
Bilsthorpe.....	1	-	4	2	4	4
Ollerton.....	1	2	2	4	5	5
Silverhill.....	1	4	2	3	4	4
Thoresby.....	1	2	2	2	4	4
Clipstone.....	1	4	2	2	5	5
Cresswell.....	1	3	1	2	4	4
Sherwood.....	1	4	4	3	4	4
Harworth.....	1	3	2	2	5	5
Sutton.....	1	4	2	2	4	4
Bevercotes.....	1	4	2	3	4	4
Cridling Stubbs.....	1	1	1	2	5	5
New Fryston.....	1	3	4	1	5	5
Newmarket.....	1	3	4	4	5	5
Peckfield.....	1	2	4	1	5	5
Rothwell.....	1	5	4	4	5	5
Savile.....	1	5	4	3	5	5
Bentinck.....	1	3	4	2	4	4
Moorgreen.....	1	3	4	3	4	4
Pye Hill.....	1	3	4	4	5	5
Babington.....	1	5	4	2	4	5
Hucknall.....	1	3	4	3	4	4
Arkwright.....	1	4	4	1	4	4
Bolsover.....	1	4	4	3	4	4
Warsop.....	1	2	2	4	4	4
Barrow.....	1	4	4	1	4	4
Dodworth.....	1	3	4	4	4	4

Conclusions

We have been using the above physical criteria as standards for some 6 yrs in the planning and design of restoration schemes and in specifying treatments, and for almost 10 yrs in the assessment of the achievements on restored sites. We believe the approach has been both robust and of practical benefit, and has resulted in more satisfactory and cost-effective restoration. The same criteria are now used routinely as the basis for assessing the environmental impact of developments on agriculture and forestry. As a planning and design tool it has also been accepted by industry, local government, and their statutory advisers.

The occurrence of semi natural vegetation types such as heath, mire, and bog is also influenced by similar physical criteria. At present there are no published criteria for semi natural vegetation and habitat types for wildlife and nature conservation interest in the UK. There is no reason why such criteria cannot be developed for wildlife land uses and semi natural vegetation types. Currently we are involved in developing a comparable approach for restoring and assessing semi natural vegetation and habitat types for wildlife and nature conservation (Humphries 1994).

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QUANTIFYING TOPOGRAPHIC DIFFERENCES BETWEEN PREMINING AND RECLAIMED LANDSCAPES AT A LARGE SURFACE COAL MINE[1]

Joseph D. Friedlander[2]

Abstract: Premining and reclaimed landscapes on 2,300 acres were compared quantitatively at a surface coal mine in western North Dakota. Weighted average slopes were determined by calculating acres in each slope class (0% to 3%, 3% to 6%, 6% to 9%, 9% to 15%, and >15%) and dividing by the total number of acres. Weighted average slopes were 3.8% and 3.0% for all premining and reclaimed lands respectively. Weighted average slopes were 2.7% and 2.1% for premining and reclaimed croplands respectively. Reclaimed native rangeland slopes are 5.5% compared to 7.0% premining. The acreage of native rangeland with >9% slopes decreased more than 50% during reclamation. For all mined lands there is 25% more reclaimed land with 0% to 3% slopes and 25% less reclaimed land with >6% slopes than before mining. Topographic changes caused by surface mining and reclamation enhance soil conservation, moisture retention, and agricultural productivity. A cross-sectional profile had 26.8 and 15.5 premining and reclaimed land up-down gradient changes per mile respectively, indicating reclaimed land is less diverse than it was prior to mining. Slope class map units average 3.5 and 4.4 acres respectively on premining and reclaimed lands, further indicating reduced diversity. Landscape improvements conflict with regulatory diversity requirements to determine reclamation success.

Additional Key Words: reclamation, topography, surface mining

Introduction

Surface coal mining affects thousands of acres annually across the United States. Reclamation success is directly related to characteristics of the reclaimed landscape. This paper quantifies differences between premining and reclaimed landscapes at a large surface coal mine in the Northern Great Plains.

Topography is an important variable in the stability and productivity of agricultural lands. Steeply sloping and convex hillslopes are more prone to runoff and erosion than gently sloping and concave hillslopes. Researchers have made recommendations for reconstructing landscapes following surface mining. The Surface Mining Control and Reclamation Act requires that mined land be returned to "approximate original contour" or AOC. Brenner and Steiner (1987) and Gregory et al. (1987) state that grading spoils to AOC requirements may actually be detrimental to a reclaimed landscape; they suggest landscape improvements be made during spoils grading. Vogel (1987) recommends slopes of about 2%, stating that constructing broad flat areas should be avoided, as differential settlement could result in poor drainage. Bauer et al. (1976) recommend mine spoils be graded to slopes less than 5%. Stiller et al. (1980) recommend reducing slope lengths on reclaimed lands. Richardson and Wollenhaupt (1983), Law (1984) and Schaefer et al. (1979) stress the importance of slope shape, and favor creation of concave slopes in reclaimed landscapes.

Research has been conducted to quantify landscape characteristics and their effect on erosion and agricultural production. Lentz et al. (1993) modeled a relationship between certain landscape parameters and ephemeral gully erosion on unmined cropland. Halvorson and Doll (1991) developed a topographic slope factor to quantify the topographic effects of landscape on surface water runoff and runon. This factor correlated

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[1]Paper presented at the International Land Reclamation and Mine Drainage Conference and the Third International Conference on the Abatement of Acidic Drainage, Pittsburgh, PA, April 24-29, 1994.

[2]Joseph D. Friedlander, Environmental Manager, The Coteau Properties Company, Beulah, ND.

closely with wheat yields in western North Dakota. Schroeder and Carter (1992) found this slope factor equally useful on reclaimed lands.

Little work has been conducted to quantifiably compare premining and reclaimed landscapes. Gregory et al. (1987) addressed the current subjectivity associated with evaluating reclaimed landscapes by suggesting several quantitative measures, including frequency distribution of slopes.

Probably the greatest stumbling block associated with quantifying a comparison between premining and reclaimed landscapes is that landforms do not lend themselves to normal randomized sampling techniques and statistical comparisons of sampled landscape parameters. An evaluation of entire land tracts provides the best comparison between premining and reclaimed landscapes.

Location and Mining Operations

The Freedom Mine, located about 90 miles northwest of Bismarck, N.D., is owned and operated by The Coteau Properties Company, a subsidiary of the North American Coal Corporation. Lignite coal mining began in 1983, and current production is about 15.6 million st/yr. From 400 - 700 acres are mined and reclaimed annually. About 3,000 acres have been reclaimed to date.

The undisturbed landscape is characterized by mostly gently rolling and well-drained croplands used for dryland wheat production. About 25% of the mine is composed of steeply sloping mixed grass native rangeland used for livestock grazing. Most areas used for grazing have slopes too steep and soils too shallow to support grain production.

Earth moving for mining is accomplished through the use of tractor-scrapers, bulldozers, a truck-shovel fleet, and two 110-cu-yd class draglines. Spoils are regraded immediately behind active pits, and topsoil and subsoil are respread and seeded immediately after regrading. Reclaimed lands are currently being used for hay and grain production and livestock grazing.

Comparison of Premining and Reclaimed Landscapes

About 2,300 acres of reclaimed land were evaluated and compared with the same lands prior to mining. Slopes were compared by class: 0% to 3%, 3% to 6%, 6% to 9%, 9% to 15%, and >15%. The acreage of each slope class was measured for premining and reclaimed croplands, rangelands, and all lands (croplands, rangeland, and small tracts of industrial and shelterbelt land uses).

Because values represent the entire field or population of premining and reclaimed lands, no statistical analyses are applied. Differences between premining and reclaimed lands are absolute for all land considered here.

A weighted average slope for each type of land was determined by multiplying the midpoint of each slope class by the number of acres in that class, and dividing by the total number of acres evaluated. A 15% value was used for slopes greater than 15%.

Weighted average slopes decreased for all three categories of land (table 1). All categories of land have lower average slopes on reclaimed land compared to premining land. The greatest decrease was on reclaimed rangeland. These slope reductions represent landscape improvements positively affecting water runoff, erosional stability and ultimately agricultural productivity.

Table 1. Comparison of premining and reclaimed lands at the Freedom Mine.

	Slope class					Average weighted
	0% to 3%	3% to 6%	6% to 9%	9% to 15%	>15%	slope by percent
Total lands reclaimed (2,294.6 ac.):						
Premining acreage	1,244.1 (54%)	709.4 (31%)	181.7 (8%)	107.3 (5%)	52.1 (2%)	3.77
Reclaimed acreage	1,557.3 (68%)	479.5 (21%)	175.2 (8%)	71.9 (3%)	10.7 (0%)	2.99
Croplands reclaimed (1,447.6 ac.):						
Premining acreage	960.5 (66%)	434.0 (30%)	38.1 (3%)	15.0 (1%)	0.0 (0%)	2.66
Reclaimed acreage	1,185.2 (82%)	247.6 (17%)	13.9 (1%)	0.9 (0%)	0.0 (0%)	2.08
Native rangeland reclaimed (405.6 ac.):						
Premining acreage	98.0 (24%)	116.9 (29%)	91.0 (22%)	54.5 (13%)	45.2 (11%)	6.97
Reclaimed acreage	119.2 (29%)	119.3 (29%)	119.7 (29%)	40.3 (10%)	7.1 (2%)	5.48

Slope class acreage changes further demonstrate landscape improvements. For all mined lands there is an increase in the acreage with 0% to 3% slopes, from 1244 acres prior to mining to 1557 acres after mining and reclamation (fig. 1). This is a 25% increase in the amount of land with most favorable slopes. This change correlates with decreased amounts of land having steeper slopes. About half of all lands prior to mining had slopes greater than 3%, but after mining and reclamation only a third of all lands have slopes greater than 3%.

The same trend is evident for mined and reclaimed croplands, where shallow slopes are especially beneficial (fig. 2). The amount of cropland with 0% to 3% slopes increased 23%, from 960 acres premining to 1,185 acres following mining and reclamation. Again, there are subsequent decreases in acreage with steeper slopes. Following reclamation, the amount of cropland with slopes greater than 6% has decreased to less than a third of its premining acreage.

Both premining and reclaimed native rangeland have steeper slopes than croplands, although reclaimed rangelands also exhibit favorable landscape changes (fig. 3). Rangeland with 0% to 3% slopes increased 20%, from 98 acres premining to 119 acres following mining and reclamation. Changes in rangeland acreage with 6% to 9% slopes are very similar. The amount of rangeland with slopes greater than 9% decreased from 100 acres premining to 47 acres following mining and reclamation. This is more than a 50% decrease in the amount of rangeland with unfavorably steep slopes. Not reflected in described values are badland type rangelands, many of which were impassable with four-wheel drive trucks or farm tractors prior to mining, either because of their steep slopes or very rocky conditions. These have been eliminated entirely, further improving the quality of this land for livestock production. Many rangeland areas that were previously too steep or rocky for farm equipment can now be cut for hay. Although they may not qualify for use as cropland, their economic potential has increased tremendously over premine conditions.

Decreasing slopes increases soil moisture retention, decreases erosion, and enhances crop and

forage productivity. It also provides more favorable conditions for livestock grazing. These positive landscape changes are accompanied by a decrease in landscape diversity caused by mining and reclamation.

Topographic diversity has a profound effect on vegetative diversity, for which standards must be met to achieve bond release on reclaimed rangelands. High vegetative diversity standards may be difficult to attain if rangelands are reduced in slope and improved for productivity. Thus productivity and diversity standards for successful reclamation may be in conflict.

Several vegetative diversity indices are available. However, there are no such diversity indices currently in use for evaluating and comparing landscapes. Premining and reclaimed landscapes can be compared graphically and quantitatively through the use of topographic maps and cross-sectional profiles (figs. 4-5-6). A cross-sectional profile approximately 7,500 feet long was developed along the same northwest-southeast line through premining and reclaimed rangeland. Reduced slopes, combined with fewer changes in slope and up-down gradient, results in a reclaimed landscape less diverse than before mining. The number of hilltops and valley bottoms per length of profile line relate directly to the total number of up-down gradient changes and can be used as a quantitative measure of landscape diversity. For the 7,500 feet long profile described here, there were 38 hilltops and valley bottoms (gradient changes) prior to mining, and 22 following mining and reclamation. This equates to 26.8 gradient changes per linear mile prior to mining, compared with 15.5 per linear mile for the same land after reclamation. This 40% reduction in the number of gradient changes quantitatively describes a reduction in landscape diversity.

Another measure of landscape diversity is the total number of discrete slope class map units in a given area. Two areas of equal size may have the same weighted average slope. However, one might have a more diverse topography if the area were composed of several discrete, discontinuous map units of different slope classes (fig. 7). Slope class map unit analysis was conducted for 1,777 acres at the Freedom Mine. There were 502 premining slope

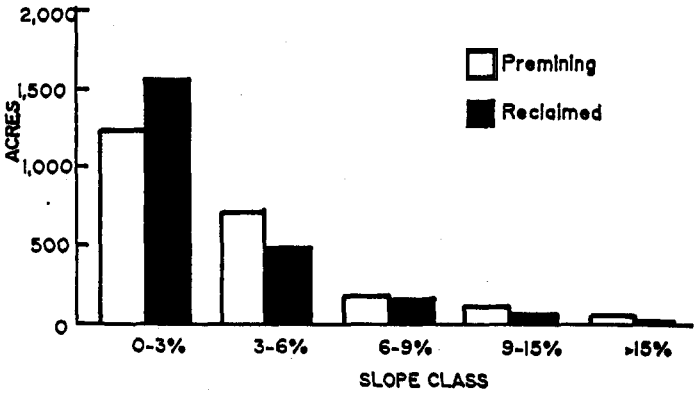

Figure 1. Comparison of premining and reclaimed land slope classes for all lands at the Freedom Mine.

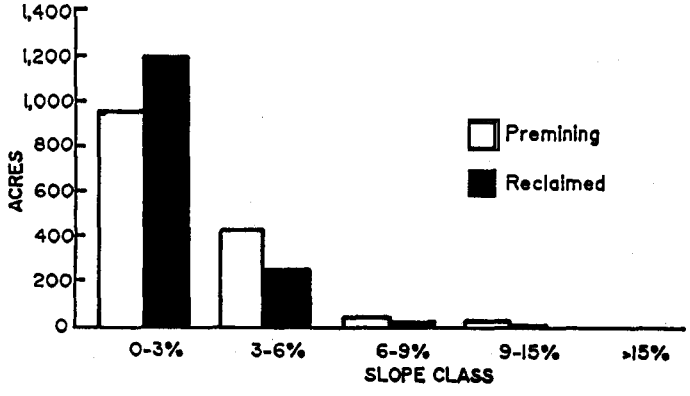

Figure 2. Comparison of premining and reclaimed land slope classes for cropland at the Freedom Mine.

Figure 3. Comparison of premining and reclaimed land slope classes for native rangeland at the Freedom Mine.

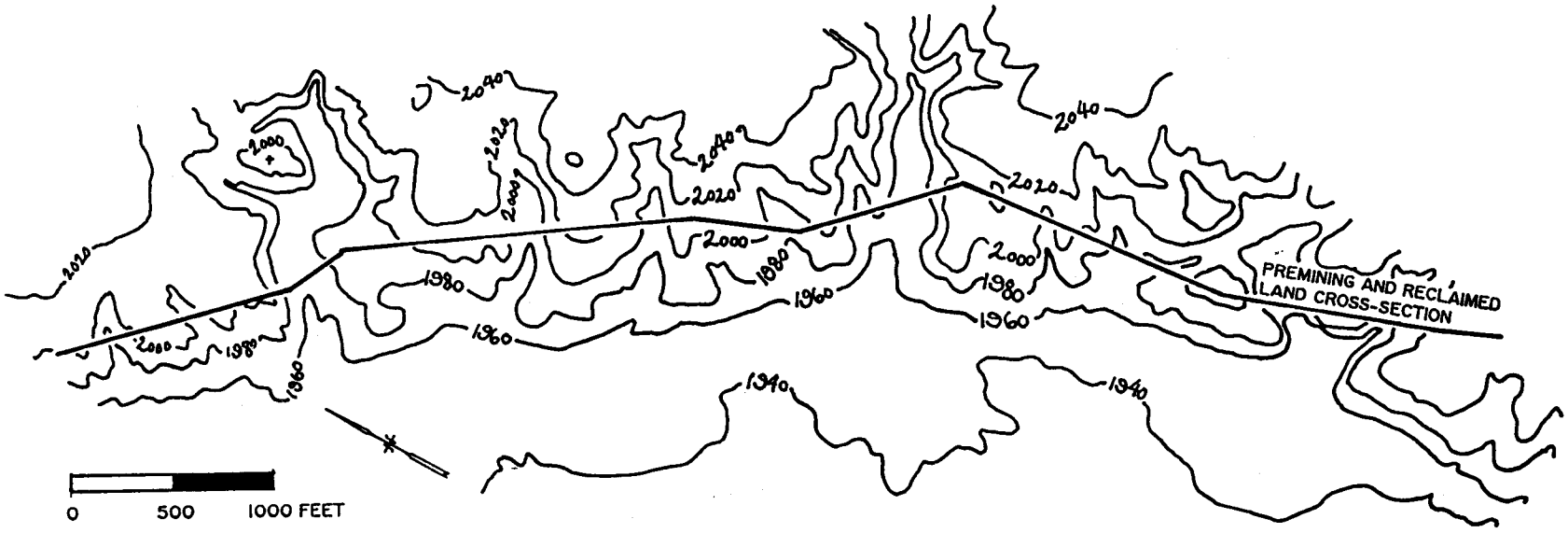

Figure 4. Topographic map of premining native rangeland.

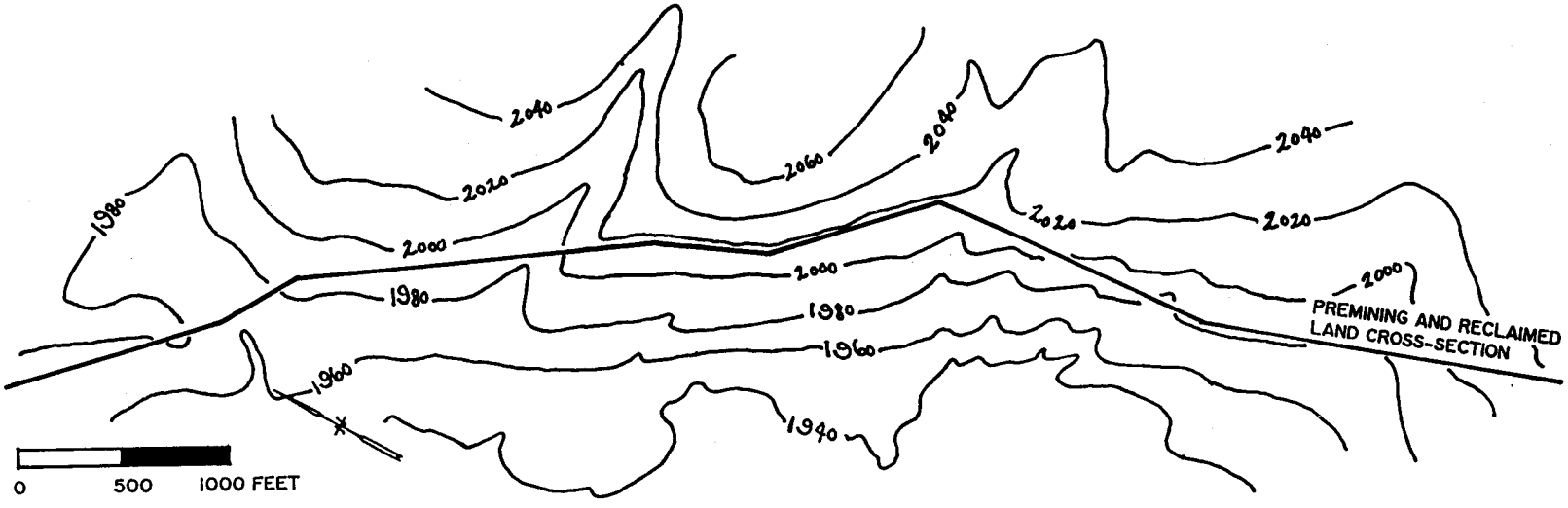

Figure 5. Topographic map of reclaimed native rangeland; this is the same area shown in Figure 4.

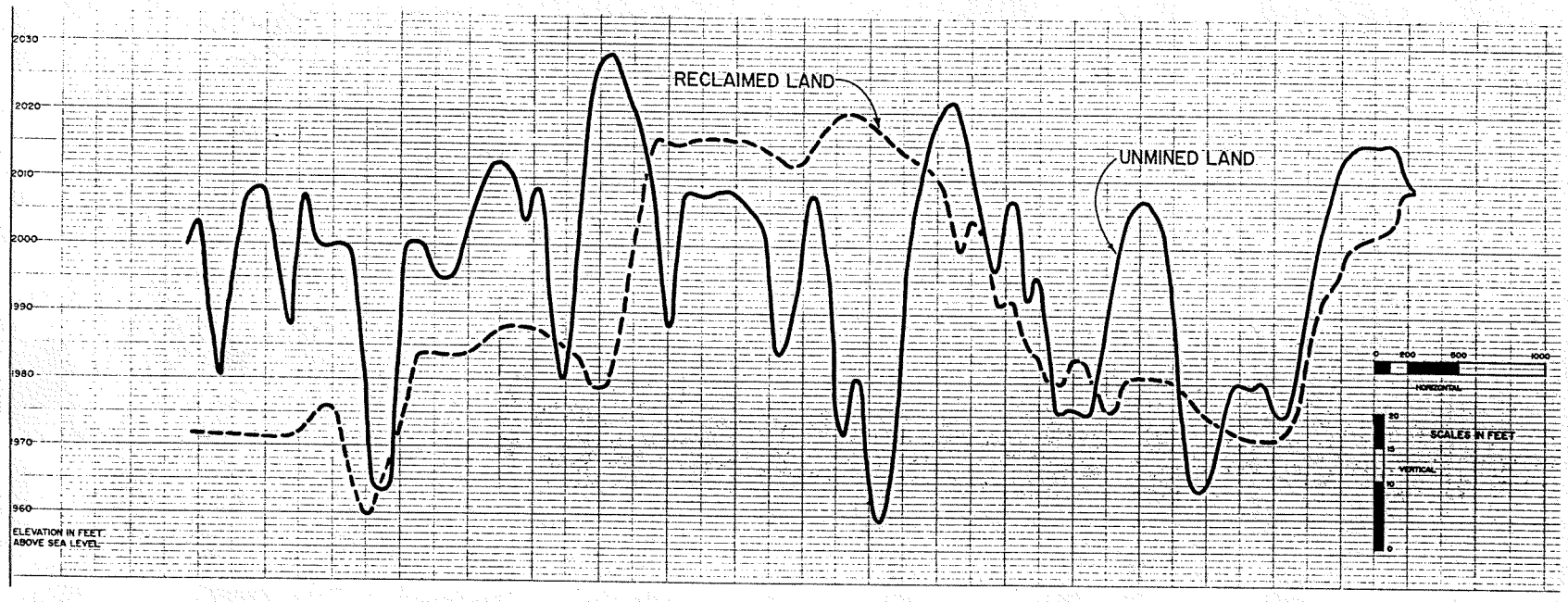

Figure 6. Cross-sectional profiles before and after mining and reclamation.

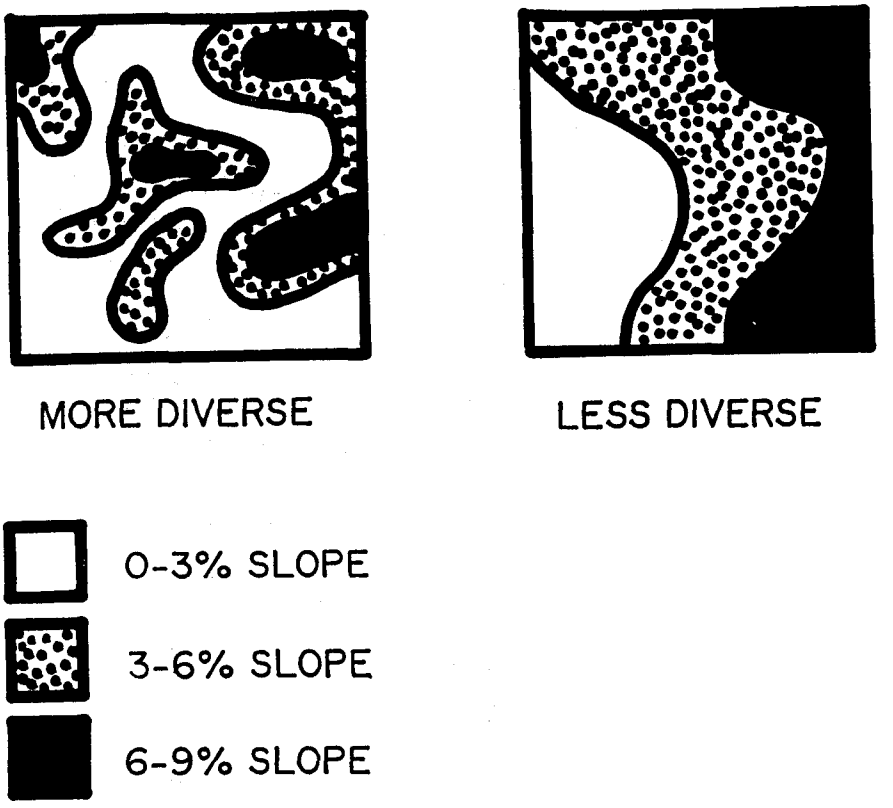

Figure 7. Comparison of two areas with the same weighted average slope, but different slope map unit distributions make one area more topographically diverse than the other.

class map units, with an average 3.54 acres per map unit. After mining and reclamation, there are now 408 slope class map units, with an average 4.36 acres per map unit. This is a 19% decrease in the number of discrete slope class map units and a corresponding 23% increase in average map unit size. This information can also be used to quantify changes, in this case a decrease, in landscape diversity.

Summary and Conclusions

Premining and reclaimed landscapes were compared at the Freedom Mine, a large surface coal mine in western North Dakota. This comparison demonstrates that average slopes decreased for both cropland and native rangeland. These slope changes are considered favorable from a production agriculture standpoint. Landscape diversity decreased when comparing quantitative measures of up-down gradient changes and numbers of slope class map units. Reduced landscape diversity appears to correlate with landscapes improved for agriculture, suggesting that reclamation standards for vegetative production and diversity are in conflict. Possibly a vegetative diversity standard is necessary that ties to landscape diversity, instead of premining conditions which no longer exist in the reclaimed landscape.

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MODELING AGRICULTURAL IMPACTS OF LONGWALL MINE SUBSIDENCE: A GIS APPROACH[1]

Robert G. Darmody and Scott L. Vance[2]

Abstract: Illinois is both a major agricultural State and one of the leading coal-producing States. The future of coal mining in Illinois is longwall mining. One of the advantages of longwall mining, and the most noticeable consequence, is immediate subsidence. Mitigation of subsidence effects is the responsibility of the coal company. Research has shown that mitigation is usually effective, but may be difficult in many cases. Minimizing subsidence impact by avoiding sensitive soils in the mine plan is a possibility that should be considered. Predicting agricultural impacts of subsidence would give mine designers and regulating agencies an additional tool to use when evaluating mine plans. This paper reports on the development and an application of a predictive model of agricultural soil subsidence sensitivity (SSS). The SSS model involves integration of selected soil properties in a GIS (geographical information system) to assign a subsidence sensitivity class to a given area. Predicted crop yield losses at a proposed longwall mine in southern Illinois, using corn (*Zea mays* L.) as a reference, were 6.8% for the longwall panel area but ranged from 4.1% to 9.5% for the individual panels. The model also predicted that mitigation of the affected areas would reduce yield losses to 1.2% for the longwall area and to 0.5% to 1.7% for the individual panels.

Additional Key Words: underground coal mining, environmental impact, crop production, corn yields, mitigation.

Introduction

Longwall mining in Illinois has a deleterious effect on crop yields. Subsidence subsequent to mining forms depressions that may cause ponding in agricultural fields. Ponded areas pose several problems to agriculture, which may include difficult cultivation, poor stand establishment, loss of nutrients, poor root development, and increased diseases. These problems can be quantified by measuring yield reduction at harvest. In an earlier study, crop yield reduction, as estimated using actual corn yields at subsidence sites selected from aerial photographs, was 4.7% (Darmody et al. 1988a, 1989a). This estimate was for the entire mine area, individual sites identified as having severe subsidence effects had an average loss of 95%. Sites identified as having moderate effects had 43% reduction and sites with slight effects had 2% yield reduction. Coal companies are required to mitigate subsidence effects to restore land use. There are no crop yield requirements as with surface mining. Research indicates that mitigation is successful in restoring land use and soybean (*Glycine max* L.) yields, but not corn (*Zea mays* L.) yields (Hetzler and Darmody 1992). Corn yields in mitigated sites that had severe impact of subsidence average 19% lower than on adjacent undisturbed sites (Darmody et al. 1992).

Soil properties that contribute to deleterious effects of longwall mining are primarily related to hydrologic aspects of the soil (Darmody et al. 1988b, 1989b). These include water table depth, flooding probability, slope, natural drainage, and hydraulic conductivity. Unfortunately, many soils in Illinois, particularly those on the Illinoian till plain, have many of these properties that make them sensitive to subsidence (Fehrenbacher et al. 1984). Longwall mining to date has been largely in the portion of the State that includes soils that are relatively insensitive to the deleterious effects of subsidence. The work reported here involves the development of a predictive model for determining the sensitivity of agricultural soils to subsidence. This

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[2] Robert G. Darmody, Associate Professor of Pedology and Scott L. Vance, Research Specialist, Department of Agronomy, University of Illinois, Urbana, IL, USA.

model should be helpful both to the industry and to regulatory agencies. It is particularly important as longwall mining moves into the extensive areas of highly subsidence sensitive soils on the Illinoian till.

The objectives of this research were to (1) develop a predictive model of agricultural soil sensitivity to subsidence (SSS) associated with longwall mining and (2) apply the model to a proposed longwall mine plan.

Methods and Materials

Subsidence Yield Reduction Computation

Soil properties that influence sensitivity to subsidence were given a weight to assign an SSS score to each soil map unit in the county (table 1). The soil properties used (table 2) were depth to seasonally high water table, slope, probability of flooding, natural soil drainage, and hydrologic group (Drablos and Moe 1984, Darmody et al. 1989b, Hodges 1990, Darmody et al. 1988b). SSS scores ranged from 0 to 20. SSS classes were assigned to soil mapping units based on the sum of their scores. An SSS score of <6 was assigned to SSS class

Table 1. Properties used to assign subsidence sensitive scores to soils.[1]

Slope class	Drainage group	Flooding		Hydrology group	Water table depth (cm)		Subsidence score
		Frequency	Duration		Perched	Apparent	
A	3A, 4A, 4B	Frequent	Brief and longer	D	<0	<30	4
	2A, 3B, 3C, 4C	Frequent	Very brief	C		30-60	3
B	2C, 2B	Occasional	Brief	B	0-30	60-180	2
	1C	Occasional	Very brief	A	31-180		1
C		None	---		>180	>180	0
D							-3
>D							-4

[1] See table 2 for property definitions.

Table 2. Definitions of soil property criteria classes.

Slope class	Drainage group		Hydrology (runoff)	Flooding frequency
	Drainage	Permeability (in/h)		
A= 0-2%	A = Poorly	1=Rapid (>2)	A= Low	Occasional= <50%
B= 2-5%	B = Somewhat poorly	2= Moderate (0.6-2)	B= Low moderate	Frequent= ≥50%
C= 5-10%	C = Well + moderately well	3=Mod. slow (0.2-0.6)	C= High moderate	Very brief= <2 days
D= 10-15%		4=Slowly (<0.2)	D= High	Brief= ≥2 days

A, none to slight subsidence hazard; 6 to 10 was class B, slight hazard; 11 to 15 was class C, moderate hazard; >15 was class D, severe subsidence hazard.

The crop yield impact associated with each SSS class was determined from previous research on the impact of longwall subsidence on corn yields (Darmody et al. 1988b). The proportion of the yield reduction found in that earlier study associated with each SSS class was determined (table 3). The effectiveness of subsidence mitigation for restoring corn yields was estimated from the results of a previous research project (Darmody et al. 1992). Reference corn yields for each soil map unit were taken from the Soil Survey of Macoupin County (Hodges 1990).

Table 3. Generalized impact of subsidence on crop yields.[1]

Subsidence	Yield reduction, %		Extent (%) of soils in each class				Average yield loss, %	
risk class	Subsided	Mitigated	None-slight	Slight	Moderate	Severe	Subsided	Mitigated
None-slight	0	0	90	10	0	0	0.2	0.0
Slight	2	0.4	82	15	3	0	1.6	0.3
Moderate	43	8.6	56	35	8	1	5.1	1.0
Severe	95	19	47	40	6	7	10.0	2.0
Average	35	7	69	25	4	2	4.2	0.8

[1] Data derived from Darmody et al. 1988b and Darmody et al. 1992.

GIS Software

The geographical information system (GIS) software package GRASS (USACERL 1991) was used to generate the maps and tabular data of soil properties, SSS information, and crop yields. The Soil Conservation Service (Tom D'Avello, personal communication 1993) provided the digital soil map that was used as a base map and from which the other maps were derived. The mine plan map was hand-digitized with a digitizing board.

Research Area

The study area was a proposed longwall permit area in Macoupin County, IL (fig. 1). It was in the Honey Creek watershed which includes many of the county soils (Hodges 1990). However, because of the proximity of the creek, it was more sloping than much of the county (fig. 2). The physiography of the southern portion of the study area is dominated by the creek valley and its associated sloping soils; the northern portion of the study area is on the nearly level, poorly drained Illinoian till plain (Fehrenbacher et al. 1984).

Figure 1. Location (*) of longwall mine subsidence modeling study in Illinois.

The proposed longwall mine plan was provided by the Illinois State Geological Survey (Robert Bauer, personal communication 1993). The plan consists of a longwall permit area of 2,044 ha (5,048 acres) with 17 longwall panels. The panels are approximately

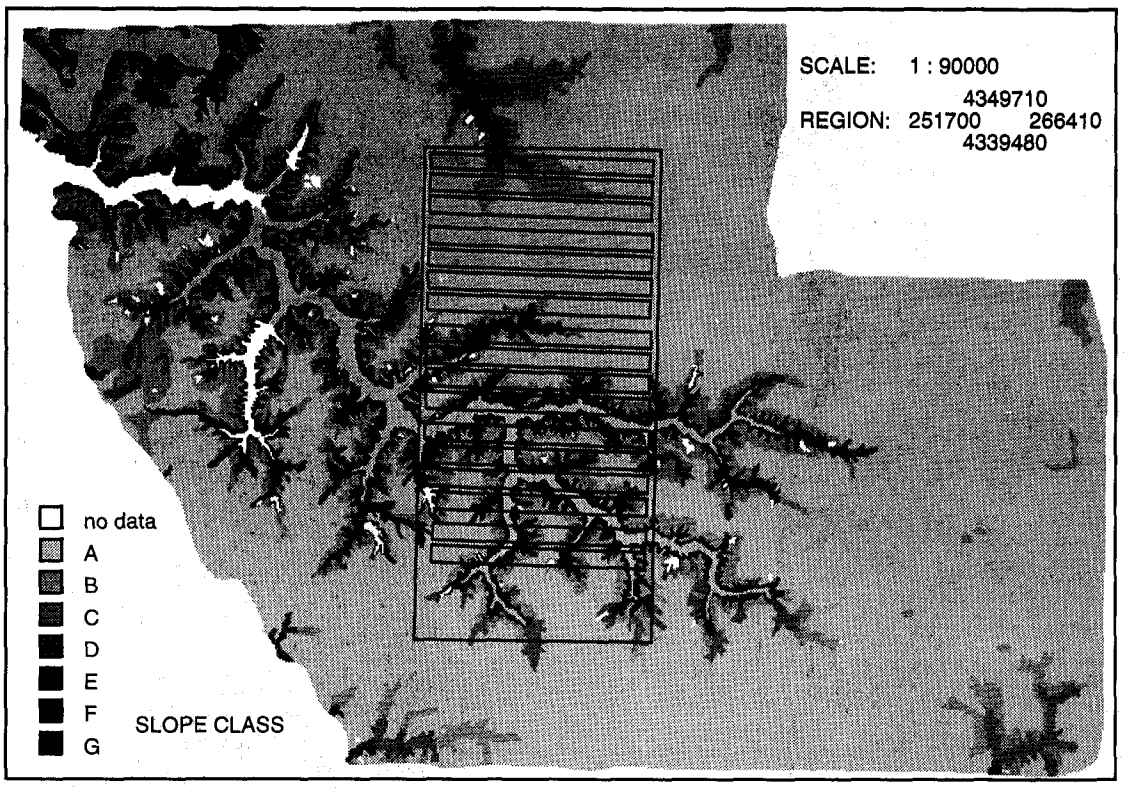

Figure 2. Soil slope classes of Honey Creek watershed, longwall permit area is outlined in black.

3,000 m (10,000 ft) long and 200 m (600 ft) wide and are oriented east-west. They are numbered for research purposes from 1 to 17, south to north.

Results and Discussion

SSS scores for the soils of the study area ranged from 1 to 18, with a weighted average score of 12.3. The scores tended to be higher in the northern portion of the study area, away from the creek valley and towards the Illinoian till plain (fig. 3). Severe was the most extensive SSS class generated from the SSS scores, it covered 37% of the study area (table 4). The northern portions of the study area on the Illinoian till plain are primarily in the moderate and severe classes, the southern portion of the study area includes some none-slight and slight classed land (fig. 3). If the mine were located more to the north or east on the nearly level Illinoian till plain, the SSS scores would be higher and unmitigated subsidence would pose a greater hazard to agriculture.

Table 4. Soil subsidence sensitivity (SSS) classes for the longwall permit area.

SSS	Area		
class	Acres	Hectares	%
None.....	842	341	17
Slight.....	1,043	422	21
Moderate..	1,278	518	25
Severe.....	1,886	764	37
Total.....	5,048	2,045	100

Most of the soils in the study area are well suited for corn growth. The weighted average corn yield for the study area is 112 bu/ac (table 5). The most productive soils have a predicted yield of 141 bu/ac and are in the northern part of the study area (fig. 4). Some soils, particularly on the steeply sloping valley walls in the southern portion of the study area,

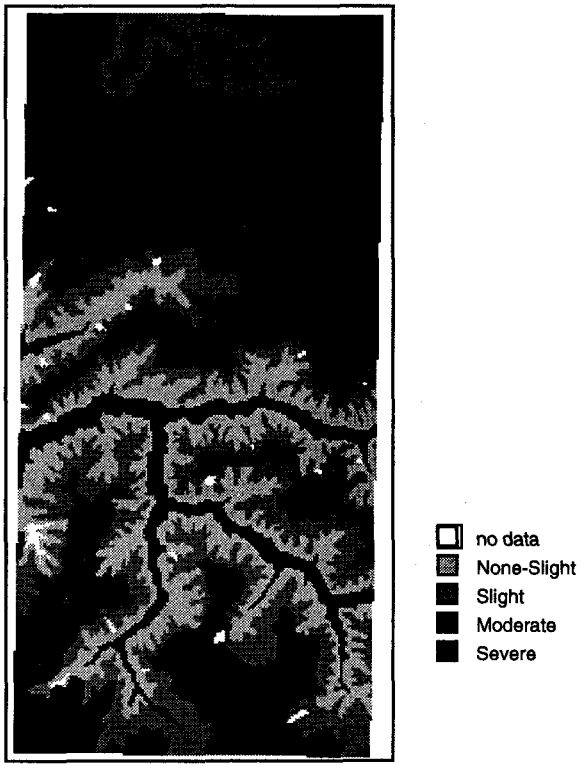

Figure 3. Soil subsidence sensitivity (SSS) classes for the longwall permit study area.

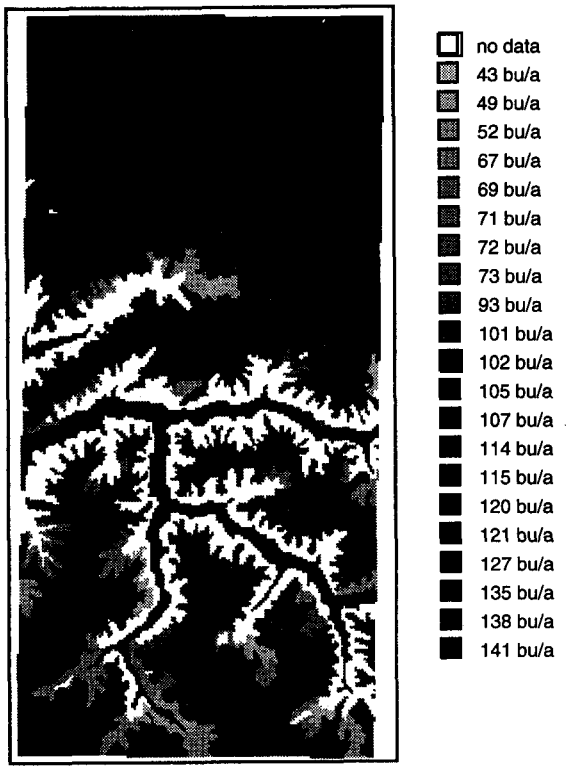

Figure 4. Predicted premining corn yields for the study area.

Table 5. Predicted corn yields for the total longwall permit area.

Condition	Total corn yield		Weighted average corn yield		
	Bushels	Kilograms	bu/ac	kg/ha	%
Unmined.....	488,031	14,400,298	112	7,049	100.0
Subsided.....	454,931	13,426,612	105	6,572	93.2
Mitigated.....	482,216	14,233,746	111	6,967	98.8

are not suitable for corn growth and are shown in white in figure 4. The predicted impact of subsidence in the longwall permit area, based upon the weighted average of all of the soils of the study area, is a 5.4% reduction in crop yields (table 6). If the portion of the study area unsuitable for corn production is excluded from the analyses, the predicted corn yield after subsidence is 105 bu/ac (table 5). This is a reduction of 6.8%. This estimate includes the entire area under consideration. Previous reports of corn yield reductions of 95% referred to those limited portions of the mine area that were most highly affected (Darmody et al. 1989a). Figure 5 displays the generalized distribution of predicted corn yields,

Table 6. Predicted subsidence impact on crop yield for the longwall permit area by SSS class.

SSS	Yield	Area		
class	loss, %	Acres	Hectares	%
None.....	0.2	842	341	16.7
Slight.....	1.6	1,043	422	20.7
Moderate	5.1	1,278	517	25.3
Severe.....	10.0	1,886	763	37.3

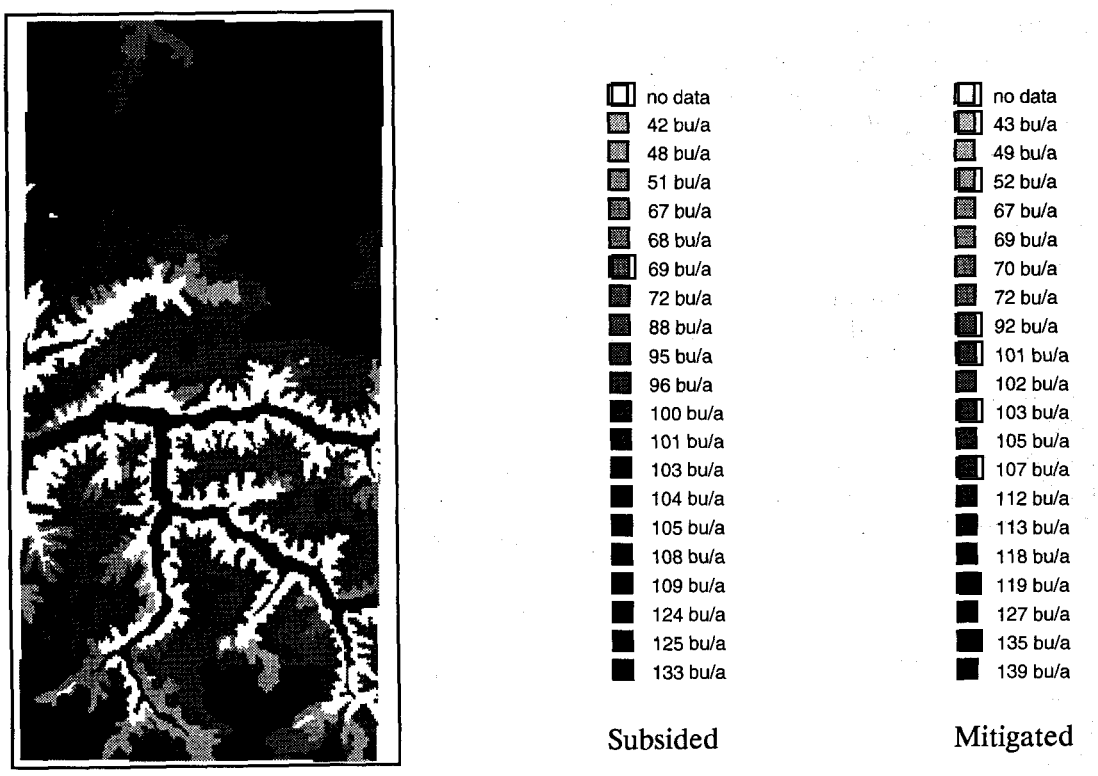

Figure 5. Predicted corn yields for the study area, after longwall mining.

which range from 42 to 133 bu/ac, for the study area after subsidence.

Subsidence impact would not be expected to be uniformly distributed over a given panel as shown in the model. The resolution of the available topographic data would not permit displaying individual portions of a panel. This apparent deficiency in the model is, however, accommodated by adjusting the impact over the entire panel according to known impact distributions from previous studies (table 3).

Mitigation of the subsidence effects is predicted to lessen the impact on crop yields. The weighted average loss in corn yields after subsidence mitigation, when all the soils in the study area are included, is predicted to be 1.2% (table 5). Figure 5 shows the generalized distribution of predicted corn yields, which range from 43 to 139 bu/ac, for the study area after subsidence mitigation.

Because the northern and southern portions of the proposed mine are so different, the subsidence

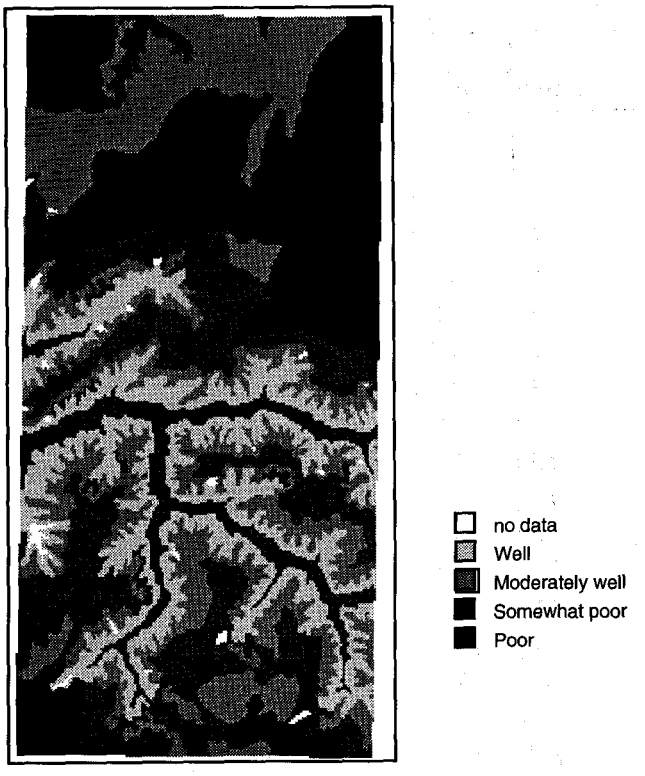

Figure 6. Soil drainage classes of the soils of the longwall mine permit study area.

impact on individual panels varies considerably. The southern panels are in the more sloping soils near Honey Creek and the northern panels are on the nearly level Illinoian till plain (fig. 2). Sloping ground tends to be less sensitive to subsidence because closed depressions that can accumulate runoff are less likely to form (Darmody et al. 1989b). Another way the two areas differ is in natural soil drainage (fig. 6). In the north, soils tend to be poorly drained and to have shallow water tables. In the southern portion of the permit area, drainage is generally better and water tables are deeper. Poorly drained soils are more sensitive to subsidence because they routinely suffer from excessive water and tend to have slow hydraulic conductivity. Drain tiles generally are not recommended for slowly permeable soils, especially if an outlet is not available (Drablos and Moe 1984). Subsidence would render existing drain tiles useless, and may even cause them to run backwards, because of the change in gradient subsidence causes. In addition, water tables tend to remain at the same elevation after subsidence, and shallow ones cause a greater sensitivity to subsidence (Darmody et al. 1989b).

The differences in topography and soils among the panels are reflected in different SSS scores (table 7). SSS scores in the southern panels range from 8.6 to 10.7. In the north, SSS scores range from 11.8 to 16.9. These SSS scores result in predicted crop yield losses of 4.1% to 6.5% for the southern panels and of 5.8% to 9.5% for the northern panels. Crop yield reductions after mitigation are predicted to be from 0.5% to 1.1% for the southern panels and from 1.1% to 1.7% for the northern panels.

Summary

The SSS model is a useful tool for predicting the impact that longwall mine subsidence will have on production agriculture. Different mine plans can be tested to predict the impact mining will have on crop yields. This can be done to estimate mitigation costs, to test different mine plans to minimize impact, or to predict loss in crop yields.

Table 7. Predicted corn yields before and after longwall mining in the study area.

	SSS		Initial yield		Subsided yields			Mitigated yields		
Panel	Score	Class	Total bu	bu/ac	Total bu	bu/ac	% loss	Total bu	bu/ac	% loss
1	9.9	3.4	12,241	103	11,653	98	4.8	12,150	102	0.7
2	10.7	4.1	12,854	102	12,136	96	5.6	12,740	101	0.9
3	8.7	2.9	10,531	101	10,042	96	4.6	10,460	100	0.7
4	10.4	3.6	13,712	105	13,019	99	5.1	13,611	104	0.7
5	8.9	2.5	11,047	100	10,590	96	4.1	10,988	99	0.5
6	8.6	2.7	10,590	105	10,091	100	4.7	10,521	104	0.6
7	8.9	3.6	10,186	106	9,526	99	6.5	10,078	105	1.1
8	10.0	2.9	12,611	101	12,086	97	4.2	12,540	100	0.6
9	12.9	5.6	15,477	108	14,393	101	7.0	15,300	107	1.1
10	11.8	4.9	13,590	103	12,664	96	6.8	13,444	102	1.1
11	15.1	7.5	18,733	114	17,245	105	7.9	18,464	112	1.4
12	16.9	9.3	19,952	122	18,060	111	9.5	19,607	120	1.7
13	16.5	9.0	20,751	128	18,853	116	9.1	20,394	126	1.7
14	16.0	8.4	21,664	130	19,812	119	8.5	21,301	128	1.7
15	15.7	8.3	21,019	128	19,256	118	8.4	20,683	126	1.6
16	13.4	5.7	20,760	129	19,555	121	5.8	20,542	127	1.1
17	13.6	6.1	18,144	124	17,012	116	6.2	17,923	123	1.2
All	12.3	5.3	263,861	114	245,993	106	6.8	260,746	113	1.2
Permit	12.3	5.4	488,031	112	454,931	105	6.8	482,216	111	1.2

The predicted impact, as measured by corn yield loss, for the proposed mine in Macoupin County, IL is 6.8% for the permit area and ranges from 4.1% to 9.5% for the individual panels. These estimates include the entire area under consideration. Mitigation is predicted to lessen the impact to 1.2% loss of corn yield for the permit area and to 0.5% to 1.7% for the individual panels.

Acknowledgements

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EFFECTIVENESS OF TREATMENTS TO ESTABLISH TREES ON MINELANDS DURING DROUGHT AND WET YEARS[1]

Merlyn M. Larson, David A. Kost, and John P. Vimmerstedt[2]

Abstract: In two studies, green ash (Fraxinus pennsylvanica) and white pine (Pinus strobus) were planted on three minesoils (graded topsoil, ripped topsoil, and gray cast overburden). Mixtures of grasses and/or legumes were seeded at different times in relation to tree planting. In the first study, tree planting was followed by several weeks of drought; in the second study, precipitation was above average for the first two growing seasons following planting. In the drought year, survival of green ash was influenced by minesoil type, herbaceous mixture, and time of seeding in relation to tree planting. Among minesoils, ash survival on ripped topsoil was 28% above that on graded topsoil in one seeding treatment, but mean survival was highest (87%) on cast overburden. Seeding grasses the fall before planting resulted in poor ash survival (40% to 47%) compared with seeding at time of planting (82% to 85%). Ash survived well (81% to 94%) on legume-seeded plots owing, in part, to slow early development of legumes the first season. When tree planting was followed by two wet seasons, survival at 4 and 5 yr ranged from very good to excellent, with much smaller differences between treatments. Total height of ash trees on cast overburden averaged 31% less than that of trees on topsoil. Mean height of ash on legume-seeded subplots was 75 cm, compared with 58 cm on grass subplots, although herbaceous biomass was greater on legume subplots. The three minesoils proved unsuitable for white pine.

Additional Key Words: minesoils, ripping, cast overburden, green ash, white pine, herbaceous competition.

Introduction

Minelands that have been topsoiled and seeded with herbaceous species are often hostile sites for planted tree seedlings. Not only are surface soils compacted during grading and spreading of topsoil, but many herbaceous species severely compete with planted trees for moisture and nutrients.

Various treatments and modifications of reclamation practices can be employed to mitigate the effects of soil compaction and herbaceous competition. Some commonly used measures to enhance tree establishment are to reduce the density of compacted soils by ripping, reduce herbaceous cover in selected areas by mechanical or chemical means, and use herbaceous species that are less competitive with trees.

Success in tree planting requires adequate precipitation after planting. Drought is a common cause of planting failures, and the ultimate objective of all the above-mentioned treatments is to increase availability of soil moisture to trees during dry periods.

Reported here are results of two field studies designed to investigate tree seedling survival and growth on three different minesoils, each seeded with various grasses and/or legume mixtures either prior to or at time of tree planting. In the first study, trees planted in April were subjected to a record drought lasting about 10 weeks. Results of this study, now terminated, were reported earlier (Larson and Vimmerstedt 1990). In a

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[2]Merlyn M. Larson, Professor, David A. Kost, Senior Research Associate, and John P. Vimmerstedt, Associate Professor, The Ohio State University, School of Natural Resources, Wooster, OH, USA. Salaries and research support were provided by cooperative agreement with the American Electric Power Service Co. and by State and Federal funds appropriated to the Ohio Agricultural Research and Development Center, The Ohio State University; Manuscript number 223-93 of the Ohio Agricultural Research and Development Center.

second study of similar design installed on the same area, precipitation following tree planting was above average during the first two growing seasons. Preliminary results of this on-going study were reported by Larson et al. (1993). The two studies allowed us to evaluate the effectiveness of various treatments to enhance seedling establishment during both drought and wet years.

Soil Materials

Plots were established on a recently mined area of the Muskingum Mine, Central Ohio Coal Co., located 5 km south of Cumberland, OH. Soil materials and treatments included the following: (1) topsoiled (25 to 30 cm deep), standard graded in accordance with current practice, (2) topsoiled, graded, and then ripped to 30 cm deep to reduce soil compaction, and (3) gray cast overburden, leveled after mining. Each soil material was applied to one of three plots (each 30.5 m by 40.2 m) in each of six blocks.

The chemical and physical properties of topsoil and gray cast overburden on the Muskingum Mine are given in table 1. Cast overburden has higher cation exchange capacity, more exchangeable potassium, calcium, and magnesium, but lower phosphorus than topsoil. Also, cast overburden has less clay, more sand, and a higher rock content than topsoil. pH of both soils was neutral.

After final grading in 1987, 8-32-16 fertilizer was applied at 448 kg/ha. The site also was seeded with annual ryegrass (Lolium multiflorum)and fertilized with 112 kg/ha ammonium nitrate to stimulate growth and help reduce soil erosion before the first study was established in autumn 1987.

Precipitation

Annual precipitation in southeast Ohio averages 907 mm with about 432 mm occurring during April through August (NOAA 1989). In 1988 (the drought year), a severe drought began after the trees were planted on the study area in April. The April through June precipitation was 151 mm, which was 51% below normal based on long-term records of a nearby meteorological station at Cambridge, OH. The drought continued 2 more weeks until mid-July, during which time a rain gage was installed on the study area. Precipitation was

Table 1. Chemical and physical properties of topsoil and gray cast overburden on the Muskingum Mine.

	Topsoil	Overburden
Available nutrients, kg/ha:		
Phosphorus	16	13
Potassium	285	377
Calcium	5,824	14,380
Magnesium	877	1,157
pH	7.0	7.0
Cation Exchange Capacity cmol/kg	17	37
Extractable Na+ µg/ml	22	968
Particle-size analysis, %:		
Sand	28	36
Silt	40	41
Clay	32	23
Rock content >4mm %	18	30

abundant from late July through August (243 mm) and ended the drought.

For the 2 wet years, the April through August precipitation totaled 768 mm (77% above normal for the Cambridge area) in 1989, and 611 mm (41% above normal) in 1990, 320 mm in 1991, 528 mm in 1992, and 451 mm in 1993.

Seeding Herbaceous Species and Tree Planting

Each plot was divided into subplots (each 6.7 m by 12.4 m), which received one of three herbaceous seedings (grasses, legumes, or grass-legume mixture)(table 2) applied either the autumn before tree planting or at time of planting in the spring. In addition to these six treatments, one adjacent subplot was left unseeded but planted with trees. The grass-legume mixture at 56 kg/ha is the standard species and rate used by the Central Ohio Coal Co.

One-half of each subplot was planted with 1-yr-old green ash (*Fraxinus pennsylvanica*) seedlings, and the other one-half was planted with 2-yr-old white pine (*Pinus strobus*) seedlings. Each species was planted in three rows of five trees each, with two of the three rows sprayed in strips 60 cm wide with simazine at 4.5 kg/ha 3 weeks after planting to reduce competition from seeded herbaceous species. Spacing was 1.2 m between trees and 2.4 m between rows. Spacing was maintained in ripped subplots without regard to the location of the rips. In total, 1,890 green ash and 1,890 white pine seedlings were planted on 126 subplots during March 1988. Tree survival counts were made in mid-August 1988, at which time the first study was terminated.

The second study was started on the same area in late August 1988. The plots were sprayed with glyphosate at 3.5 L/ha to remove existing vegetation and prepare the site for seeding herbaceous mixtures. Plots were again divided into subplots, and each received one of the seven treatments described above. Three additional treatments were added in which trees were planted 1 yr after subplots were seeded with one of the three herbaceous types in the spring (i.e., seeded in March 1989 and planted with trees in March 1990). In total, 1,890 green ash and 1,890 white pine seedlings were planted during March 1989, and an additional 1,080 trees of each species were planted in March 1990.

Table 2. Species and seeding rates of 3 herbaceous mixtures used on the study subplots.

	Kg/ha
Grass species	
Orchardgrass (*Dactylis glomerata*)	13.4
Timothy (*Phleum pratense*)	11.2
Perennial ryegrass (*Lolium perenne*)	9.0
Kentucky bluegrass (*Poa pratensis*)	5.6
Total ... grass species	39.2
Legume species	
Ranger alfalfa (*Medicago sativa*)	5.6
Birdsfoot trefoil (*Lotus corniculatus*)	5.6
Mammoth red clover (*Trifolium pratense*)	5.6
Total ... legume species	16.8
Grass-legume mixture	
Total of the above species	56.0

Since application of simazine damaged green ash in the first study, results reported here are based only on data of trees planted in untreated strips. In the second study, application of herbicide was without significant effect on trees, and these data are combined in the tables.

Measurement of Herbaceous Vegetation

The quadrat method was used to measure both cover (defined as the ground area within the canopy of a plant or group of plants expressed as a percentage of the total area of the quadrat) and above ground biomass.

Quadrat size was 70.7 by 70.7 cm. Plant samples were taken in August 1989 and in June of 1990, 1992, and 1993 from 180 subplots. Areas between rows of trees were sampled. Within the quadrat, vegetation was clipped level to the soil surface, placed in paper bags, dried to constant weight at 60° C, and weighed.

Measurement of Trees and Data Analysis

Survival and total height of trees were measured each year. The study design was a randomized split-plot design with six blocks, and the experimental unit was a single five-tree row. Analysis of variance was in accordance with standard procedures for this design (Steel and Torrie 1960). Survival data were subjected to arc sin transformation.

Results

Survival of White Pine

Mean survival of white pine in the first study was only 19% at the end of the drought season. In the second study, survival after 2 wet years was 36%, but few pines were alive at the end of the third year on any subplot regardless of treatment. These minesoils clearly are not suited to white pine, partly due to high soil pH and salinity (table 1). Thus, the results that follow are only for green ash.

Effects of Treatments - Drought Year

In the first study (drought year), the benefits of ripping topsoil to reduce soil compaction were mixed. Ripping increased green ash survival 28% above that of trees on graded topsoil but only when both seeding and tree planting were carried out in the spring (table 3). Ripping was without benefit to trees planted on unseeded subplots or subplots seeded the previous fall. When the three minesoils were seeded the fall before planting, ash survived best on cast overburden at 83%, compared with 58% for trees on graded topsoil and 47% on ripped topsoil (table 3). Ash survival on unseeded subplots averaged 88%, with differences between minesoils not significant.

Seeding grasses or grass-legume mixture the fall before planting resulted in low ash survival (40% to 47%), compared with seeding only legumes (94% survival) or seeding at time of tree planting (81% to 85% survival)(table 4).

Effects of Treatments - Wet Years

In the second (wet years) study, differences in green ash survival among the minesoil and herbaceous seeding treatments were relatively small. Ash survival 4 to 5 yr after planting generally exceeded 90% for all treatments (tables 3 and 4). One exception was that trees planted on cast overburden and seeded the previous spring had 85% survival, which was 9% to 14% lower survival than trees in other treatments (table 3).

Table 3. Percent survival of green ash when planting was followed by a drought (1st study) or by 2 wet years (2nd study) on 3 minesoil types seeded either before tree planting or at time of planting with herbaceous species, or unseeded.[1]

	Time of herbaceous seeding and tree planting			
Seeding time.....	Autumn	Spring	Spring	Unseeded
Planting time.....	Spring	Spring	Spring+1	Spring
	1ST STUDY - DROUGHT YEAR			
	Year 1	Year 1		Year 1
Minesoil type:				
Topsoiled, graded.....	58 c	66 bc	---	87 a
Topsoiled, ripped.....	47 c	94 a	---	83 ab
Cast overburden.....	83 ab	84 ab	---	93 a
	2ND STUDY - WET YEARS			
	Year 5	Year 5	Year 4	Year 5
Minesoil type:				
Topsoiled, graded.....	99 a	99 a	97 a	98 a
Topsoiled, ripped.....	98 a	99 a	96 a	98 a
Cast overburden.....	98 a	94 ab	85 b	97 a

[1] 1st study terminated at end of 1st (1988) growing season; in 2nd study, subplots were seeded in autumn 1988 and spring 1989, trees planted in 1989 (spring) and 1990 (spring+1); survival measured in autumn 1993. Within each study, any 2 survival values not followed by a common letter are significantly different at P=0.05.

Although survival of green ash on cast overburden was generally very good, mean height was only 49.3 cm, which is 31% less than that of trees on topsoil. However, 30.5% of the trees on cast overburden showed visible signs of deer browsing, compared with only 12% for trees on graded or ripped topsoil. Tallest trees were found on subplots in ripped and graded topsoil (86.8 cm and 82.2 cm, respectively) that were seeded with legumes (table 5).

In general, mean tree heights were greatest on subplots seeded with legumes, and heights declined in order from subplots seeded with legumes to grass-legume mix to unseeded to grasses (tables 5 and 6). Browsing of trees for the three seeded herbaceous types was similar, but trees planted in 1990 were more heavily browsed (23.6%) than trees planted in 1989 (15.4%).

In the second (wet years) study, timing of seeding herbaceous species in relation to planting had little effect on tree height after 4 and 5 yr (table 6). In fact, seeding legumes the fall prior to tree planting resulted in taller trees (88.7 cm) than when subplots were seeded at time of planting (79.4 cm) or unseeded (72.5 cm). Even trees on subplots seeded with legumes 1 full yr prior to planting were slightly taller than trees of the same age on unseeded subplots.

Table 4. Percent survival of green ash when planting was followed by a drought (1st study) or by 2 wet yr (2nd study) on subplots seeded with legumes, grasses, or grass-legume mixtures either prior to or at time of tree planting.[1]

	Time of herbaceous seeding and tree planting		
Seeding time.....	Autumn	Spring	Spring
Planting time.....	Spring	Spring	Spring+1
1ST STUDY - DROUGHT YEAR			
	Year 1	Year 1	
Seeded Species:			
Legumes (3 species).....	94 a	81 a	---
Grasses (4 species).....	47 b	82 a	---
Grass-legume mix.....	40 b	85 a	---
2ND STUDY - WET YEARS			
	Year 5	Year 5	Year 4
Seeded Species:			
Legumes (3 species).....	99 a	98 ab	94 ab
Grasses (4 species).....	99 a	96 ab	91 b
Grass-legume mix.....	98 ab	97 ab	94 ab

[1] 1st study terminated at end of 1st (1988) growing season; subplots in 2nd study seeded in autumn 1988 and spring 1989, trees planted in 1989 (spring) and 1990 (spring+1), survival measured in autumn 1993. Within each study, any 2 survival values not followed by a common letters are significantly different at P=0.05.

Table 5. Total height(cm) of green ash trees planted on 3 minesoil types in subplots seeded with legumes, grass-legume (G/L) mixtures, grasses, or unseeded.[1]

	Seeded herbaceous species			
Minesoil type	Legumes	G/L Mix	Grasses	Unseeded
Topsoiled, Graded.....	82.2 a	74.8 b	68.3 c	70.0 bc
Topsoiled, Ripped.....	86.8 a	69.8 bc	60.1 d	69.1 bc
Cast Overburden.....	56.4 de	49.6 ef	46.1 f	45.3 f
Herbaceous mean.....	75.1	64.7	58.2	61.5

[1] Tree heights measured in fall 1993; data of trees planted in 1989 and 1990 combined for the analysis; means followed by 1 or more common letters are not significantly different at P=0.05.

Table 6. Total height (cm) of green ash trees planted in subplots seeded with legumes, grass-legume (G/L) mix, or grasses either prior to or at time of planting, or in unseeded subplots.[1]

Seeding time	Planting time	Seeded herbaceous species			
		Legumes	G/L mix	Grasses	Unseeded
Autumn 1988	Spring 1989.....	88.7 a	74.1 bc	64.7 d	---
Spring 1989	Spring 1989.....	79.4 b	69.0 cd	64.4 d	---
Spring 1989	Spring 1990.....	57.3 a	51.0 ab	45.4 b	---
Unseeded	Spring 1989.....	---	---	---	72.5 bc
Unseeded	Spring 1990.....	---	---	---	50.4 ab

[1] Heights measured in fall 1993, trees planted in spring 1989 completed 5 growing seasons, trees planted in spring 1990 completed 4 growing seasons. Within each planting year, heights not followed by 1 or more common letters are significantly different at P=0.05.

Herbaceous Cover and Biomass

The amount of herbaceous cover 4 to 5 years after seeding on subplots was not related to minesoil type or time of seeding treatments, nor does soil erosion appear to be excessive on any minesoil-herbaceous treatment combination. The amount of fertilizer applied to the subplots was significantly reduced in order to keep herbaceous competition at a low level, and yet meet the minimum standard of 30% ground cover required by Federal law. For the three herbaceous types, above ground biomass was 29% higher for subplots seeded with legumes than for subplots seeded with grasses. Biomass on legume-seeded subplots averaged 183.8 g per 0.5 m^2, and these also had the highest estimated total ground cover at 39.3%, of which 36.3% was legume cover (table 7).

Table 7. Estimated total cover and legume cover, and above ground biomass of subplots seeded with 3 herbaceous types.[1]

Seeded Herbaceous Type	Total cover %	Legume cover %	Above ground biomass g per 0.5 m^2
Legumes.....	39.3 a	14.3 a	183.8 a
Grass/Legume Mix.....	37.9 a	13.9 a	162.1 b
Grasses.....	31.5 b	10.7 a	142.4 c

[1] Measured in early summer 1993; cover estimates based on living plant material above ground; data of subplots seeded in 1989 and 1990 combined; Within each column, any 2 means not followed by a common letter are significant at P=0.05.

Discussion

These results, combined with those of other research, provide a good understanding of the effects of minesoils, herbaceous species, and timing of seeding of herbaceous species on initial establishment of tree seedlings during wet and dry growing seasons.

Survival of White Pine and Green Ash

White pine failed on these minesoils, even when planting was followed by wet growing seasons. Both topsoil and cast overburden, with pH of 7.0, are probably too alkaline for this species. Torbert et al. (1989) noted that white pine grown in soils with pH above 6.5 become chlorotic and stunted.

When tree planting is followed by drought, initial survival of green ash may be better on cast overburden than on topsoiled areas because the seeded herbaceous species appeared to be slow to develop on cast overburden (Larson et al. 1993). Grass species seeded in the fall were first to become established the following spring, and newly planted trees suffered high mortality, whereas legumes seeded the previous fall or grasses seeded at time of planting were much less competitive with trees.

Treatments to enhance establishment of green ash were not needed during wet years. Tree survival was excellent on all minesoils, and seeding herbaceous species did not reduce survival except when grasses were seeded a full year before tree planting. In Illinois, green ash performed better on areas seeded with a grass-legume mixture at time of planting than on areas seeded the previous year (Ashby et al. 1988).

Other research has shown that green ash planted on ungraded spoil in 1947 survived and grew better on plots seeded only with sweetclover (Melilotus officinalis) than on plots seeded with a mixture of lespedeza (Lespedeza cuneata) and orchardgrass (Dactylis glomerata)(Larson and Vimmerstedt 1983). Vogel (1980) reported reduced survival when trees were planted in dense stands of legumes, although growth of surviving trees was later increased. Olah (1988) suggests that 1 yr of herbaceous vegetation be incorporated into the soil before tree planting. The commonly recommended practice is to delay seeding herbaceous species until the same time or after trees are planted (Vogel 1980, Ashby et al. 1988).

Ripping to reduce topsoil compaction did not consistently benefit ash survival in the drought year. It is possible that the ripping operation was not uniformly applied to plots, but this was not measured. The influence of compaction may yet appear in future measurements. In Illinois, black walnut (Juglans nigra) planted on ripped minesoil had 58% greater root depth and 89% greater lateral root spread in the second year than trees planted on graded soils that were not ripped (Philo et al. 1982). There is abundant evidence of detrimental effects of compaction and the importance of good physical properties of replaced topsoil (Andersen et al. 1989, Ashby et al. 1988, Chong et al. 1986, Davidson et al. 1984, Kolar et al. 1981, Pope 1989, Vogel 1981).

Height Growth of Green Ash

Although ash trees survived well on cast overburden, height growth was inhibited the very first year (Larson, et al. 1993) and continues to lag behind that of trees on topsoil after 5 yr. Trees on cast overburden have been somewhat more heavily deer browsed than trees on topsoil, although the reason for this is not apparent.

Seeding legumes instead of grasses improved height growth of ash trees on all minesoils, including cast overburden. After 5 yr, subplots originally seeded with legumes had about one-third of the total cover in legumes. Legumes have also successfully invaded subplots seeded with grasses. The fact that low amounts of fertilizer were used, and tree heights were greatest in legume-seeded subplots suggest that soil nitrogen

deficiency may be a primary growth limiting factor. Others have suggested that nitrogen-fixing trees and shrubs be planted on overburden mixtures without topsoil (Wade et al. 1985).

Cast Overburden as a Resoiling Material

This research suggests that cast overburden is a suitable alternative resoiling material during both wet and dry years if a suitable tree species such as green ash is planted, and the seeded mixtures contain mostly legume species that do not aggressively compete with trees. Cast overburden sustained herbaceous cover as well as topsoil except during the first growing season (Larson et al. 1993). However, additional measures to increase tree growth may be needed if the trees are to be harvested within a normal rotation period.

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DEVELOPMENT OF BROADLEAVED WOODLAND ON COLLIERY AND OPEN PIT COAL MINES IN THE UNITED KINGDOM[1]

R. Neil Humphries[2] and Gordon E. McQuire[3]

Abstract: Broadleaved woodland is an important land use and vegetation type in the United Kingdom (UK), and potentially the most effective landscape and restoration treatment for colliery waste tips and open pit coal sites.

A field-based national survey of collieries in England and Wales in 1986 and 1987 showed that establishment was satisfactory in only half of the schemes, and growth was deemed satisfactory in less than one-fifth. Also, the majority of the plantations had little affinity with woodland types found locally or regionally. The main causes of poor establishment were poor stock viability, and competition with herbaceous ground cover vegetation for available soil water in the immediate post planting period. The poor growth was due to competition for water with herbaceous vegetation and low inherent available water capacity of the soil profiles.

There are standard forestry practices whereby stock quality can be assured, and herbaceous vegetation controlled or eliminated by the use of herbicides. During the restoration of the site, depending on choice of species, adequate soil water can be provided by the selection of appropriate soil types and thicknesses, and adoption of appropriate soil handling and decompaction practices. The low affinity of the plantations with local and regional types was partly due to the planting of non native species and partly due to the failure to match species with site and soil characteristics. There is no reason why woodlands of a local and regional character cannot be established by planting the associated species. A matrix of fast-growing tree and/or shrub species should be used to promote early woodland development. These would be removed during normal management which is essential for the ultimate success of the woodland. Planting schemes should also incorporate woodland structural elements and understorey and ground flora species.

Provided that these measures are fully implemented, significant improvements in establishment, growth, and woodland development on restored sites should be achieved.

Additional Key Words: broadleaved woodlands, establishment, growth, species composition, colliery tip restoration.

Broadleaved Woodlands

Broadleaved woodland is an important land use and vegetation type in the United Kingdom (Peterken 1981, Evans 1984, Rodwell 1991). Potentially, the planting of broadleaved woodland on restored colliery sites and open pit coal mines is often the most effective landscape and restoration treatment. It also provides important wildlife habitats, a source of home-grown timber products, and a means of supporting and diversifying rural economies, as well as environmental benefits such as assisting the redevelopment of soil structure and improving the water quality of catchments.

With recent changes in the emphasis of land use policy in the UK, from agricultural production to environmental quality and community recreation, woodland will become more prominent in restoration schemes than hitherto. There has also been a change over the last 20 yrs in the type of woodland planted, from coniferous plantations for timber production to predominantly broadleaved types for landscape and nature conservation. This is

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[2]R. Neil Humphries, Technical Director, Humphries Rowell Associates, Loughborough, UK.

[3]Gordon E. McQuire, Chief Civil Engineer, British Coal Corporation, Eastwood, Nottinghamshire, UK.

already manifest by the number of major initiatives to create large-scale wooded landscapes in lowland England. These include the new National Forest, covering some 50,000 ha in an area of 200 square miles, and a number of other extensive community woodlands.

The British Coal Corporation is involved to a significant extent in several of these projects, including the new National Forest and the Forest of Mercia in the Midlands, the South Yorkshire Forest, the Great North Forest in County Durham, and the Red Rose Community Forest in Greater Manchester. The new woodlands are to include a number of open pit coal mines and colliery sites. Indeed, the industrial heritage of coal and mineral workings was one reason why the location of the National Forest was chosen (Anon 1993). With the demise of the local coal industry, the Forest was seen as an opportunity for improving the local landscape, encouraging economic revival, and providing a focus for leisure activity and access. Elsewhere, on a local scale, the amount of broadleaved woodland to be planted on restored collieries and open pit sites is likely to continue to increase for the same reasons.

The purpose of this paper is to communicate the lessons learned from planting schemes implemented over the last 10 yrs on collieries and open pit sites in the UK, and to recommend measures that could be beneficially adopted in the current and future schemes.

Past Achievements

Historically, the methods of preparation and establishment, and the composition of woodland planted on restored colliery waste tips and open pit coal sites followed the advice of the local planning authorities and the statutory advisory body, the Forestry Commission. However, the results have often been disappointing and much criticized by both the general public and the planning authority.

In 1986 Humphries Rowell Associates was commissioned by British Coal's Headquarters Technical Department to undertake a national field-based assessment. The study included 16 newly planted and 54 existing schemes at 27 collieries in England and 7 collieries in Wales. In 1987, 21 planting schemes at 5 of the collieries in England and 3 in Wales were closely observed during implementation. The investigation was carried out using the approach we had previously developed to assess colliery tip restoration practices (Humphries et al. 1984).

The rationale behind the approach was that field surveys formed the basis of objective assessment of past and current achievement, using appropriate tests and measurements. Other methodologies, such as literature and document reviews and experimentation, need only be implemented when achievement and standard forestry practices have been shown to be unsatisfactory. From previous assessments, we have demonstrated that poor achievement is generally due to the failure to implement current recommended practices or failure to implement them properly. For this reason practices used during planting and management were also observed in this study.

Establishment

The field survey indicated that establishment was satisfactory in only half of the schemes. A quarter had establishment of less than 50% (table 1). It appeared that most of the mortalities had occurred in the first year following planting.

Table 1. Overall percentage establishment achieved in tree planting schemes on colliery wastes prior to 1988.

	Very unsatisfactory <50	Unsatisfactory 50-70	Satisfactory 70-90	Very satisfactory >90
Percentage of schemes surveyed.....	24	30	36	10

There was no evidence in the field that mineral nutrition, acidity or compaction were factors (Humphries and Benyon 1988). Neither was there evidence that poor establishment was associated with a particular planting practice or the competence of the contractor. Also, low establishment was not confined to any particular species, although the establishment success of birch (*Betula* spp.), pine species (*Pinus* spp.), larches (*Larix* spp.), and alders (*Alnus glutinosa* and *A. incana*) was particularly variable (table 2). Most of the unsatisfactory establishment was recorded in the schemes without a soil cover over the colliery shale, or where the cover was less than 20 cm deep. However, soil cover and depth did not account for all instances of unsatisfactory establishment. In three of the schemes planted in 1986-87, it was due to poor-quality work where the use of a clayey subsoil to cover the spoil had made it difficult to plant the trees properly.

During 1987, a distinct pattern to the failures was observed: i.e., early-season mortality in May and late-season mortality between May and September (table 3). Early season mortality generally accounted for a small proportion of the first-year total, and was confined either to certain species or to isolated areas on the tips. The latter was associated with persistent local waterlogged conditions, mainly at the "break" of slopes and at the "toe" of the tip. The species-specific mortality (birch and alder in England and larch in South Wales) was associated with poor transportation and handling practice, where the roots of these desiccation-sensitive species had been allowed to dry out prior to planting. The late-season mortalities generally accounted for most of the first-year losses, and were largely confined to the species alder, birch, larch, and willows (*Salix* spp.). This was attributed to physiological drought induced by the presence of a moderate to high herbaceous ground cover vegetation around the trees, the affected species being notoriously sensitive to drought. The symptoms included progressive "leaf-burn" and shoot dieback from May onwards. The herbaceous ground cover occurs as volunteer vegetation colonising the spoil and soil or from seed mixtures sown to control surface erosion and run-off.

Other, but only occasional, causes of poor establishment included browsing by rabbits or sheep, diseased stock supplied by the nursery, or extremely alkaline conditions on cement-stabilized tailings.

Table 2. Percentage establishment rates for commonly planted species at end of first year.

	1986			1987		
Species	Range	Median	n[1]	Range	Median	n
Pine	0-73	42	6	77-100	94	13
Birch	26-100	56	8	51-100	77	6
Common alder	34-100	69	5	35-100	72	15
Grey alder	17-100	63	6	70-100	93	4
Swedish whitebeam	91-100	97	3	NA	100	4
Field maple	NA[2]	100	2	NA	100	8
Sycamore	86-100	95	3	NA	100	7
Ash	29-100	82	6	NA	100	8
Oak	NA	80	2	85-100	96	10
Rowan	67-100	90	4	80-100	97	8

[1] n = Number of schemes surveyed.

[2] NA = Not applicable.

Growth

As part of the above surveys in 1986 and 1987 the growth performance of the plantations was assessed (table 4). This indicated that only 19% of the schemes older than 3 to 4 yrs were considered to have achieved satisfactory growth (i.e., >1.5m tall), and 39% were particularly poor (i.e., <0.9m tall).

In some of the unsatisfactory schemes, poor growth was not confined to any particular species, whereas in others birch, pine species, common alder (*Alnus glutinosa*), hawthorn (*Crataegus monogyna*), and field maple (*Acer campestre*) notably achieved greater height and spread than other species. There was no evidence in the field that mineral nutrition, acidity or any other site or soil physical characteristic, such as compaction, was a factor.

Table 3. Percentage mortality of newly planted trees in May and September 1987.

Colliery	May	September
Harworth.....	2.5	26.0
Rufford.....	3.0	17.0
Welbeck.....	1.5	31.5
Thoresby.....	10.8	24.4
Sherwood.....	2.0	51.0
Markham Main.....	0	3.0
South Kirkby.....	0.7	6.7
Oakdale.....	1.0	12.0
Cynheidre.....	3.0	10.5
Wernos.....	5.0	20.0
Mean.....	2.9	20.6

On closer inspection of the data, it became evident that two components were contributing to the variation of tree heights between schemes; these were the amount of extension growth and dieback of shoots. Both are related to the amount of available soil water, and this was partly determined by the amount of herbaceous ground cover around the trees. Hence, competition with herbaceous ground cover vegetation for water was identified as the principal cause of poor growth (Humphries and Benyon 1988).

Woodland Composition and Structure

From the surveys it was also possible to determine the types of woodland planted. The majority (60%) of the broadleaved plantations had little affinity with the types found locally or regionally, and in particular with the oak (*Quercus* spp.) woodland types expected for most of the sites (Humphries and Benyon 1988). Also, most of the schemes (89%) would have been expected to be a species-poor variant of oak woodland associated with shallow base-poor soils, the prevailing conditions on many of the tips (table 5).

Most of the non oak woodland types were dominated by species regarded as introduced non native species (e.g., grey alder (*Alnus incana*), false acacia (*Robinia pseudoacacia*), red oak (*Quercus rubra*), white poplar (*Populus alba*), Swedish whitebeam (*Sorbus intermedia* agg.), Japanese larch (*Larix kaempferi*), Corsican pine (*Pinus nigra* ssp. *laricio*), lodgepole pine (*Pinus contorta*)), or species alien to the expected woodland type (e.g., field maple, ash (*Fraxinus excelsior*), sycamore (*Acer pseudoplatanus*)). However, the non native species have been used widely in the reclamation of colliery spoil in the UK (Jobling and Stevens 1980) because of their perceived better performance, although there is no evidence from our surveys to indicate they perform any better than native species.

Table 4. Median maximum tree heights (m) in plantations of different ages.

	Pre-1981	1981-1983	1983-1985	1985-1986
Median.....	1.00	1.10	0.84	0.76
Upper quartile.....	1.80	1.23	1.00	0.92
Lower quartile.....	0.81	0.85	0.76	0.65
Range.....	0.66-1.89	0.67-1.71	0.49-2.30	0.58-1.30
Number of schemes.....	9	19	27	16

Table 5. The number of woodland types expected to be planted for soil characteristics and the number actually planted.

	Oak woodland types[1]						Other woodland types	Total number of schemes
	Base-poor soils				Base-rich soils			
	Species-poor		Species-rich					
Operational Areas	E[2]	P[3]	E	P	E	P	P	
Nottinghamshire.....	18	0	0	2	0	6	13	21
South Wales.....	14	0	0	4	0	1	9	14
North Yorkshire.....	14	0	0	4	0	1	10	14
South Yorkshire.....	7	0	0	3	0	3	4	10
Western.....	3	0	0	0	0	2	1	3
Total.....	56	0	0	13	0	13	27	63

[1] Hopkinson 1927, Jones 1959.
[2] E = Expected
[3] P = Planted

The woodlands surveyed were typically plantation like in nature, with trees being of an even age, and they often had a regular pattern of species distribution, or were planted in blocks of single species or simple mixtures of species. They also lacked the basic elements that may give rise to structural development in the future. There were no "edge" and "core" structural and composition elements, nor understorey and ground flora components.

Consequences

A major consequence of the low rates of establishment in some of the schemes and the universally poor growth rates was an ineffective landscape treatment, albeit in the short term. Effective landscape treatment of collieries and open pits is often the main concern of the local public. They generally expect a woodland landscape to develop quickly, i.e., within a 5 to 10 yr time frame, and are disappointed when it does not. The results achieved within this time frame have been typically scrub like in appearance with little closure of the canopy (fig. 1).

Other important consequences are the result of the species planted. Sites planted with the non native species in particular are usually neither satisfactory from a landscape point of view nor suited to exploitation for timber products and have dubious benefits for wildlife, although they may be satisfactory for recreational use.

Improved Practice

It was clear from the literature and current guidelines that provisions should have been made at planting and in the post planting maintenance period to ensure that stock was viable and that competition from herbaceous vegetation was controlled. Similarly, provisions should have been made to counter the other causes of poor establishment and growth. The field survey indicated that in many cases either no provisions had been made, or if they had, they were either ineffective or not fully implemented (Humphries and Benyon 1988).

Also at the design stage, often no provisions had been made either for local woodland types and woodland structure or for appropriate soils and soil profiles to provide sufficient soil water for normal growth.

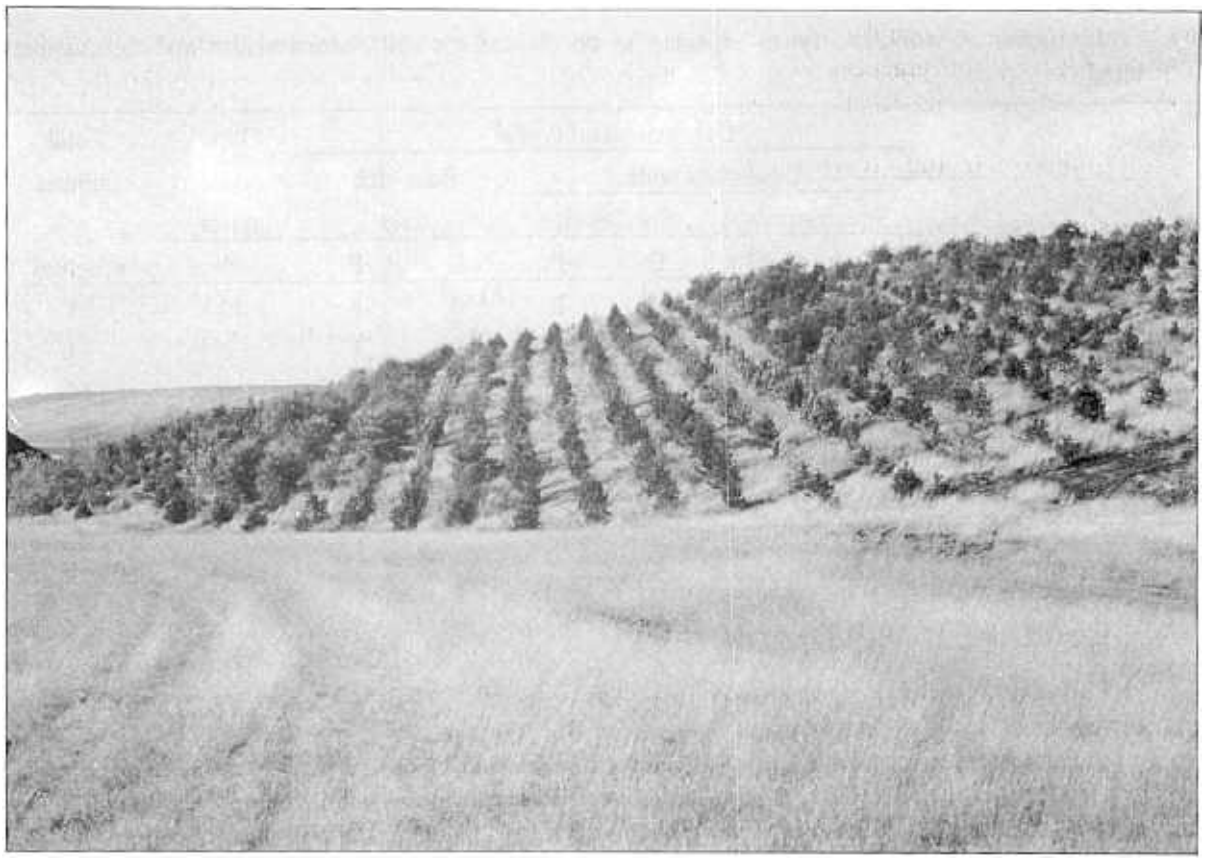

Figure 1. A typical plantation on a colliery waste tip in the Midlands, showing an incomplete tree canopy cover after almost 10 yrs and consisting of a high proportion of non native species.

Establishment

There are proven standard forestry practices that, if properly implemented, can ensure stock viability and control of ground cover vegetation at the planting stage (Evans 1984, Committee for Plant Supply and Establishment 1985, Sale et al. 1986, Davies 1987). Stock viability can be maintained if the roots of the trees and shrubs are protected from drying during handling at the nursery, during transit, and on site at planting. Herbaceous competition can be effectively controlled by the use of approved herbicides, but control must be complete prior to planting and maintained as such throughout the critical first year. Similarly, there are standard measures whereby animal damage can be controlled, such as fencing, individual tree guards, and vermin extermination programs. In the case of waterlogged areas there are a range of measures, including avoidance at planting, a policy of not replanting failures, planting with tolerant species, and drainage.

Growth

Herbicides are also the proven standard forestry practice to deal with competition from herbaceous ground cover vegetation in the post planting period (Davies 1987, Sale et al. 1986). The control around the trees and shrubs must be complete and should be maintained for the second and third year after planting, and longer if necessary.

In addition the thickness and physical characteristics of the soil profile (textural and structural) affect the amount of soil water available to the trees and shrubs and hence growth rates and dieback. Growth can be improved

and dieback minimized through the selection of appropriate soil physical and chemical types and thicknesses, and soil handling and decompaction practices (Humphries and McQuire 1994). Both should be addressed in the formulation of site restoration proposals and detailing of the planting scheme.

Woodland Composition and Structure

Species Composition. The surveys in 1986 and 1987 demonstrated that native tree and shrub species could be established successfully on colliery sites, and the same is true for open pit sites. The evidence suggests that there is no reason why species associated with local and regional semi natural broadleaved woodland types, described recently by Rodwell (1991), cannot be established, provided that the site and soil conditions are appropriate. Hence, there is no need to use non native so-called reclamation species.

Structure. Woodland structure is and will be partly dependent on species composition and planting pattern, but ultimately it will be dependent on management and time.

Planting schemes, where appropriate, should incorporate woodland edge and core mixtures. The former should typically be co-dominated by shrub species and species of the understorey layer. There is no reason why both woodland understorey and ground flora species cannot be planted as part of the schemes.

An important element is the development of the vertical structure of the woodland. This can be accelerated in the early years by the use of fast-growing tree and/or shrub species when planted as a matrix with the other species.

In the UK fast-growing broadleaved and conifer tree species have been used as nurse crops to promote plantation development (Evans, 1984). While such species, in conjunction with slower growing woodland trees and shrubs, were often used in reclamation schemes in the UK from the mid-1960's through the 1970's (Jobling and Stevens 1980), the practice has rarely been accepted since. Recently, the practice has been successfully re-adopted at ARC's Huntley Wood sand quarry in the Midlands. Here oak was planted at 4 m centers in a matrix of common alder at 2 m centers on a former silt lagoon. A "closed" canopy developed within 3 to 4 yrs and, after 5 yrs, the oaks were typically 3 m in height and the alders 5 m. Both externally and internally, the plantation has a woodland appearance (fig. 2).

There are many benefits to planting a matrix of fast growing trees and shrubs. These are largely due to the achievement of a rapid canopy cover which provides a visual effect in the shortest time possible. The matrix usually induces more rapid growth of the other woody species, largely through shelter, but also by providing appropriate light conditions to promote the etiolation of certain species such as oak and beech (*Fagus sylvatica*).

The more rapid closure of the canopy is also necessary for the establishment of woodland ground flora, through reducing competitive ability of aggressive species and creating appropriate light conditions (Finegan et al. 1983, Buckley and Knight 1989, Street and Mond 1992). The more rapid closure of the canopy can also be beneficial by reducing the amount of effort required to control ground cover vegetation competing with the trees and shrubs. Other benefits include the earlier provision of cover, which is beneficial for birds and other wildlife.

Suitable matrix species for the UK are those typical of primary colonization of impoverished soils and early successional vegetation (Whyte and Sisam 1949, Finegan et al. 1983, Humphries and Guarino 1987). These include native birches, alders, and willows, and shrubs such as buddleia (*Buddleja davidii*). For more sheltered sites and fertile soils, where conditions are more synonymous with secondary succession, suitable species include the shrubs hawthorn and blackthorn (*Prunus spinosa*). If the matrix species used are alien to the proposed woodland type, they should be totally removed during subsequent management.

Figure 2. Inside a plantation on a former slurry lagoon in the Midlands, showing complete canopy closure within 5 yrs and the development of woodland conditions. The woodland consists only of native species.

Woodland management is probably the most critical aspect of woodland development, and the one that has been most neglected. It is particularly important where a matrix of faster growing species is planted. A thinning program must be implemented. In the first 20 yrs two thinnings would normally be expected: between years 6 and 10 and between years 11 and 20 (Woolhouse personal communication).

The thinning has several benefits as well as promoting the continued development of the woodland. From a timber point of view it enables the selection of the "best" trees and provides a source of material for fencing and other products such as wood chips, and hence a potential income which can be offset against the management costs. Alternatively the material can be left within the wood as a source of organic matter, which can promote soil development and habitat diversification. Thinning must be carefully planned, however, if the aim is to promote the development of the ground flora and increase its value as wildlife habitat.

Conclusions

Provided that the measures recommended above are fully implemented, along with other good standard forestry and site restoration practices (Evans 1984, Bacon and Humphries 1987, Humphries and McQuire 1994), significant improvements in establishment growth and development of woodland schemes on colliery wastes and restored open pit sites should be achieved. This, together with the use of fast-growing tree and shrub species as a matrix should ensure that both visually and functionally woodland development is accelerated, at least in the short term.

There need be no inhibition about establishing woodlands with local and regional characteristics, in terms of composition and structure. The determining factors are site and soil conditions and management practice. This should also result in a more satisfactory landscape treatment and be more satisfactory from a wildlife point of view.

Guidelines for Broadleaved Woodlands

To achieve more successful and cost-effective restoration, along with the associated environmental, landscape, wildlife, and community benefits, broadleaved woodlands of a local and regional character should be established wherever possible and appropriate.

The choice of woodland composition and structure, both for the area of woodland generally and locally within it, must be related to site and soil conditions, and the proposed management regime and use (e.g., source of wood products, amenity or wildlife). The former is to be determined by appropriate desk and field assessments, and the latter through liaison with the intended users and administrators.

There is no need to use non native species. The use of matrix species to promote woodland development is strongly recommended. Where it is necessary to use species alien to the woodland type as the matrix, they must eventually be removed during later management.

The planting plan should include the fundamental structural elements of core and edges, understorey and field layer. Additional structural characteristics and diversity (i.e., glades and rides) should be introduced during management in the longer term.

It is essential that current recommended forestry and reclamation practices be fully implemented. Viable stock must be supplied and planted; this will need to be specified and checked. During the establishment phase, it is essential that any herbaceous ground cover around the newly planted trees be completely controlled for the first 3 yrs. Where a ground cover is not required to control soil erosion and a woodland flora is not planted, the whole area may be kept bare. There needs to be provision to control animal damage by fencing, the use of guards, and/or extermination methods.

Finally, there needs to be a detailed planting plan that gives all the necessary details of which plants are to be planted where, and what amelioration work, etc., is required. The plan should also include a detailed management plan for at least the first 20 yrs, which should be constantly reviewed on a 3 to 5 yr basis. Ultimately the success of the woodland depends on post planting management, whether it be replacements or thinnings.

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TREE ESTABLISHMENT ON PHOSPHATE MINED LANDS IN FLORIDA AS AFFECTED BY PLANT INTERACTIONS [1]

by

Steven G. Richardson, Andre F. Clewell and Curt D. Johnson [2]

Abstract: The effects of plant competition on tree growth and survival on mined overburden in central Florida were assessed in periodically weeded and unweeded plots of primrose willow (*Ludwigia peruviana*), a wetland shrub, and of saltbush (*Baccharis halimifolia*), an upland shrub. In an additional study, wax myrtle (*Myrica cerifera*), slash pine (*Pinus elliotii*) and bushy beardgrass (*Andropogon glomeratus*) were evaluated as nurse crops, and their effects on tree growth and survival were compared with the effects of a hay mulch, an established perennial cover of bahiagrass (*Paspalum notatum*), and a temporary winter cover of annual grasses (cereal rye, *Secale cereale*, plus ryegrass, *Lolium perenne*). After 4 growing seasons, competition from primrose willow at a reclaimed wetland site has had only a minor effect on baldcypress (*Taxodium distichum*) growth. On a drier reclaimed upland site, saltbush competition cut laurel oak growth (*Quercus laurifolia*) in half but only reduced live oak (*Quercus virginiana*) growth by 20%. Interplanting of wax myrtle as a woody nurse crop, or application of a hay mulch each enhanced the growth of laurel oak trees on a mesic overburden site in comparison with the temporary annual grass cover crop. The slash pine and bahiagrass were not significantly different from the annual grass control in their effects on laurel oak growth, but bahiagrass caused a slight reduction in tree survival after four years. The bushy beardgrass treatment resulted in reduced tree growth, but this appeared to be related more to the rapid invasion of other plant species than to the bushy beardgrass itself, which was rather sparse.

Additional Key Words: *Ludwigia, Taxodium, Myrica, Baccharis, Quercus*

Introduction

The warm, humid subtropical climate of central Florida, combined with the fertility of the overburden soils created by surface mining for phosphate (more fertile than the native sandy soils), promotes rapid plant growth and increases the potential for competition from less desirable plant species. The possible value of reducing plant competion to promote tree establishment and growth must also be balanced against the need to control erosion. Not all plant interactions are necessarily negative. The concepts of plant succession and the intentional practice of planting nurse crops are based on the premise that some plants may modify the environment to favor other plant species.

Plant Competition

Florida regulations require that "nuisance" plants, such as **primrose willow** (*Ludwigia peruviana*), a native flowering shrub, must be limited to less than 10% of the total cover on forested wetlands. To comply with these requirements on reforested wetlands, the primrose willow must be controlled chemically with repeated applications of herbicides, mechanically, or by labor-intensive manual means. But is this noxious weed control necessary ecologically or just aesthetically? There is growing speculation that the desired tree species will eventually overtop the weeds and shade them out. Perhaps patience is required to let plant succession solve the problem.

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[2] Dr. S.G. Richardson is Director of Reclamation Research and Mr. C.D. Johnson is Senior Biological Scientist, Florida Institute of Phosphate Research, Bartow, FL. Dr. A.F. Clewell is President, A.F. Clewell, Inc., Quincy, FL.

Saltbush (*Baccharis halimifolia*) is a native Florida upland shrub that commonly invades phosphate mined lands. Its effects on tree establishment had not been documented prior to this study.

Bahiagrass (*Paspalum notatum*) is an exotic grass commonly used in the reclamation of phosphate mined lands in Florida. It is not uncommon for trees to be planted after sites have been stabilized with bahiagrass, but bahiagrass may inhibit tree growth (Fisher and Adrian 1981).

Nurse Crops

Nurse crops are planted as relatively short term cover crops to facilitate the establishment of another crop, in this case forest trees. Nurse crops may reduce the drying effects of sun and wind, augment soil nitrogen (through N-fixing symbionts), add organic matter to the soil (through decomposition of leaf litter and roots), physically condition the soil (e.g. loosen compacted soil through root action), or a combination of these. Woody nurse crops may help shade out highly competitive, but shade-intolerant plants, while allowing shade tolerant forest trees to become established and grow. The following potential nurse crops were tested in this study.

Wax Myrtle (*Myrica cerifera*) is a large shrub or small tree native to central Florida. It sometimes occurs in the understory of riverine forests and commonly colonizes reclaimed mines. Its evergreen foliage provides year-round shade and protection from wind. Actinomycetes associated with wax myrtle roots fix nitrogen (Permar and Fisher 1983). Plass (1977) found that interplanting of European alder (*Alnus glutinosa*), a nitrogen fixing tree, among several tree species on mine spoils in Kentucky enhanced growth of the other trees. Wax myrtle may serve a similar purpose in Florida.

Slash Pine (*Pinus elliottii*) is a pioneer species which is very tolerant of poor soils and a wide range of site conditions. Southeastern pines generally occur in fallow field succession (Kurz 1942, Snedaker and Lugo 1972) and could aid hardwood establishment. A nurse crop such as slash pine can be easily removed mechanically without fear of resprouting, or if we are patient, the hardwoods are expected to eventually replace the shade-intolerant pine.

Bushy Beardgrass (*Andropogon glomeratus*) is an indigenous bunchgrass that frequently dominates recently abandoned croplands with relatively wet soils (Kurz 1942). Trees readily colonize bushy beardgrass dominated sites, and bushy beardgrass, therefore, seems to serve as a natural nurse crop. Where no other perennial cover has been planted, bushy beardgrass frequently volunteers on mined lands over a period of several years and largely replaces other weedy colonizing plants. There is a possibility of accelerating this natural process by intentionally planting trees and concurrently planting bushy beardgrass, or its close relative broomsedge (*Andropogon virginicus*), as a nurse crop on freshly reclaimed land. Broomsedge often occurs on drier sites than bushy beardgrass.

Objectives

The specific objectives of this study were to: (1) evaluate the competition effect of primrose willow on baldcypress and the competitive effect of saltbush on live oak and laurel oak, and (2) evaluate the effects of 3 potential nurse species (wax myrtle, slash pine and bushy beardgrass) on growth and survival of laurel oak in comparison with the effects of an annual grass cover crop.

Methods

Competition

To evaluate the effects of primrose willow competition on tree growth, a natural stand of primrose willow on a seepage wetland on mined land at the IMC Fertilizer Co.'s Clear Springs Mine was cut to the ground in the dry season with a tractor-drawn mower (to approximate postmining conditions) prior to tree planting. Primrose willow was allowed to regrow on the "unweeded" study plots while the "weeded" plots were periodically mowed

with a steel-bladed, gasoline powered mechanical trimmer. A similar approach was taken in the study of saltbush competition. Natural stands of saltbush on an upland overburden site at the Clear Springs Mine were utilized, but because saltbush does not resprout to any great extent, only the weeded plots were mowed with a mechanical trimmer.

There were eight replicate blocks of paired weeded and unweeded plots. Each individual competition plot had 5 trees of each species for a total of 40 trees per species per treatment. Laurel oak (*Quercus laurifolia*) and live oak (*Quercus virginiana*) were planted in the saltbush plots, and baldcypress (*Taxodium distichum*) was planted in the primrose willow plots. The trees were gallon-sized nursery stock (15 cm diameter by 15 cm depth pots) and were planted 2 m apart.

Nurse Crops and Cover Crops

The studies of nurse crops and cover crops were conducted at IMC Fertilizer Co.'s Noralyn and Clear Springs phosphate mines, located south of Bartow in central Florida. At each mine, two sites (total of four sites) were established on areas of recently (1985-87) reclaimed overburden. The overburden was in the loamy sand textural classification (average of 83% sand, 7% silt, and 10% clay), and there were few rocks or pebbles. Overburden pH averaged 5.7, and Mehlich III extractable nutrients included 22 mg/kg of K and 650 mg/kg of P. The sites were considered mesic. Annual precipitation measured at a station within 6 miles of the study sites was 100, 87, 121 and 96 cm for 1989, 1990, 1991 and 1992, compared to a long term average of 115 cm per year. Each individual test plot was 0.028 ha in area, and each individual treatment was replicated four times (once at each site).

The study sites had been stabilized with mostly bahiagrass cover, but to better simulate freshly graded overburden typical of postmining conditions, the established vegetation and 15 to 20 cm of surface overburden were removed with bulldozers and the sites were graded during 1988. The bare areas, except for the bushy beardgrass treated plots, were disked and broadcast seeded with 50 kg of cereal rye (*Secale cereale*) and 22 kg of ryegrass (*Lolium perenne*) per ha in October 1988. All the scalped plots were fertilized at the time of seeding with 45 kg N, 45 kg P_2O_5 and 45 kg K_2O per ha. The primary purpose of the fertilizer was to promote establishment and growth of the cover crop. The rye plus ryegrass treatment alone was considered as the "control."

Gallon-sized nursery stock of laurel oak was planted in all trials. Outplanting occurred from January 30 through February 10, 1989. All nurse crops and soil amendments were in place by February 1989. Each tree planting location was marked with a stake wire flag, and each tree was tagged with a permanent aluminum tag with an individual identification number. All test trees were measured for initial height shortly after planting. Trees that died during the initial winter dry season were replanted in the following wet season (July 10-21, 1989) from extra trees of the original stock that were reserved at the nursery. The replanting was done to maintain adequate numbers of trees for statistical analysis of growth parameters in subsequent years.

Woody nurse species (tubeling slash pine and gallon wax myrtle) were planted between rows of test trees (1 m from test trees) in early February 1989. The herbaceous nurse crop, bushy beardgrass, was planted in late October 1988, prior to tree planting, by scattering recently harvested panicles and lightly disking the surface soil.

A hay mulch treatment, which was expected to reduce plant competition below that in the rye plus ryegrass control, was included in the study. The test plots were seeded and fertilized as above, but in addition, squares of bahiagrass hay about 8 cm thick were sliced from hay bales, cut in half with a machete, and centered around each tree after it was planted.

Established stands of bahiagrass near the nurse crop test plots were planted with laurel oak in the same manner as the nurse crop tests except that the bahiagrass plots were not fertilized.

Measurements

Tree heights and stem diameters at a 15-cm height were measured at the time of planting and each winter thereafter. Survival was assessed at the time of summer replanting and each winter thereafter. Crown diameters (average of two perpendicular measurements) were determined during the fourth growing season. At the end of the fourth growing season, stem diameter at breast height (DBH), rather than at a 15-cm height, was measured on baldcypress because of the basal swelling of the rapidly growing trees. Growth data were only analyzed and presented for those trees still alive at the end of the fourth growing season.

Results and Discussion

Competition

By the end of four growing seasons, stem diameter at breast height (DBH) and crown diameter of baldcypress were about one-third greater on plots where primrose willow was repeatedly cut to the ground (Table 1). Baldcypress trees were about 13% taller on the weeded than on the unweeded plots, but this difference was only statistically significant at the 0.10 level. It should be noted that more than half of the baldcypress trees had overtopped the 2.7-m-tall primrose willow on the unweeded plots. Height growth of baldcypress was actually greater the first year on the unweeded primrose willow plots, suggesting a possible beneficial nurse crop effect (Table 2). However, baldcypress growth was clearly better on the weeded plots in the third and fourth growing seasons. Primrose willow seems to only slightly inhibit the growth of baldcypress, and the fact that the majority of the baldcypress has overtopped the primrose willow in four years, suggests that the baldcypress may eventually shade out the primrose willow. Elderberry (*Sambucus canadensis*) intermingled with the primrose willow at the study site appeared to be doing well, and at another site, Clewell and Raymond (1991) have observed good growth of planted trees and even the colonization of new seedlings of such species as popash (*Fraxinus caroliniana*) and red maple in the presence of primrose willow. Intensive and expensive efforts to control primrose willow in the early stages of forested wetland development may not be necessary.

The saltbush competition study site was drier than the nurse crop study sites, and laurel oak growth was correspondingly less (compare Tables 3 and 5). Live oak was better adapted to the saltbush study site and its growth was much greater than that of laurel oak (Table 3). Live oak height and stem diameter after four years were about 20% less in the presence of saltbush than with saltbush removed, but crown diameter and survival were not significantly affected by saltbush presence. The less vigorous laurel oak was inhibited more by saltbush competition than was live oak; laurel oak heights and crown diameters were 50% less and survival was 18% less in the presence of saltbush than without it. Height growth of live oak and laurel oak was not affected by saltbush in the first growing season, but inhibition by saltbush was evident in subsequent years (Table 4).

Table 1. Primrose willow effect on size (cm) and survival of baldcypress after four growing seasons.

Treatment	Height	DBH	Crown Diameter	Percent Survival
Unweeded	327	3.6	109	98
Weeded	377	5.0*	145*	95

* Significantly different from unweeded control at 0.05 level

Table 2. Primrose willow effect on height growth (cm) of baldcypress during four growing seasons.

	Growing Season				
	1	2	3	4	Total
Unweeded	51	73	72	57	253
Weeded	29*	89	94*	93*	306

* Significantly different from unweeded control at 0.05 level

Table 3. Saltbush effect on size (cm) and survival of live oak and laurel oak after four growing seasons.

		Height	Stem Diameter	Crown Diameter	Percent Survival
Live Oak					
	Unweeded	201	3.0	95	93
	Weeded	242*	3.8*	107	95
Laurel Oak					
	Unweeded	62	0.8	32	78
	Weeded	93*	1.1*	49*	95*

* Significantly different from unweeded control at 0.05 level

Table 4. Saltbush effects on height growth (cm) of live oak and laurel oak.

		Growing Season				
		1	2	3	4	Total
Live Oak						
	Unweeded	11	31	58	49	149
	Weeded	11	45	63	72*	191*
Laurel Oak						
	Unweeded	6	1	8	13	28
	Weeded	7	11	21*	18	57*

* Significantly different from unweeded control at 0.05 level

Nurse Crops and Cover Crops

After four growing seasons, the wax myrtle nurse crop and the hay mulch had resulted in greater laurel oak heights and crown diameters than did the control treatment (Table 5). The hay mulch also caused increases in stem diameter. Planting of bushy beardgrass instead of rye and ryegrass resulted in decreased height and stem diameter of laurel oak. The beneficial effect of the hay mulch on height growth of laurel oak was apparent the first growing season, whereas the wax myrtle nurse crop did not have any apparent beneficial effects until the third year (Table 6). Even the slash pine nurse crop caused a significant increase in laurel oak height growth in the third year, and although the cumulative effect after four years was not quite statistically significant, the pine nurse crop did seem to promote laurel oak growth compared to the control. The stunting of laurel oak in the bushy beardgrass treatment became evident in the second growing season.

Table 5. Effects of nurse crops, cover crops, and hay mulch on size (cm) and survival of laurel oak after four growing seasons.

	Height	Stem Diameter	Crown Diameter	Percent Survival
Control	143	1.67	75	93
Pine	174	1.78	79	96
Wax Myrtle	183*	1.74	88*	85
Bushy Beardgrass	99*	1.23*	63	90
Hay Mulch	184*	2.28*	100*	94
Bahiagrass	136	2.00	81	78*

* Significantly different from control at 0.05 level

Table 6. Effects of nurse crops, cover crops, and hay mulch on height growth (cm) of laurel oak.

	Growing Season				
	1	2	3	4	Total
Control	3	12	29	38	82
Pine	6	14	44*	47	111
Wax Myrtle	3	16	43*	55*	117*
Bushy Beardgrass	2	0*	5*	26*	33*
Hay Mulch	9*	17*	49*	49	126*
Bahiagrass	2	4	19	39	65

* Significantly different from control at 0.05 level

The enhancement of laurel oak growth by the wax myrtle nurse crop may have been due to two factors. First, the dense canopy effectively shaded out most of the weedy plants found on plots with the other treatments. The accumulation of leaf litter may also have had some effect on weed germination. Apparently, laurel oak is sufficiently shade tolerant to grow within the wax myrtle canopy. Second, wax myrtle fixes nitrogen through an actinomycete symbiont (Permar and Fisher 1983). The slash pine may also have suppressed weeds but not to the same extent as wax myrtle. The hay mulch inhibited weeds and probably helped reduce soil moisture loss as well. The stunting of trees in the bushy beardgrass treatment probably had nothing to do with the presence of the limited bushy beardgrass cover; rather, this warm season grass was slow to establish and allowed rapid invasion of potentially competitive plants, including *Indigofera hirsuta, Cassia nictitans, Baccharis halimifolia, Ambrosia artemisifolia, Eupatorium capillifolium, Paspalum notatum, Andropogon virginicus, Rhyncheletrum repens,* and *Cynodon dactylon*. On the control and other treatments, the winter annual grasses (rye and ryegrass) became quickly established and helped inhibit invasion of other plants. Even after the rye and ryegrass died back in the summer the dead plant material continued to serve as a mulch and slowed, although it did not stop, plant invasion.

Bahiagrass (*Paspalum notatum*) and other grasses have been observed to inhibit tree growth (Fisher and Adrian 1981, Plass 1968, Smith 1989). Laurel oak trees were planted into established stands of bahiagrass near the nurse crop and soil amendment study plots, and although survival was reduced slightly, it appears that bahiagrass was no more harmful to laurel oak growth than the volunteer vegetation in the control plots (Tables 5 and 6).

Conclusions and Recommendations

Primrose willow competition was not a major factor in the establishment and growth of baldcypress on a seepage wetland site. Primrose willow even appeared to have a beneficial nurse crop effect on bald cypress the first year. Mechanical and chemical means available for controlling primrose willow may often do more harm than good. For example, although baldcypress may be somewhat tolerant of the broadleaf herbicides used to control primrose willow, many other tree and understory species are not. Our results suggest that intensive and repeated efforts to control primrose willow may not be necessary when baldcypress is planted. Baldcypress should overtop primrose willow in a few years and, if it is planted densely enough (e.g. 1.7 to 2.0 m apart), should eventually shade out the primrose willow.

Saltbush competition with live oak on an upland site was not serious enough to justify the expense of control measures. In contrast, laurel oak growth would likely be aided by control of saltbush on a similar site. However, laurel oak should probably not even be planted on a site where it is not well adapted.

Wax myrtle appears to be a useful nurse crop for interplanting among desired hardwoods such as laurel oak on mesic sites. The beneficial effects of the wax myrtle became apparent in the third growing season when its canopy cover exceeded 50%. We are uncertain what the relative contributions of symbiotic nitrogen fixation or the inhibition of weeds by shading may be to wax myrtle's enhancement of laurel oak growth . The optimum density for interplanting of wax myrtle has not yet been determined, but we feel that wax myrtle may not need to be planted as densely as in this study.

The hay mat mulch around the base of each newly planted tree was beneficial the first growing season, and although the mulch disappeared in the third growing season, the head start in growth was valuable.

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INFLUENCE OF SEEDING LEVEL UPON PLANT COMMUNITY DYNAMICS ON RECLAIMED MINED LANDS IN PENNSYLVANIA[1]

Brian E. Washburn[2], H. Glenn Hughes[3], and Gerald L. Storm[4]

Abstract: Current mined land reclamation focusing on dense herbaceous cover may result in low plant species diversity and adversely affect survival of planted tree seedlings. Reducing the amount of grasses and legumes by alternate seeded and unseeded strips may reduce competition among seeded herbaceous plants, invading species, and planted woody species. During 1991-93, we evaluated plant community development at six sites reclaimed in the bituminous coal region in north-central Pennsylvania. One-half of each site received full seeding, and the other half was strip seeded. The full seeding treatment (high-competition) consisted of areas completely (100%) planted with a herbaceous seed mixture. The strip seeding treatment (low-competition) consisted of alternate seeded and unseeded strips, which resulted in 50% of the area being planted with the herbaceous mixture. We visually estimated the amount of plant cover during each growing season by sampling 0.5- by 1.0-m (0.5 m^2) plots in each treatment at each site. Survival of six species of native hardwood seedlings planted in the treatments was determined by examining each seedling after each of two or three growing seasons. Plants from the seed mixtures and other plants invaded the unseeded strips within 2 to 3 years. Seeded annuals were present only in the first growing season, whereas seeded perennials were present in all years and increased in each successive growing season. Species richness and percent cover of invading species varied by site, and the sites reclaimed in 1991 had more invaders than sites reclaimed in 1992. Survival of all six species of hardwoods was higher in low-competition treatments compared to high-competition treatments on sites reclaimed in 1991 and 1992. This study provided evidence that a 50% reduction in planted herbacous cover resulted in increased survival of native hardwood species.

Additional Key Words: reforestation, competition, plant community development.

Introduction

Revegetation of mined lands currently focuses on establishing dense herbaceous cover to stabilize soil and control erosion and sedimentation. Recontouring, liming and fertilizing, seeding with grasses and legumes, and mulching is the most cost-effective way to satisfy current regulatory requirements and recover bond monies. Grasses and legumes are commonly used in reclamation because of their vigor, availability of seed sources, and reported success in erosion control (Ashby et al. 1988).

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[2] B.E. Washburn, Graduate Research Assistant, School of Forest Resources, The Pennsylvania State University University Park, PA, USA.

[3] H.G. Hughes, Associate Professor of Wildlife Technology, The Pennsylvania State University, DuBois, PA, USA.

[4] G.L. Storm, Adjunct Associate Professor of Wildlife Management, PA Coop. Fish & Wildl. Res. Unit, School of Forest Resources, The Pennsylvania State University, University Park, PA, USA.

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Sites reclaimed with grasses and legumes are characterized by low species diversity and often lack woody vegetation. Previous research has shown that dense herbaceous cover may reduce the survival and growth of tree seedlings (Ashby 1990, Ashby et al. 1988, Davidson et al. 1984, Hughes and Garthe 1989, Smith 1980, Vogel 1980). Wade (1989) found that planted grasses used in reclamation are very competitive and impede the invasion of native species.

Competition among plants may be an important factor in community development on reclaimed mined lands. More information on the effects of grasses and legumes on survival of planted tree seedlings is needed in areas of western Pennsylvania where management of reclaimed mined lands may involve reforestation using native hardwoods.

The objectives of our study were to (1) describe and compare the development of herbaceous cover at sites treated with full seeding (100%) and strip seeding (50%), (2) determine plant community composition (consisting of species in seed mixtures, contaminants in the seed mixtures, and invaders from adjacent areas) within and among sites and among years, and (3) compare survival of six species of hardwoods under full and strip seeding treatments.

Methods

Six study sites, three each in 1991 and 1992, were established in the bituminous coal region of western Pennsylvania. At each site, we created two levels (low and high) of herbaceous cover. Herbaceous mixtures were seeded in April and May of 1991 and 1992. High competition treatments were planted using a seed drill with all ports open so that the area was completely (100%) seeded. Low competition treatments were planted using a seed drill with half of the ports plugged so that the area consisted of alternating seeded and unseeded strips, which resulted in 50% of the total area being seeded. Planted seed mixtures varied by site. The seed mixture planted on the Brandy Camp (1991) and Harmony (1991) sites consisted of oats (Avena sativa) L., perennial ryegrass (Lolium perenne) L., timothy (Phleum pratense) L., birdsfoot trefoil (Lotus corniculatus) L., and alsike clover (Trifolium hybridum) L. The seed mixture planted on the Brockport (1992) and Clearfield (1992) sites consisted of oats, perennial ryegrass, timothy, red top (Agrostis alba) L., birdsfoot trefoil, and alsike clover. The seed mixture planted on the Kylertown I (1991) and Kylertown II (1992) sites consisted of buckwheat (Fagopyrum sagittatum) Gilib., oats, K-31 tall fescue (Festuca arundinacea) Schreb., timothy, red top, birdsfoot trefoil, alsike clover, and red clover (Trifolium pratense) L.

One-year-old tree seedlings were hand-planted in each high-competition treatment and in the unseeded component of each low-competition treatment in May and June of 1991 and April and May of 1992. Seedlings were planted immediately following seeding of herbaceous mixtures. Species planted in 1991 were black birch (Betula lenta) L., black locust (Robinia pseudoacacia) L., quaking aspen (Populus tremuloides) Loeve & Loeve, red oak (Quercus rubra) L., and red maple (Acer rubrum) L. Species planted in 1992 were black cherry (Prunus serotina) Ehrh., black locust, quaking aspen, red oak, and red maple.

After the first growing season, deer exclosures were constructed around one-half of each site, resulting in a randomized split-split plot design with two replications per site. Determining the effect of deer browsing on tree seedling survival involves long-term study. Therefore, only survival data of seedlings inside the deer exclosures are presented in this report.

Percent cover of each plant species and all plant species combined was ocularly estimated (Barbour et al. 1987) by sampling in 0.5- by 1.0-m plots. Twenty-four plots were measured in each high-competition and low-competition treatment. The low-competition treatments consisted of seeded and unseeded components, each 0.8 m in width. Plant cover was measured at sites established in 1991 during August and September of 1991, July and August of 1992, and July and August of 1993. Plant cover was measured at sites established in 1992 during July and August of 1992 and July and August of 1993. The 0.5- by 1.0-m sampling plots were oriented so that separate

measurements were made in the seeded and unseeded components of the low-competition treatment. Seeded components were referred to as "seeded low-competition," whereas unseeded components of the low-competition treatment were referred to as "unseeded low-competition." Species not part of the prescribed seed mixture were contaminants in the seed mixture or invaders from surrounding areas. We did not determine which species were contaminants or invaders.

Each tree seedling inside the deer exclosures was examined and recorded as alive or dead after each growing season. Survival was determined in September of 1991, 1992, and 1993 on sites planted in 1991. Survival was determined in September of 1992 and 1993 on sites planted in 1992.

We used analysis of variance (ANOVA) techniques to test for year and seeding treatment differences in the amount of plant cover. Least-squares means procedures were used for multiple comparison tests among seeding treatments within each year (Neter et al. 1990).

Results and Discussion

Because of differences in climatic conditions between 1991 and 1992 (Washburn et at. 1992), we present separate results for sites reclaimed in 1991 and 1992. Thus, analytical comparisons among years and seeding treatments were also conducted separately for sites reclaimed in 1991 and 1992.

Herbaceous Cover - 1991 Sites

Seeded perennial cover (grasses and legumes) increased significantly ($p<0.05$) in high- and low-competition treatments at all three sites from 1991 through 1993 (fig. 1). The increase in seeded perennial cover appeared to be more pronounced between 1991 and 1992 compared to 1992 and 1993. As expected, seeded perennial cover in the unseeded component of the low-competition treatments was significantly lower ($p<0.05$) than in the high-competition and in the seeded component of the low-competition treatments (fig. 1).

Seeded annual cover (grains and forbs) was present in 1991 and consisted of oats or oats and buckwheat. These seeded annuals were absent on all three sites in 1992 and 1993.

Plant cover of contaminant and invader species was virtually absent at Brandy Camp and Kylertown I in 1991, but increased in 1992 and 1993 (fig. 1). Considerable amounts of contaminant and invader cover were present at Harmony in all 3 years. The amount of contaminant and invader cover at the Harmony site was higher than at the other two sites. High amounts of contaminant and invader cover at Harmony was probably due to greater immigration of seed from adjacent forest communities.

The number of species of contaminant and invader plants increased in all treatments at all three sites from 1991 through 1993 (table 1). Compared to climatic conditions in 1991, conditions during 1992 and 1993 were more favorable to plants, allowing for increased numbers of contaminant and invader species. Higher numbers of contaminant and invader species were present in unseeded component of low-competition treatments compared to high-competition and low-competition seeded components in 1992 and 1993. This was expected due to lower amounts of plant cover and therefore less competition in the unseeded component of the low-competition treatments.

The Harmony site had considerably more contaminant and invader species in all years, and the number at the Brandy Camp and Kylertown I sites were similar (table 1). Nine species of invasive plants were found at the Harmony site but not at the other two sites. The occurrence of sedges (Carex sp.) and cattails (Typha spp.) at Harmony and not at the other sites was probably due to more small pools of water at the Harmony site.

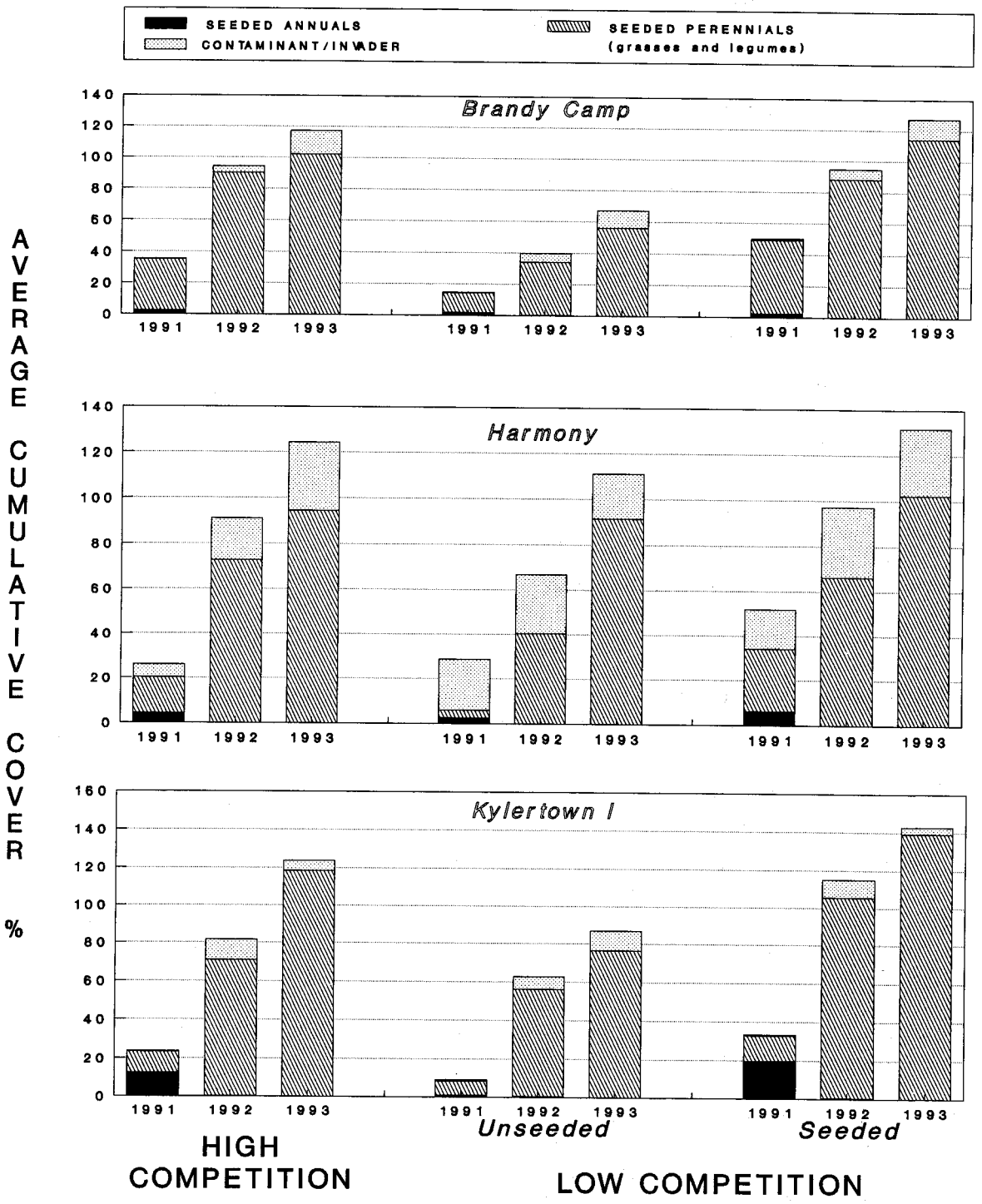


Figure 1. Average cumulative percent cover of seeded annuals, seeded perennials, and contaminant and invader species under two seeding treatments at the Brandy Camp, Harmony, and Kylertown I sites from 1991 through 1993.

Table 1. Number of contaminant and invader species present in high- and low-competition treatments at seeding at three sites reclaimed in 1991.

	Brandy Camp			Harmony			Kylertown I		
	High	Low		High	Low		High	Low	
Year		Unseeded[1]	Seeded[2]		Unseeded[1]	Seeded[2]		Unseeded[1]	Seeded[2]
1991	8	6	7	15	18	15	7	6	4
1992	9	13	7	18	30	22	8	15	6
1993	16	17	8	22	31	31	8	16	7

[1] Unseeded refers to the unseeded low-competition treatments.
[2] Seeded refers to the seeded low-competition treatments.

A total of sixty-three species of contaminant and invader plants were found on the six study sites during 1991-93. Nineteen (table 2) of the sixty-three species occurred at two or more sites in two or more years.

Herbaceous Cover - 1992 Sites

Seeded perennial cover (grasses and legumes) increased significantly ($p<0.05$) during 1992-93 in low- and high-competition treatments at Brockport and Kylertown II (fig. 2). However, the increase in seeded perennial cover at Clearfield was not as pronounced as at the other two sites. The smaller increase in seeded perennial cover at Clearfield was due to birdsfoot trefoil not increasing as it did at the other sites (Washburn, unpubl. data). As expected, perennial cover in the unseeded components of the low-competition treatments was significantly lower ($p<0.05$) than in the high-competition and in the seeded components of the low-competition treatments (fig. 2).

Seeded annual cover (grains and forbs) was present in 1992 and consisted of oats or oats and buckwheat. These seeded annuals were absent on all three sites in 1993.

The unseeded component of the low-competition treatments at Brockport and Kylertown II had significant contaminant and invader plant cover in the first growing season (1992), but the amount was markedly reduced in the second growing season (1993). The unseeded component of the low-competition treatment at Clearfield exhibited low plant cover by contaminant and invader species in 1992. This lower amount of contaminant and invader cover were probably due to the lack of available seed sources and because site preparations occurred after germination of invasive annuals.

Invasive plants were less prevalent on the sites reclaimed in 1992 than on the sites reclaimed in 1991. The favorable growing conditions in 1992 resulted in high amounts of seeded plant cover, creating more competition and less opportunity for contaminant and invader species to become established.

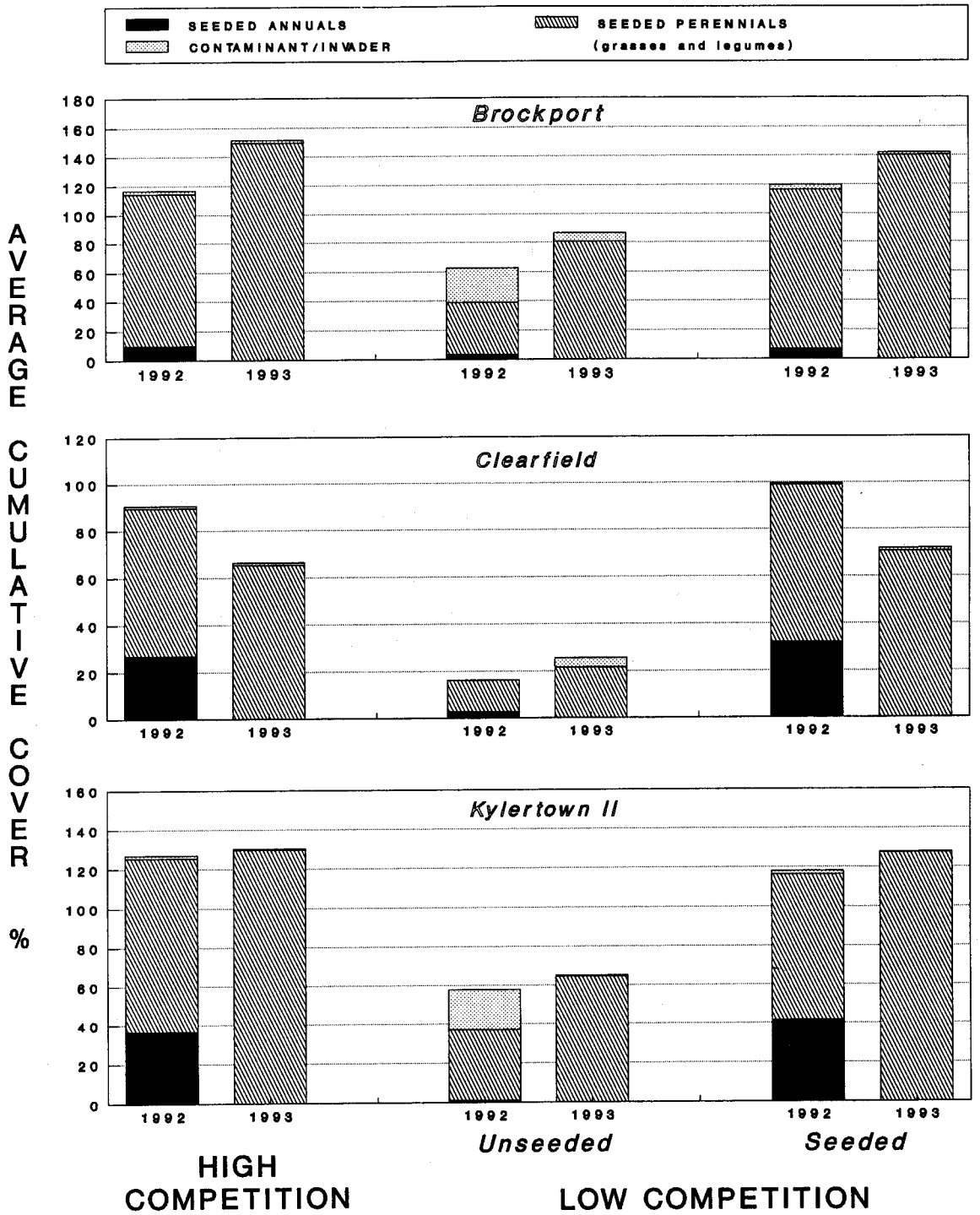


Figure 2. Average cumulative percent cover of seeded annuals, seeded perennials, and contaminant and invader species under two seeding treatments at the Brockport, Clearfield, and Kylertown II sites from 1992 through 1993.

Table 2. Names of 19 contaminant and invader species present in at least 2 years on 2 or more study sites in north-central Pennsylvania.

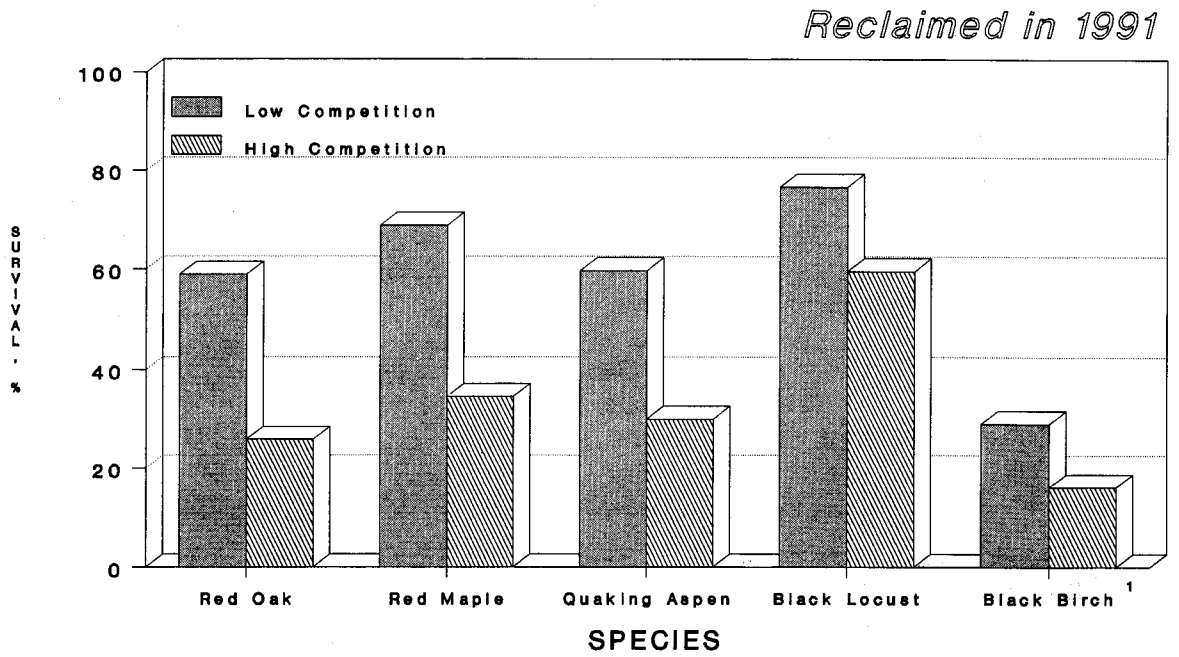
Common Name	Botanical Name
K-31 Tall Fescue	*Festuca arundinacea* Schreb.
Orchardgrass	*Dactylis glomerata* L.
Fall Panicum	*Panicum dichotomiflorum* Michx.
Common Barnyardgrass	*Echinochloa crusgalli* Beauv.
Deertongue Panicum	*Panicum clandestinum*
Yellow Sweetclover	*Melilotus officinalis* Lam.
White Sweetclover	*Melilotus alba* Medik.
Red Clover	*Trifolium pratense* L.
Goldenrod spp.	*Solidago* spp. L.
Curly Dock	*Rumex cripus* L.
Pink Knotweed	*Polygonum pensylvanicum* L.
Common Ragweed	*Ambrosia artemisiifolia* L.
Plantain spp.	*Plantago* spp. L.
Dwarf Cinquefoil	*Potentilla canadensis* L.
Oxeye Daisy	*Chrysanthemum leucanthemum* L.
Common Yellow Sorrel	*Oxalis stricta* L.
Staghorn Sumac	*Rhus typhina* L.
Blackberry	*Rubus* spp. L.
Pin Cherry	*Serotina pensylvanica* L.

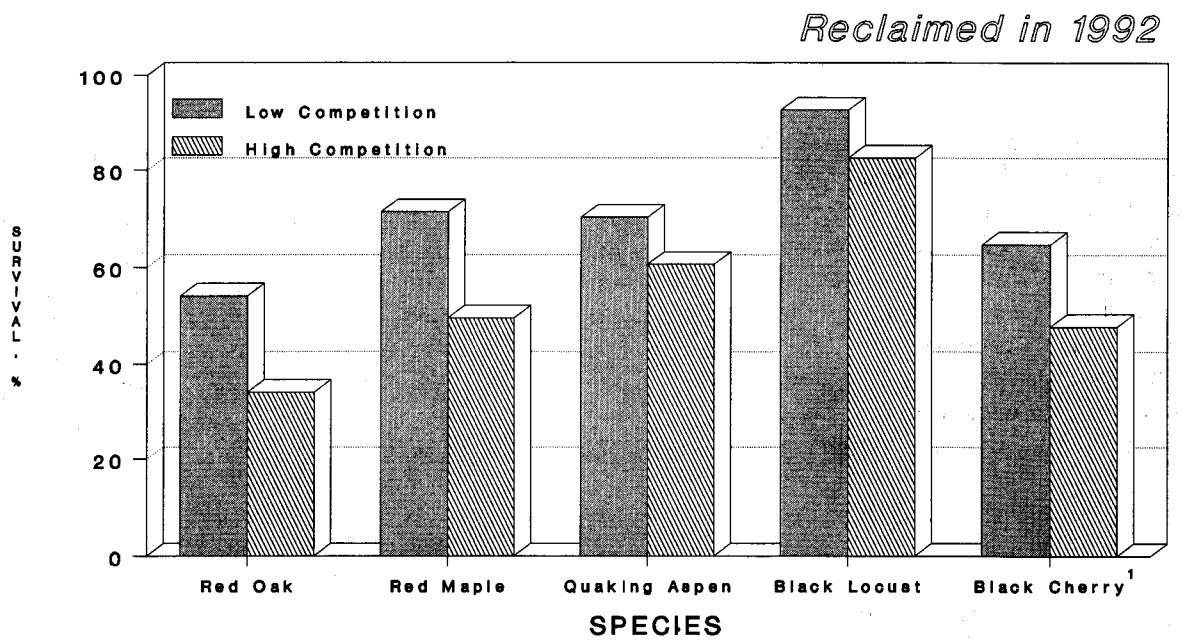
The number of species of contaminant and invader plants at the Brockport and Kylertown II sites paralleled the results found with the 1991 sites -- that more species were present in the unseeded low-competition components (table 3). Although the Clearfield site had fewer contaminant and invader species in the unseeded component of the low-competition treatment compared to the other two seeding treatments in 1992, the number of species were similar in 1993.

The numbers of species of contaminants and invaders present on sites reclaimed in 1991 increased or remained the same in the high- and low-competition treatments during 1992-93. However, no distinct pattern exists for changes in the numbers of contaminant and invader species present on sites reclaimed in 1992 in the high- and low-competition treatments during 1992-93.

Seedling Survival

Survival of all six species of hardwoods was higher in low-competition treatments compared with high-competition treatments at sites established in 1991 and 1992 (fig. 3). Of the four species that were planted in both 1991 and 1992, black locust exhibited the highest survival whereas red oak exhibited the lowest survival after 2 and 3 growing seasons.





[1] Black birch and black cherry were planted in only one year.

Figure 3. Average survival of six species of native hardwoods planted in low and high levels of herbaceous cover at reclaimed site in north-central PA; survival was determined for seedlings inside deer exclosures after three growing seasons for three sites established in 1991 and after two growing seasons for three sites established in 1992.

Table 3. Number of contaminant and invader species present in high- and low-competition treatments at three sites reclaimed in 1992.

	Brockport			Clearfield			Kylertown II		
	High	Low		High	Low		High	Low	
Year		Unseeded[1]	Seeded[2]		Unseeded[1]	Seeded[2]		Unseeded[1]	Seeded[2]
1992	12	18	11	6	1	5	6	14	5
1993	5	19	5	7	7	4	1	5	3

[1] Unseeded refers to the unseeded low-competition treatments.
[2] Seeded refers to the seeded low-competition treatments.

Conclusions

As expected, the amount of plant cover in the unseeded components of the low-competition treatments, especially the seeded perennial species, was reduced on all six sites during 1991-93. Reduced amounts of plant cover provided space where competition was minimal for invaders, as exemplified by the presence of more species of contaminants and invaders in the unseeded components of the low-competition treatments. The amount of contaminant and invader cover and the number of species of contaminants and invaders increased from 1991 to 1993 on sites reclaimed in 1991. However, no clear pattern existed for the amount of contaminant and invader cover or the number of species of contaminants and invaders on sites reclaimed in 1992. Survival of all six species of hardwoods was higher in low-competition treatments compared to survival in high-competition treatments after two and three growing seasons. The results of the present study indicated that black cherry, black locust, quaking aspen, and red maple may be used in reclamation of mined lands with native species.

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GROUND COVER AND TREE GROWTH ON CALCAREOUS MINESOILS: GREATER INFLUENCE OF SOIL SURFACE THAN NITROGEN RATE OR SEED MIX[1]

David A. Kost and John P. Vimmerstedt[2]

Abstract: Growth of ground cover and trees was evaluated for five growing seasons on calcareous coal minesoil surfaces (standard graded topsoil, graded and ripped topsoil, graded gray cast overburden) in southeastern Ohio. Soil surface plots were seeded in September 1987 with either a standard herbaceous seed mix [orchardgrass (Dactylis glomerata L.), timothy (Phleum pratense L.), perennial ryegrass (Lolium perenne L.), Kentucky bluegrass (Poa pratensis L.), Ranger alfalfa (Medicago sativa L.), Mammoth red clover (Trifolium pratense L.), Empire birdsfoot trefoil (Lotus corniculatus L.), and wheat (Triticum aestivum L.)], or a modified mix using no alfalfa and half the rate of orchardgrass. Nitrogen (45, 90, or 135 kg ha/N) was applied as ammonium nitrate in September 1987 and April 1989. White ash (Fraxinus americana L.), silver maple (Acer saccharinum L.), northern red oak (Quercus rubra L.), and eastern white pine (Pinus strobus L.) were planted in spring 1989 into 0.8 m-wide strips sprayed with glyphosate herbicide at 2.24 kg/ha in October 1988. Sethoxydim was applied at 0.54 kg/ha in June 1989 to control dense grass cover. Total herbaceous canopy cover and total aboveground live biomass were greater on standard topsoil than on gray cast overburden in 1989 and 1993, but the reverse was true in 1992. Legumes (canopy cover, aboveground live biomass, or fraction of total biomass) were generally more abundant on cast overburden than on the two topsoil surfaces until 1993. Total cover (84%) and total biomass (239 g/0.5 m^2) were highest in July 1989, following the last application of nitrogen fertilizer in April 1989. Total cover ranged from 44% to 56%, and total biomass ranged from 102 to 162 g/0.5 m^2 from 1990 to 1993. Total cover and total biomass were lower at the lowest nitrogen rate in 1989 only. Type of herbaceous seed mix did not affect growth of ground cover or trees. Overall tree survival was 82.0% the first year but declined to 40.6% after 5 yr. Survival varied significantly among all tree species (3.5% for pine, 22.2% for oak, 38.5% for maple, 98.1% for ash). White ash was taller on ripped topsoil (111 cm) than on cast overburden (94 cm). Silver maple survived better on standard topsoil (51.5%) and ripped topsoil (42.9%) than on cast overburden (21.0%). Red oak also survived better on both topsoil surfaces (33.4% or 23.9%) than on cast overburden (9.4%). Maple (43.8% versus 32.7%) and oak (33.6% versus 20.5%) survived better at the intermediate than at the low nitrogen rate. Survival was not reduced by the high nitrogen rate. The use of herbicides to control ground cover around the trees combined with above average rainfall during the one year in which nitrogen fertilizer increased ground cover probably explains the absence of detrimental effects on tree survival at the higher nitrogen rates.

Additional key words: standard graded topsoil, ripped topsoil, graded cast overburden, legume cover, legume biomass.

Introduction

Herbaceous ground cover competes with tree seedlings for water, nutrients, and light, and may also produce allelopathic chemicals that inhibit tree seedling growth. For these reasons, tree seedling survival and growth may be expected to increase with decreasing density and competitiveness of ground cover. Because nitrogen availability limits growth of ground cover on new minesoils, reducing the rate of nitrogen fertilizer may decrease ground cover density and increase tree survival and growth. Herbaceous seed mixes used in

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[2]David A. Kost, Senior Research Associate, and John P. Vimmerstedt, Associate Professor, School of Natural Resources, The Ohio State University and Ohio Agricultural Research and Development Center, Wooster, OH, USA.

reclamation often contain a variety of grass and legume species with differing abilities to compete with trees. We believed that Ranger alfalfa and orchardgrass were the more competitive species in the standard seed mix on the Muskingum Mine and that a less competitive mix could be formulated by omitting alfalfa and decreasing the rate of orchardgrass.

This paper presents 5-yr results of the effects of soil surface, ground cover seed mix, and nitrogen fertilizer rate on yield of ground cover and trees. The study was part of a project testing various cultural practices to enhance tree survival and growth on calcareous minesoils.

Methods

The study used a split-split-split plot design testing three soil surfaces (standard topsoil, ripped topsoil, gray cast overburden), two herbaceous seed mixes (standard mix or modified mix), three nitrogen fertilizer rates (45, 90, or 135 kg/ha N applied on two occasions), and four tree species, with six replications. The study area is on Central Ohio Coal Co.'s Muskingum Mine, 5 km south of Cumberland in Noble County, OH. 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. Reconstructed soil surfaces consisted of graded gray cast overburden without topsoil, graded cast overburden with 30 cm of replaced topsoil (standard topsoil), and a similarly constructed topsoil surface that was ripped to 30-cm depth after topsoil grading and then disked (ripped topsoil). Four replications were on a gently sloping (2% to 10% slope) area facing northeast, and two were in a steep valley (29% slope) facing southwest. Main soil surface plots measured 21.9 by 40.2 m in four replications and 14.6 by 40.2 m in two replications.

Topsoil has 77% soil-sized (<2 mm) particles, clay loam texture, pH of 6.9, and 27.4 kg/ha Bray 1-extractable P. Ammonium acetate-extractable K was 245 kg/ha, Ca was 5,967 kg/ha, and Mg was 652 kg/ha. Gray cast overburden has 50% soil-sized particles, loam texture, pH of 7.3, and 7.3 kg/ha Bray 1-extractable P. Ammonium acetate-extractable K was 407 kg/ha, Ca was 12,443 kg/ha, and Mg was 1,558 kg/ha. Electrical conductivity (1:2 soil-water mix) of fresh gray cast overburden was 4.08 dS/m, indicating high soluble salts. Conductivity decreased to 1.15 dS/m after 5 yr of weathering.

Herbaceous ground covers were seeded in September 1987. The standard seed mix contained the following species and seeding rates (kg/ha): orchardgrass (13.4), timothy (11.2), perennial ryegrass (9.0), Kentucky bluegrass (5.6), Ranger alfalfa (5.6), Mammoth red clover (5.6), Empire birdsfoot trefoil (5.6), and wheat (1.3 hL/ha). The modified seed mix differed by containing no alfalfa and a lower rate (6.7 kg/ha) of orchardgrass. Plots were mulched with hay after seeding. Seed mix subplots measured 21.9 by 20.1 m in four replications and 14.6 by 20.1 m in two replications.

Nitrogen fertilizer (45, 90, or 135 kg/ha N) was hand broadcast using ammonium nitrate in September 1987 and April 1989. Fertilizer sub-subplots measured 7.3 by 20.1 m in four replications and 4.9 by 20.1 m in two replications. To promote uniform fertilizer distribution, each sub-subplot was divided into 2.4 by 20.1 m strips for fertilizer application. All plots had received 448 kg/ha of 8-32-16 (N-P-K) fertilizer in August 1987.

Each fertilizer sub-subplot was planted with two tree rows with half of each row assigned to a different tree species. Tree spacing was 2.4 m between rows in a sub-subplot and 1.2 m within rows. Each tree species sub-sub-subplot contained seven or eight trees and was the experimental unit for survival and height measurements. About 840 trees each of white ash, silver maple, northern red oak, and white pine were hand-planted in March-April 1989. Hardwood stock was 1 yr old and obtained from the Zanesville, OH state nursery. White pine stock was 2 yr old from the Vallonia, IN state nursery. Each species was graded and pruned to a relatively uniform stem caliper and height before planting.

In October 1988 glyphosate herbicide was applied in 0.8 m-wide strips at 2.24 kg/ha to kill vegetation

in the tree rows before tree planting in spring 1989. Sethoxydim was applied at 0.54 kg/ha in June 1989 to control dense grass cover resulting from the nitrogen fertilization.

Tree survival was measured in June and October 1989 and August 1990. Tree survival and total height (nearest cm) were measured in July 1991 and August 1993.

Herbaceous percent cover and aboveground dry biomass were measured in 0.5 m^2 quadrats in June or July from 1989 through 1993. On each sampling date one sample was located systematically in each fertilizer treatment sub-subplot (total of 108 samples per date). Samples were taken between the tree rows in areas that had not been sprayed with herbicides. Previously sampled locations were avoided during successive years of sampling. Total canopy cover (projected onto the ground), mulch cover, rock cover, and bare soil were estimated as percentages to sum to 100 for each sample. Beginning in 1990, percent canopy cover due to legumes only was also estimated. In each sample, the presence or absence of alfalfa was recorded, and other legumes, grasses, or weedy plants were noted. Vegetation was clipped to 1 cm height, separated into nonlegume and legume components beginning in 1991, dried at 60 °C for at least 72 h, and weighed.

Data were subjected to analysis of variance using the ANOVA or GLM procedures of Statistical Analysis Systems software (SAS Institute Inc. 1987). Survival and canopy cover percentages were transformed with the arcsine-square root function before analysis. When analysis of variance showed significant ($p=0.05$) effects, differences among means were tested at $p=0.05$ using Duncan's multiple range test.

Results

Herbaceous Cover and Aboveground Dry Biomass

There were no long-term relationships relating total canopy cover and total aboveground biomass to soil surface treatments. Total cover and total biomass (between herbicide-treated tree rows) were higher on standard topsoil than on gray cast overburden in 1989, were lower on standard topsoil than on the other two soil surfaces in 1992, and were higher on standard topsoil than on cast overburden in 1993 (tables 1 and 2).

The sum of total canopy cover plus dead mulch not under the canopy provides an indication of protection from soil erosion. In 1993 total cover plus mulch averaged 89% on cast overburden (range 45% to 100%), 94% on ripped topsoil (range 62% to 100%), and 96% on standard topsoil (range 75% to 100%).

The various measures of legume productivity (legume canopy cover, legume aboveground biomass, legume fraction of total biomass) were usually greater on cast overburden than on the two topsoil surfaces until 1993 (tables 1-3). Legumes increased on all soil surfaces from 1991 to 1992. There were additional increases on the two topsoils but not on cast overburden in 1993, with the result that legumes were no longer more abundant on cast overburden. Over the 5-yr period, birdsfoot trefoil and red clover were generally the most abundant legumes on the topsoil surfaces, and volunteer yellow sweetclover (Melilotus officinalis Lam.) was dominant on gray cast overburden.

Total canopy cover and total aboveground biomass were highest in July 1989 following the last application of nitrogen fertilizer in April 1989. Nitrogen fertilizer rate affected total cover and total biomass only in 1989, when cover and biomass were lower at the lowest nitrogen rate (tables 1-2). The response of total cover and total biomass to nitrogen fertilization was, therefore, very short-lived, possibly due to loss of excess nitrogen by leaching and/or volatilization.

Legume canopy cover was greater at the 45- than at the 135-kg/ha nitrogen rate in 1990 (table 1). Legume fraction of total biomass was greater for the lowest nitrogen rate in 1991 (table 3). These effects did not persist but are consistent with the idea that high rates of nitrogen fertilization favor nonlegumes over

Table 1. Total canopy cover and legume canopy cover as influenced by soil surface, seed mix, and nitrogen fertilization treatments.[1]

	Total live canopy cover, %					Legume live canopy cover, %			
	July 1989	July 1990	June 1991	July 1992	July 1993	July 1990	June 1991	July 1992	July 1993
Soil Surface:									
Standard topsoil.....	92 a	44	44	41 b	62	2 b	2 b	14 b	38
Ripped topsoil.....	84 ab	50	45	51 a	57	9 ab	4 ab	22 b	32
Gray cast overburden....	76 b	49	43	51 a	50	19 a	10 a	35 a	29
Seed mix:									
Standard.....	86	46	43	49	56	8	5	25	32
Modified.....	83	50	45	46	56	12	6	22	34
Nitrogen Rate:[2]									
45 kg/ha N.....	73 b	48	45	50	55	15 a	7	27	29
90 kg/ha N.....	88 a	50	44	48	59	9 ab	5	25	37
135 kg/ha N.....	92 a	46	43	46	55	6 b	4	20	33

[1]Within a column and factor, means followed by no letters or similar letters are not significantly different at $p = 0.05$ using Duncan's multiple range test for soil surface and nitrogen rate, and LSD test for seed mix.

[2]Applied in September 1987 and April 1989.

Table 2. Total aboveground live biomass and legume aboveground live biomass as influenced by soil surface, seed mix, and nitrogen fertilization treatments.[1]

	Total aboveground live biomass, g/0.5m^2					Legume live biomass, g/0.5m^2		
	July 1989	July 1990	June 1991	July 1992	July 1993	June 1991	July 1992	July 1993
Soil surface								
Standard topsoil.....	277.8 a	92.4	107.5	119.8 b	188.9 a	3.9 b	14.2 b	84.8
Ripped topsoil.....	240.6 ab	108.5	101.5	152.7 a	156.8 ab	3.8 b	36.9 b	62.0
Gray cast overburden....	199.1 b	118.0	95.3	149.4 a	139.7 b	18.2 a	79.9 a	55.9
Seed mix:								
Standard.....	239.9	105.7	98.5	133.8	158.5	9.9	37.2	64.7
Modified.....	238.3	106.9	104.4	147.5	165.1	7.4	50.1	70.5
Nitrogen rate:[2]								
45 kg/ha N.....	177.0 b	112.0	102.7	152.7	162.2	13.0	55.4	64.6
90 kg/ha N.....	260.5 a	105.3	100.8	131.6	162.8	7.5	37.0	70.1
135 kg/ha N.....	280.0 a	101.6	100.8	137.5	160.3	5.4	38.4	68.0

[1]Within a column and factor, means followed by no letters or similar letters are not significantly different at $p = 0.05$ using Duncan's multiple range test for soil surface and nitrogen rate, and LSD test for seed mix.

[2]Applied in September 1987 and April 1989.

Table 3. Legume aboveground biomass as a percentage of total aboveground biomass as influenced by soil surface, seed mix, and nitrogen fertilization treatments.[1]

	June 1991	July 1992	July 1993
Soil surface			
Standard topsoil	3.0 b	10.1 b	39.2
Ripped topsoil	3.8 b	19.0 b	33.4
Gray cast overburden	16.5 a	44.8 a	37.4
Seed mix:			
Standard	8.7	23.3	35.4
Modified	6.9	26.0	37.9
Nitrogen rate:[2]			
45 kg/ha N	13.0 a	28.1	34.7
90 kg/ha N	6.5 b	24.3	37.9
135 kg/ha N	3.9 b	21.5	37.4

[1]Within a column and factor, means followed by no letters or similar letters are not significantly different at p = 0.05 using Duncan's multiple range test for soil surface and nitrogen rate, and LSD test for seed mix.

[2]Applied in September 1987 and April 1989.

legumes.

Type of herbaceous seed mix did not affect total canopy cover, legume canopy cover, total aboveground biomass, legume aboveground biomass, or legume fraction of total biomass on any sampling date (tables 1-3). From 1989 through 1993 alfalfa was present in 15.9% (43 of 270) of samples on subplots seeded with the standard mix and in 4.4% (12 of 270) of samples on subplots seeded with the modified mix. Omitting alfalfa from the modified seed mix, therefore, reduced but did not eliminate the presence of alfalfa in the ground cover. For the standard seed mix, alfalfa was present in 4.4% (4 of 90) of samples on standard topsoil, 5.6% (5 of 90) of samples on ripped topsoil, and 37.8% (34 of 90) of samples on cast overburden. For the modified seed mix, alfalfa was present in 1.1% (1 of 90) of samples on standard topsoil, 0.0% (0 of 90) samples on ripped topsoil, and 12.2% (11 of 90) samples on cast overburden. Sparse occurrence of alfalfa on standard or ripped topsoil indicates that alfalfa is not likely to be an important competitor with trees on these soils. From 1991 through 1993 orchardgrass was found in 54.3% (88 of 162) of samples on standard mix subplots and in 48.8% (79 of 162) of samples on modified mix subplots. Reducing the rate of orchardgrass in the modified seed mix thus had little effect on the presence of orchardgrass, possibly because orchardgrass seed was present in the hay used for mulching the plots.

Tree Survival and Growth

Overall tree survival was 82.0% the first year (1989), decreased to 48.8% following a drought in the third year (1991), and was 40.6% after 5 yr (1993). Type of seed mix did not affect survival or height of any tree

species. Nitrogen rate affected survival of silver maple and red oak but not height of any species.

White ash survived well (greater than 97%) on all treatments during the study. White ash was 29% taller on standard topsoil (79.1 cm) and ripped topsoil (79.8 cm) than on cast overburden (61.4 cm) in 1991. From 1991 to 1993 height increased 23.1 cm on standard topsoil, 31.6 cm on ripped topsoil, and 33.0 cm on cast overburden. By 1993 only trees on ripped topsoil (111.4 cm) were taller (18%) than those on cast overburden (94.4 cm), and trees on standard topsoil (102.2 cm) were intermediate. For the seed mix or nitrogen rate treatments, white ash height ranged from 72.4 cm to 74.8 cm in 1991 and from 101.2 cm to 104.2 cm in 1993.

Silver maple survived well on all treatments for the first two growing seasons and then declined sharply during the 1991 drought (table 4). After five growing seasons survival was greater (1) on both topsoil surfaces than on cast overburden and (2) at the intermediate than at the low nitrogen rate. Silver maple was taller on the two topsoil surfaces than on cast overburden in 1991 and 1993 (table 4). There was no increase in height from 1991 to 1993 owing primarily to deer browsing and rabbit cutting.

Red oak had lower first-year survival (table 5) than white ash or silver maple (table 4). Survival decreased substantially the second year and further the third year (table 5). After five growing seasons survival was low but was greater on both topsoil surfaces than on cast overburden. Beginning in 1990, red oak survived better at the intermediate (90 kg/ha N) than the low (45 kg/ha N) nitrogen rate. Red oak height was not affected by soil surface (table 5). Although we did not measure height at planting, we pruned the trees before planting to an initial height around 15 cm. The small height increases by 1993 reflect poor growth and

Table 4. Silver maple survival and height as influenced by soil surface, seed mix, and nitrogen fertilization treatments.[1]

	Survival, %				Total height, cm	
	1989 (yr 1)	1990 (yr 2)	1991 (yr 3)	1993 (yr 5)	1991 (yr 3)	1993 (yr 5)
Soil surface:						
Standard topsoil	97.6	97.6	73.1 a	51.5 a	36.6 a	35.0 a
Ripped topsoil	97.6	98.3	57.8 ab	42.9 a	41.3 a	36.3 a
Gray cast overburden	99.3	99.3	45.3 b	21.0 b	28.5 b	27.6 b
Seed mix:						
Standard	98.1	98.6	59.5	39.1	35.9	33.3
Modified	98.1	98.1	57.9	37.8	35.0	33.8
Nitrogen rate:[2]						
45 kg/ha N	99.0	99.3	46.5 b	32.7 b	34.4 ab	32.8
90 kg/ha N	97.2	97.9	69.6 a	43.8 a	38.4 a	35.5
135 kg/ha N	98.2	97.9	60.1 a	38.8 ab	33.3 b	32.1

[1]Within a column and factor, means followed by no letters or similar letters are not significantly different at p = 0.05 using Duncan's multiple-range test.

[2]Applied in September 1987 and April 1989.

Table 5. Red oak survival and height as influenced by soil surface, seed mix, and nitrogen fertilization treatments.[1]

	Survival, %				Total height, cm	
	1989 (yr 1)	1990 (yr 2)	1991 (yr 3)	1993 (yr 5)	1991 (yr 3)	1993 (yr 5)
Soil surface:						
Standard topsoil	79.0	55.1 a	40.8 a	33.4 a	15.4	20.4
Ripped topsoil	74.4	43.3 ab	29.9 a	23.9 a	18.4	19.4
Gray cast overburden	70.1	28.9 b	11.9 b	9.4 b	NA	NA
Seed mix:						
Standard	76.6	47.3	29.6	23.9	[3]16.6	[3]22.3
Modified	72.4	37.6	25.5	20.6	18.8	20.5
Nitrogen rate:[2]						
45 kg/ha N	75.4	37.2 b	20.5 b	15.6 b	[3]15.6	[3]19.7
90 kg/ha N	73.8	48.8 a	33.6 a	27.2 a	17.2	21.6
135 kg/ha N	74.3	41.3 ab	28.4 ab	23.8 ab	18.3	18.3

NA indicates not available because of low survival.

[1]Within a column and factor, means followed by no letters or similar letters are not significantly different at p = 0.05 using Duncan's multiple range test.

[2]Applied in September 1987 and April 1989.

[3]Means for standard topsoil and ripped topsoil.

extensive damage by rabbits.

White pine had lower first-year survival (56.0%) than any other species. Survival declined moderately in 1990 (40.0 % survival) and sharply during the 1991 drought (10.5% survival). White pine survived better on standard topsoil than on cast overburden in 1989 (69.1% on standard topsoil versus 39.6% on cast overburden) and 1990 (56.0% on standard topsoil versus 24.8% on cast overburden). After 5 yr it had failed (less than 10% survival) on all soil surfaces.

For the first 2 yr white pine survival responded strongly to slope and drainage conditions. In 1989 survival averaged 45.3% on the four gently sloping replications versus 77.3% on the two steeply sloping replications. Survival on cast overburden alone was almost four times less on the gently sloping replications (20.2%) than on the steeply sloping replications (78.6%). In 1990 survival averaged 28.8% on the four gently sloping replications versus 62.2% on the two steeply sloping replications. Survival on cast overburden alone was 6.2% on the level area versus 62.2% on the steep area. Rainfall (for the April-August growing season) was 77% above average in 1989 and 41% above average in 1990. These results indicate that excessive soil moisture or compaction, particularly on cast overburden, affected white pine survival on the level site.

Discussion and Conclusions

Effects of the experimental treatments on herbaceous ground cover development can be considered from two viewpoints. Mining regulations require a ground cover density that will control erosion, and any treatment that failed to produce the required density would not be acceptable. In contrast, treatments that result in excessively dense ground covers should be avoided if tree establishment is the goal of reclamation.

Our 5-yr results indicate that topsoil replacement is not necessary for ground cover establishment on the Muskingum Mine. There was year-to-year variation in the relative productivity of ground cover on standard topsoil and cast overburden, but ground cover on cast overburden was adequate for erosion control.

The relatively large fraction of legumes in the ground cover after 5 yr has important implications for both persistence of the ground cover and tree survival and growth. Legumes are an important source of the nitrogen capital needed for continued growth of the ground cover in a developing ecosystem (Bradshaw 1983), and some of the N fixed by legumes is available to trees in the short term. On the negative side, some legumes have allelopathic effects on tree growth. Larson and Patel (1992) reported that adding birdsfoot trefoil litter to the growing medium strongly inhibited growth of white ash seedlings in the greenhouse. Birdsfoot trefoil was a dominant species on the topsoils in this study.

Modifying the herbaceous seed mix or varying the rate of nitrogen fertilizer did not strongly affect ground cover competitiveness with trees. The differences in composition of the two seed mixes were not great enough to produce important differences in ground cover. Alfalfa's presence was reduced by using the modified mix, but this would be important only on cast overburden because alfalfa was rare on the topsoils regardless of seed mix. Further research on alternative seed mixes should use mixes that have major differences in species composition. Eliminating rather than reducing the rate of orchardgrass would be a good starting point in formulating an alternative mix. Torbert and Burger (1990) reported success in developing tree compatible ground covers in Virginia. In other studies on the Muskingum Mine we have had poor tree survival associated with dense stands of volunteer tall fescue on topsoil and volunteer sweetclover or seeded legumes on cast overburden. In this study nitrogen fertilizer affected ground cover only during the year in which fertilizer was applied.

The responses of tree survival and height to the various experimental treatments varied with tree species. Silver maple and red oak survived better on the two topsoil surfaces than on cast overburden. The same was true for white ash and silver maple height. These results indicate that this topsoil is better than this cast overburden as a tree growth medium. The pH and soluble salt levels in cast overburden are higher than most trees prefer.

Increased silver maple and red oak survival at the intermediate nitrogen rate may be related to the extensive animal damage suffered by these species. At the intermediate N rate trees may have developed larger root systems that facilitated sprouting in response to continuing rabbit cutting. Lack of a growth response to N by white ash is puzzling. This species has a high N requirement (Mitchell and Chandler 1939) and so would be expected to respond to increasing N. Two factors reduced any harmful effects of increased nitrogen supply on ground cover competitiveness with trees. We used herbicides to control ground cover in strips around the trees, and therefore our results probably do not apply to conditions where herbicides are not used. Because we had above average rainfall during the year in which ground cover varied with nitrogen rate, competition between ground cover and trees for soil moisture probably was not important.

Although early white pine survival was affected more by slope and drainage conditions than by the experimental treatments, continued decline on both the gently and the steeply sloping sites indicates that white pine is not adapted to these soils. Because of poor height growth and continuing mortality associated with animal damage, it is questionable how many of the surviving silver maple and red oak will persist. These two

species would probably benefit from tree shelters. White ash is the only tree species that is definitely established after 5 yr.

In summary, trees responded more to soil surface type than to nitrogen rate or seed mix. Whether topsoil replacement is economically justified will depend on continuation of the greater tree productivity on topsoil. Information from other studies in the project that includes this study will help to answer this question.

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THE POTENTIAL OF IN SITU AMENDMENTS TO IMPROVE ALDER ESTABLISHMENT IN POTS OF METALLIFEROUS MINE WASTE[1]

Pete H. Abrutat[2], R. P. Edwards,[2] and C. Wilkins[3]

Abstract: Several inherent characteristics of metalliferous mine wastes inhibit the establishment of vegetation: nutrient deficiency, acidity, poor water retention, metal toxicity, instability, and poor soil structure. They may be overcome by in situ amendments. Most reclamation programs in west Cornwall, UK, involve extensive tree-planting schemes. Alder species are usually a major component of such schemes because of their N-fixing capability. A pot experiment (based on a quarter replicate of a full 2^7 factorial design) was established at the Camborne School of Mines, to investigate the effects of seven amendments (and their interactions) on the growth of common alder (*Alnus glutinosa* (L.) Gaertner). The amendments were ammonium nitrate, mineral phosphate and potash fertilizers, calcareous sea sand, dewatered sewage sludge, diatomaceous earth, and a superexpanding polyacrylamide gel polymer. The waste was analyzed for total and ammonium acetate-extractable metals and nutrients, pH, and total exchangeable bases at the outset of the experiment, and was found to contain 0.3% As and very low concentrations of the major nutrients. Tree heights and survival were recorded monthly for two growing seasons. Leaf, stem, and root dry weights were recorded after harvesting, from which various resource allocation ratios were calculated. Sewage was the best amendment, followed by diatomaceous earth. The N, P, and K fertilizers, sand, and gel had little effect, being deleterious in some cases. Of the interactions, the traditional mineral phosphate-calcareous sand treatment was ineffective. The importance of soil structure improvement by organic material was apparent.

Introduction

Cornwall contains approximately 3,200 ha of derelict mined land (DOE 1991). Most of this has resulted from centuries of intense metalliferous mining activity. When this land is reclaimed, trees are major constituents of most schemes. Once established, tree root systems are strong and very extensive, stabilizing the soil at depth. Soil organisms proliferate in the rhizosphere secreting soil-binding mucous substances, thus imparting the all-important "crumb texture" to soils. This improves aeration and drainage and reduces compaction (Cherfas, 1992).

An in situ amendment can be described as a material that is intimately mixed with the mine waste in order to improve the substrate for plant growth (Mitchell 1991a). The most common are lime, fertilizers, and organic wastes. The main problems with metalliferous mine wastes are their low nutrient concentrations, their poor soil structure and water retention, and the presence of heavy metals which any amendment would need to render unavailable to plants. This can be done in situ by either complexation or precipitation. The application of lime and/or phosphate fertilizers reduces the solubilities of some heavy metals and metalloids. Organic molecules in sewage sludges can also complex heavy metals. Mitchell and Atkinson (1991) have investigated the metal-complexing properties of ion-exchange minerals. Mitchell (1991b) has discussed their relevance to field reclamation practices. The possibility of using one of Mitchell's promising amendments and a superexpanding polyacrylamide gel, in combination with the more conventional amendments, has been investigated in this work.

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[2]Pete H. Abrutat, Postgraduate, and Richard P. Edwards, Senior Lecturer of Geology, Camborne School of Mines, University of Exeter, Pool, Redruth, Cornwall TR15 3SE, UK.

[3]Carolyn Wilkins, Geochemist, CSM Associates, Rosemanowes, Herniss, Penryn, Cornwall TR10 9DU, UK.

This experiment investigated the effects of seven amendments on the growth of common alder (*Alnus glutinosa* (L.) Gaertner) in metalliferous mine waste. Common alder was chosen owing to its importance in reclamation schemes. N, P, and K fertilizers provided essential plant macronutrients (P may also render some metals insoluble). Calcareous sea sand was used as a locally available lime source to increase the pH and therefore decrease the availability of some metals (as used in local reclamation schemes). Dewatered sewage sludge has many useful properties: it is a slow-release nutrient source; and its organic content retains water, improves the soil texture, and may complex metals (Byrom and Bradshaw 1991). Diatomaceous earth is believed to sequester metals as insoluble silicates (Mitchell and Atkinson 1991). Superexpanding gel polymers are used in horticulture primarily as water-retaining agents (Johnson 1984). They also sequester metals in solution (Mitchell 1991a). A factorial experiment was designed to investigate the potential of established and novel amendments, both individually and interactively with other amendments.

Methods

Collection and Preparation of Waste Samples

Approximately 4 t of mine waste was collected from United Downs, west Cornwall, UK (grid ref: SW7441). The site is composed of several abandoned copper mines that formed the highly productive United Mines group in the 19th Century. Waste from the top 30 cm of an area of the site was sieved through a 25 mm stainless steel mesh over a skip in order to extract the minus 25 mm material. The waste was mixed within the skip during collection and after emptying. Five 5 kg grab samples were taken at intervals and prepared for analysis.

The five samples were dried in an oven at 40°C to constant weight, and then sieved in stainless steel sieves to extract the minus 2 mm fraction. Approximately 200 g of this was riffled out using a stainless steel riffle, and about 20 g of this material was milled to a floury texture in a tungsten carbide tema mill.

Chemical Analysis

Weakly held metals were extracted using molar ammonium acetate adjusted to the pH of the waste (3.7) with glacial acetic acid. The procedure followed was that of Grimshaw (1989). The concentrations of Al, Ca, Cd, Cu, Fe, K, Mg, Mn, Na, Pb, and Zn were measured by atomic absorption spectrophotometry (AAS). The total exchangeable bases (TEB) were calculated by summation of the Ca, K, Mg, and Na concentrations in milliequivalents to derive a measure of the waste's effective cation exchange capacity (Grimshaw 1989). Waste pH was measured using the paste method of Hesse (1971).

The nitric-perchloric acid digestion method of Hesse (1971) and AAS were used to determine the "total" concentrations of each element, except As, which was measured by hydride generation. Total P was quantified colorimetrically using the molybdenum blue method of Grimshaw et al. (1989) with the additional sodium metabisulfite step of Olsen and Dean (1965) in order to reduce interference from arsenic acid and ferric iron.

Total N was extracted and measured using the semi-micro Kjeldahl digestion and distillation methods of Grimshaw et al. (1989). The Markham steam distillation apparatus was used (Markham 1942). Mineralizable N was measured using the method of Grimshaw et al. (1989). Total S was measured on a LECO HF10 Induction Furnace coupled with a sulfur titrater. Appropriate quality control standards were used throughout.

Preparation and Maintenance of the Experimental Treatments

The experiment was designed as a quarter replicate of a 2^7 factorial design with four replicates per treatment. Two-year-old (1+1) alder trees were planted in 10 L plastic pots at a rate of one tree per pot, in approximately 9 L of waste. One or more of the amendments was mixed with the waste according to the experimental design at the rates given in table 1. Addition rates were calculated according to volume (10 L) or weight (12 kg dry weight per pot), depending on the amendment. Fertilizer addition rates were obtained from

Williamson et al. (1982). Calcareous sea sand was added after measuring the pH and calculating the acid-neutralizing capacity (ANC) of the waste and its future acid generation potential (i.e., iron pyrite content) using the methods given in Williamson et al. (1982). The pH was 3.7, the ANC was 0.6%, and the pyrite content was 0.1%. It was calculated that 420 g of sand (at 40% calcium carbonate) per pot would be sufficient to neutralize the present and potential acidity. The application rates for diatomaceous earth and gel were more subjective owing to the novelty of this use. A 5% w/w rate has been used in work with zeolites as nutrient storage materials, and this was suggested as a suitable rate for diatomaceous earth (A. Moffat, Forestry Commission, UK, personal communication). The gel is usually applied at a rate of 0.1% w/v. An addition rate of 0.3% w/v was used after consultation with the manufacturers. The N, P, and K treatments received a maintenance top dressing of half the initial rate at the start of the second season (Williamson et al. 1982). The treatments were arranged in four randomized blocks and placed in an outside enclosure. Fungal infections were controlled with a systemic fungicide. During dry periods the trees were watered with tap water in the early morning or late afternoon.

Measuring and Harvesting Regimes

The experiment ran for almost two growing seasons. Monthly height measurements were taken to the nearest 0.5 cm during the growing season. Average heights and height increments from the start, from the four replicates of each treatment, were calculated. Tree deaths were taken into account during statistical analyses. The trees were harvested by removing the stems at soil level and grouping the replicates for analysis. This allowed the calculation of treatment means, which reduced the statistical effect of dead trees and the time required for further analysis. Roots were removed from the waste and washed. The root replicates were grouped as for the shoots. The trees were dried in a warm room at 33°C. When dry, the various parts were weighed. Biomass ratios were calculated from these results.

Statistical Analyses

Growth results were analyzed using the factorial design and analysis capability of the MINITAB release 8.2 computer package. As the factorial design was not orthogonal, adjusted means were automatically calculated for each factor and the first-order interactions. A factor or interaction has a positive or negative effect if the adjusted mean of all the treatments containing the factor is respectively higher or lower than the adjusted mean of the treatments without the factor. The factorial design chosen incorporated the treatment containing no amendments. Second and higher order interactions were omitted in order to release sufficient degrees of freedom for calculation of the *t* and *F* statistics. The effects values for these higher order interactions were noted and were found to be small and probably insignificant.

Table 1. Amendment application rates used in the experiment, and the equivalent field application rates.

		Application rate (dry weights)	
Amendment	Factor	g/pot	t/ha[1]
Ammonium nitrate.....	N	25	-
Mineral phosphate.....	P	100	-
Sulfate of potash.....	K	20	-
Calcareous sea sand.....	s	420	67
Superexpanding polymer.....	g	30	4.8
Dewatered sewage sludge.....	w	450	72
Diatomaceous earth.....	d	600	95

[1] Calculated as t/ha to 22 cm depth (equivalent to the depth of the pots).
"-" denotes that there are no equivalent field application rates for the fertilizers.

Results and Discussion

Mine Waste Properties

Mine waste was removed from an area of Ericaceous vegetation common over much of the United Downs site. It consists predominantly of waste rock that is very deficient in all the major plant nutrients and is moderately toxic (As 0.3%) with a pH of 3.7 (table 2). The TEB of 0.42 meq/100g is very low thereby exacerbating the leaching caused by the area's high annual rainfall.

Effects of Individual Amendments on Tree Performance

The three fertilizers showed no significant growth enhancing abilities and were deleterious in some cases (table 3). Ammonium nitrate fertilizer has an acidity equivalent to the neutralizing ability of 60 g $CaCO_3$ per 100 g fertilizer (Williamson et al. 1982). This is caused by oxidation of the NH_4^+ producing NO_3^- and H^+. It is known that the solubilities of many minerals increase at low pHs. The nutrient effect of N applied in this form may be irrelevant due to its high solubility, despite the maintenance top dressings. Also the natural N-fixing ability of alder could account for a large proportion of the plant's N requirements.

Mineral phosphate is a slow-release phosphate fertilizer that is widely used in reclamation schemes in west Cornwall. It was a major contributor to the poor survival of some treatments, although it had negligible effects on other properties (table 3). In highly acidic soils, phosphates may become fixed as insoluble iron and aluminium phosphates (Williamson et al. 1982). This effect would be compounded by the high Fe and Al concentrations of the waste (table 2). Phosphates also combine with Ca and Mg which would further reduce their availability and further depress the already low buffering capacity of the waste. There is evidence that excessive soil phosphate may displace arsenate ions into the soil solution, thus increasing the plant availability of As (O'Neil 1990).

Table 2. Selected chemical properties of the mine waste used in the experiment (mean, standard deviation (S.D.) and range. Note: *n*=5 unless otherwise stated).

	Total extractable[1], %			Ammonium acetate[2], µg/g		
Element	Mean	S.D.	Range	Mean	S.D.	Range
Al.....	5.6	0.27	5.30-5.80	277	14.5	264-298
As.....	0.30	0.01	0.30-0.32	-	-	-
Ca.....	0.04	0.02	0.02-0.05	61 (*n*=4)	43	0.00-96.0
Cu.....	0.05 (*n*=4)	0.002	0.04-0.05	21	6.1	15.0-30.0
Fe.....	8.4	0.09	8.30-8.50	1,980	236	1600-2200
K.....	0.36	0.08	0.26-0.48	1.3	1.5	0.00-3.80
Mg.....	0.66	0.02	0.64-0.68	6.7 (*n*=4)	1.5	5.60-8.60
N.....	0.06	0.01	0.04-0.07			
P.....	43.0*	30.0	15-95			
Other properties						
Mineralizable N (mg/100g).....				0.94 (*n*=10)	0.63	0.00-1.64
pH.....				3.70	0.05	3.70-3.80
TEB (meq/100g).....				0.42 (*n*=4)	0.23	0.08-0.60

ND = not detectable. TEB = total exchangeable bases. "-" = element not analyzed for.
[1] Total extractable = nitric-perchloric acid-extractable for all elements, except N and S (see text). Units expressed as percentages unless marked "*", where units are µg/g.
[2] Ammonium acetate-extractable metals expressed as µg/g unless otherwise stated.

Table 3. Effects and significances of the various factors and interactions on several plant growth properties[1].

Factor[2]	Survivorship	Growth property						Resource allocation				
		Height	Increment	Leaf DW	Stem DW	Root DW	Total DW	RSR	LWR	StWR	ShWR	RWR
N	- ***	- N.S.	- N.S.	- *	- N.S.	- *	- N.S.	- N.S.	- *	- N.S.	- N.S.	- N.S.
P	- ***	- N.S.	- N.S.	+ N.S.	+ N.S.	+ N.S.	+ N.S.	- N.S.	+ N.S.	- N.S.	- N.S.	- N.S.
K	- *	- N.S.	- N.S.	- *	- N.S.	- *	- *	- N.S.	- N.S.	+ N.S.	- N.S.	- N.S.
s	- ***	- *	- N.S.	+ N.S.	- **	- ***	- **	- **	+ ***	- N.S.	+ N.S	- *
g	- *	- N.S.	+ N.S.	- N.S.	+ N.S.	+ N.S.	+ N.S.	+ N.S.	+ N.S.	+ N.S.	+ N.S.	+ N.S.
w	+ *	+ *	+ ***	+ ***	+ ***	+ **	+ ***	- N.S.	+ N.S.	+ N.S.	+ N.S.	- N.S.
d	- N.S.	+ N.S.	+ N.S.	+ *	+ *	+ N.S.	+ *	- *	- N.S.	+ N.S.	+ N.S.	- N.S.
N*P	+ N.S.	+ N.S.	+ N.S.	+ N.S.	+ N.S.	+ *	+ N.S.	+ N.S.	- N.S.	- N.S.	- N.S.	+ N.S.
N*K	- N.S.	- N.S.	- N.S.	- N.S.	- N.S.	- N.S.	- N.S.	+ N.S.	- N.S.	- N.S.	- N.S.	- N.S.
N*s	- *	- N.S.	- N.S.	- N.S.	- N.S.	+ N.S.	- N.S.	+ N.S.	- N.S.	- N.S.	- N.S.	- N.S.
N*g	+ N.S.	+ N.S.	+ N.S.	+ *	+ N.S.	+ N.S.	+ N.S.	+ N.S.	+ N.S.	- N.S.	+ N.S.	+ N.S.
N*w	+ N.S.	+ N.S.	+ N.S.	+ N.S.	- N.S.	+ N.S.	+ N.S.	+ N.S.	+ N.S.	- N.S.	+ N.S.	+ N.S.
N*d	- *	- N.S.	- N.S.	- N.S.	- N.S.	- N.S.	- N.S.	- N.S.	+ N.S.	- N.S.	- N.S.	- N.S.
P*K	- N.S.	- *	- N.S.	- N.S.	- *	- *	- *	- N.S.	+ N.S.	- N.S.	- N.S.	- N.S.
P*s	- N.S.	- *	- **	- *	- **	- *	- *	- N.S.	- N.S	- N.S.	- N.S.	- N.S.
P*g	- N.S.	+ N.S.	+ N.S.	+ *	+ N.S.	+ N.S.	+ N.S.	- N.S.	+ N.S.	+ N.S.	+ N.S.	+ N.S.
P*w	+ ***	- N.S.	- N.S.	+ N.S.	- N.S.	- N.S.	- N.S.	+ N.S.	+ N.S.	+ N.S.	+ N.S.	+ N.S.
P*d	- N.S.	- N.S.	+ N.S.	- N.S.	- N.S.	+ N.S.	- N.S.	+ N.S.	- N.S.	- N.S	- N.S.	- N.S.
K*s	+ N.S.	- N.S.	+ N.S.	+ N.S.	+ N.S.	- N.S.	+ N.S.	- N.S.	+ N.S.	- N.S.	- N.S.	- *
K*g	+ N.S.	- N.S.	- N.S.	- N.S.	- N.S.	- N.S.	- N.S.	+ N.S.	+ N.S.	+ N.S.	+ N.S.	+ N.S.
K*w	+ *	+ N.S.	- N.S.	- N.S.	- N.S.	- N.S.	- N.S.	+ N.S.	+ N.S.	+ N.S.	+ N.S.	+ N.S.
K*d	+ N.S.	- N.S.	+ N.S.	- N.S.	- N.S.	- N.S.	- N.S.	- N.S.	- N.S.	- N.S.	- N.S.	- N.S.
s*g	- N.S.	- N.S.	- N.S.	- N.S.	- N.S.	- N.S.	- N.S.	+ N.S.	+ N.S.	+ N.S.	+ N.S.	+ N.S.
s*w	- N.S.	+ N.S.	+ N.S.	- N.S.	- N.S.	- N.S.	- N.S.	+ N.S.	- N.S.	+ N.S.	+ N.S.	+ N.S.
s*d	- N.S.	- N.S.	+ N.S.	+ N.S.	- N.S.	- N.S.	+ N.S.	- N.S.	- N.S.	- N.S.	- N.S.	- N.S.

DW = dry weight, RSR = root/shoot ratio, LWR = leaf weight ratio, StWR = stem WR, ShWR = shoot WR, RWR = root weight ratio.
[1]Effect: + = positive effect, - = negative effect. Significance: * = $P \leq 0.05$, ** = $P \leq 0.01$, *** = $P \leq 0.005$, N.S. = not significant.
[2]See table 1 for identification of factors.

Sulfate of potash negatively affected survival and biomass (table 3). Height and stem dry weights were not affected, which suggests that the trees were preferentially allocating resources to poorer quality stem material. This could have been aggravated by the shortage of some other nutrient, e.g. P. In soils with a low cation exchange capacity (TEB in this case), potassium is rapidly leached (Williamson et al. 1982)

Calcareous sea sand was the most deleterious amendment (table 3), despite its use as a liming agent in local reclamation schemes! Its poor performance may be explained by referring to the soil chemistry of As. Arsenic occurs in soils in various oxyanionic forms and shares a similar soil chemistry to P. There is evidence to suggest that the availability of As increases as the soil pH rises from pH 7 to 9 (O'Neil 1990).

The sand had the greatest effect on resource allocation of any amendment. The root weight ratio (RWR) was significantly decreased ($P \leq 0.05$), leading to a lower overall root-shoot ratio (RSR) ($P \leq 0.01$); at the same time there was a highly significant increase in leaf weight ratio (LWR) ($P \leq 0.005$). When taken together with the other results, it seems that the sand caused a higher allocation towards the aerial parts by increasing shoot biomass without increasing height (see height and increment measurements, table 3), perhaps as a response to a stressful environment. The stem weight ratio (StWR) was not significantly affected, although stem and root dry weights were, which suggests that the actual biomass per branch was relatively low because most resource allocation was going into leaf production.

Superexpanding gel is a potential soil structure improver. It can expand up to 400x its dry volume in pure water, releasing it gradually (Johnson 1984). There is also evidence that the gel sequesters metal ions from solution (Mitchell 1991a). The gel had a significant negative effect on survival ($P \leq 0.05$). It was observed accumulating at the surface of the waste where its ameliorating action is redundant.

The most promising in situ amendment identified in this study was dewatered sewage sludge. It was the only amendment to significantly improve survival ($P \leq 0.05$, table 3) and height increment ($P \leq 0.05$). It also significantly enhanced all aspects of the biomass (table 3). Dewatered sludge is produced by the compression of raw sewage to form a cake of approximately 30% solids by weight. Its main roles here were as a soil structure improver and a slow-release source of nutrients. Improvements in soil structure are caused by the presence of organic material, which expands when wet and contracts when dry thereby, reducing compaction (and therefore root damage), and improving air and water (and therefore nutrient) permeability. The organic fraction also acts as: a substrate for the growth of microorganisms, a water storage facility, and a complexing agent for heavy metals (Bradshaw and Chadwick 1980).

Diatomaceous earth is neither a major nutrient source nor a soil structure improver. However, it has a high ion removal capacity (Mitchell and Atkinson, 1991). Increasing the number of anionic exchange and/or removal sites in the waste should ameliorate the problem of competition between phosphate and arsenate ions for the otherwise scarce exchange sites. It significantly increased leaf and stem dry weights (table 3), thus having a significant effect on the RSR ($P \leq 0.05$, table 3). There was no effect on height or increment (table 3). This could be caused by an increase in the branching frequency, thereby increasing bushiness, but not necessarily affecting overall height growth.

Effects of Interactions

Of the first-order interactions, nitrogen-sand (N*s) and nitrogen-diatomaceous earth (N*d) had significant negative effects on survival ($P \leq 0.05$), whereas phosphorus-sewage (P*w) and potassium-sewage (K*w) had significant positive effects ($P \leq 0.005$ and $P \leq 0.05$ respectively, table 3). This suggests that there was a survival-promoting interaction between a nutrient amendment and a soil structure-enhancing amendment. The waste was highly deficient in both P and K (table 2), but both these factors had negative effects on growth when considered individually. The benefit of sewage in providing extra exchange sites, thus offset the toxicity effects of added P,

and provided a top-up source of P for the increased plant demand due to K fertilization. Neither the N*s nor the N*d interaction contained a structural component; consequently there was an associated negative effect.

The phosphorus-potassium (P*K) and phosphorus-sand (P*s) interactions are the most prominent in table 3. Both had significantly negative effects on several properties. Neither contains a structure-improving amendment. A significant positive P*K interaction would be expected if potash were acting to produce poorer quality stems due to the shortage of P (for example). This was not the case, perhaps because the beneficial effects of P were being offset by an increase in As availability. However, the K*w interaction was positively significant for survival, and the negative effects on leaf, root and total dry weights have been eradicated (table 3). This suggests that the lower levels of P in sewage were enough to promote growth, but not enough to initiate As toxicity. The potentially toxic effects of any excess As release could have been reduced by the increase in exchange sites on organic molecules in the sewage. Both P and sand acted individually to increase As availability. Together they produced a synergistic effect, further worsening the situation, i.e. sand increased the solubility of As, while P increased the competition for the limited number of exchange sites in the soil.

Conclusions and Future Work

Dewatered sewage sludge was the most promising amendment. Its main benefit was that it improved the soil structure. Heavy metal sequestration, nutrient supply, and water retention were secondary-but nevertheless important-as far as alder performance was concerned. Sewage is also cheap and widely available. Traditional local amendments such as mineral phosphate and calcareous sea sand may have no positive effects on alder growth at best, and at worst may actually reduce alder performance. Of the novel amendments, the gel had little effect, but the diatomaceous earth was promising. The major drawback with both of these potential amendments is cost.

Further work is required to refine the optimal application rates in terms of tree performance and economics. Chemical analysis of the treatments is presently underway. A fieldwork program is required to test the trends outlined in this paper. A similar experiment is presently underway with sycamore (*Acer pseudoplatanus* L.) to test the modifying effects (if any) of the alder's N-fixing capability on the action of the amendments.

Acknowledgements

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HYBRID PINE FOR TOUGH SITES[1]

Walter H. Davidson[2]

Abstract: A test planting of 30 first- and second-generation pitch x loblolly pine(pinus rigida x P. taeda) hybrids was established on a West Virginia minesoil in 1985. The site was considered orphaned because earlier attempts at revegetation were unsuccessful. The soil was acid (pH 4.6), lacking in nutrients, and compacted. Vegetation present at the time of planting consisted of a sparse cover of tall fescue (Festuca arundinacea) and poverty grass (Danthonia spicata) and a few sourwood (Oxydendrum arboreum) and mountain laurel (Kalmia latifolia) seedlings. In the planting trial, 30 different hybrids were set out in 4 tree linear plots replicated 5 times. The seedlings had been grown in containers for 1 yr before outplanting. Evaluations made after 6 growing seasons showed overall plantation survival was 93 %; six hybrids and one open-pollinated cross survived 100 %. Individual tree heights ranged from 50 to 425 cm with a plantation average of 235 cm (7.7 ft). Eleven of the hybrids had average heights that exceeded the plantation average. Another test planting of tree and shrub species on this site has very poor survival. Therefore, pitch x loblolly hybrid pine can be recommended for reclaiming this and similar sites.

Additional Key Words: pitch x loblolly pine, West Virginia, reclamation revegetation

Introduction

There is a high interest in pitch x loblolly pine hybrids both as a pulp species and increasingly as a lumber species. On moderate to good sites, the hybrids can grow exceptionally fast. Even on poorer sites the hybrids survive well and, comparatively speaking, will outgrow most other species. In an earlier test on minesoils, it was shown that the hybrids were adversely affected by excess moisture (Davidson 1984). The site for the current study was chosen to avoid excessively moist conditions.

The Study

In 1985 we planted 494 first- and second-generation seedlings of pitch x loblolly hybrid pine on an abandoned minesoil in Wyoming County, WV. This planting is part of a regional study to evaluate pitch x loblolly hybrid pines. There are 67 plantings in 22 states of these first- and second- generation hybrids (Garrett 1988). Most of the test plantings are on unmined lands.

Our planting plan was a completely randomized design of four tree linear plots replicated six times. Before the planting was completed, however, some of the seedlings were stolen. As a result, the final planting arrangement was 20 hybrids and 1 pure loblolly in 5 or 6 replications, and 10 hybrids in 1, 2, or 3 replications. For this report, only the 20 hybrids and 1 pure loblolly present in replications 1 to 5 are compared.

At the time of planting the seedlings had been grown in paper tubes for 1 yr. The tubes were removed before planting. A four-in-diameter hole about 5 in deep was made with a soil auger. As a seedling was placed in the hole, soil was

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[1]Poster presented at the International Land Reclamation and Mine Drainage Conference and the Third International Conference on the Abatement of Acidic Drainage, Pittsburgh, PA, April 24-29, 1994.

[2]Walter H. Davidson, Research Forester, USDA Forest Service, University Park, PA 16802.

firmed around it with a small mattock. The seedlings were planted on June 5 and 6. Conditions were excellent: light rain each night (June 4, 5, and 6), cloudy days, and temperatures from 70° to 80° F.

The site was a reclaimed surface mine area that had been graded and seeded to a reclamation grass mix. At the time of planting, there was a sparse cover of poverty grass and tall fescue with scattered mountain laurel and sourwood seedlings. Soil analysis showed pH 4.6, low nutrients, and no high levels of toxic elements. The soil was compacted and had a clay texture. No bulk density measurements were made.

Results

Survival was 99 % after one full year, and remained at that level until age 4 when it dropped to 98 %. Three seedlings died in the 1988 growing season. The next measurements were taken in April 1991 before growth had started for the season. The plantation was 6 yr old. Between the 1988 and 1991 measurements, survival declined to 93 %. There were 30 fewer seedlings. Some of the recorded losses were due to pilferage for Christmas trees and landscaping. At the time the 1991 measurements were taken, it was noted that 3 trees had been cut and 14 trees had been dug from the ground. There may have been additional pilferage during the period since the 1988 measurements were taken; if so, the evidence was no longer conspicuous.

Survival rates were computed as though the pilfered trees were alive. Six of the hybrids and one open-pollinated cross had 100 % survival. The poorest survival rate, 80 %, was shared by three hybrids.

Heights of individual trees ranged from 50 cm (1.6 ft) to 425 cm (13.9 ft). Average height for the plantation was 235 cm (7.7 ft). Eleven of the hybrids had average heights that exceeded the plantation average. Survival and average height by hybrid crosses are shown in table 1.

Discussion

All of the hybrid pines used in this test planting performed well. Even the poorest hybrid had 80 % survival after 6 yr in the field and averaged 170 cm (5.6 ft) tall.

Another test planting adjacent to the hybrid pine plots was established in 1982. Among the species in that test was shortleaf pine (Pinus echinata). After nine growing seasons, the shortleaf pine had 30 % survival and averaged 245 cm (8.0 ft) in height (data on file).

Conclusions

Pitch x loblolly pine hybrids have excellent potential for disturbed areas following fire, hurricane, or mining. They will survive well and grow satisfactorily on these tough sites. On better sites they are capable of producing pulp or saw timber on a relatively short rotation (Little and Wolf 1980).

At the present time outstanding hybrids have been identified; the problem is seed availability. Two techniques are being explored for seed production (Garrett 1988): (1) single-clone orchards that will be mass pollinated, and (2) clonal orchards of superior first-generation hybrids that will produce wind-pollinated second-generation hybrid seed.

Tree planters should be watchful for pitch x loblolly hybrid pine seedlings as they become available from State and private nurseries.

Table 1. Pitch X loblolly pines, Wyoming County, WV. (7 yr old from seed, 6 yr on-site, twenty seedlings planted per hybrid cross).

Hybrid cross code	Survival %	Average height, cm
22 X 15 - 54	100	210
22 X 16 - 269	100	265
25 X 66	95	195
(62 X 11 - 9) X (71 X 4 - 32)	100	275
66 - OP	95	190
(71 X 4 - 32) X (71 X 4 - 32)	90	175
(71 X 4 - 32) X OP	90	240
75 X 19 - 10	95	255
(75 X 15A) X (71 X 22)	90	195
77 X 1 - 66	95	270
(77 X 4 - 32) X (71 X 4 - 32)	80	170
77 X 4 - 43	100	240
77 X 19 - 10	100	285
78 X 4 - 43	90	245
(78 X 23) X (71 X 22)	100	250
(78 X 23) X OP	100	220
78 X OP	80	230
80 X 19	90	205
81 X 11 - 700	90	320
K - P X L	80	210
NL LOB	85	305

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THE TECHNIQUE OF RECLAIMING SUBSIDENCE AREAS BY USE OF A HYDRAULIC DREDGE PUMP IN CHINESE COAL MINES[1]

Zhenqi Hu, Longqian Chen, Jiacong Lin and Haibin Liu[2]

Abstract: The technique of reclaiming disturbed lands by use of a hydraulic dredge pump (HDP) (which is a set of hydraulic machines for earthwork), called "digging deep to fill shallow," has been used in the reclamation of subsidence areas in Chinese coal mines. This paper discusses the operation principle of the technique and its available conditions and summarizes its reclamation procedures, which are being used in current reclamation practice. The characteristics of the soil reclaimed by HDP were studied. The results showed that the reclaimed soil by use of HDP resulted in a massive structure soil, which was the mixture of original topsoil and subsoil, and had high clay content and no district horizontal layers. The moisture characteristics of the reclaimed soil was the most severe influence factor to plant growth because of the high moisture content (almost close to saturate) and slow infiltration. The soil fertility assessment indicated that the reclaimed soil was very poor. Therefore, the improvements of the reclaimed soils and the HDP technique are needed. Based on the analysis of the previous technical processes, a new technical process for the HDP reclamation method was presented in this paper. The author thought that the two processes such as "excavation of soil" and "backfill and sedimentation" needed to be improved, and the other important processes such as "removal and backfill of topsoil," "establishment of drainage system," etc., should be added.

Additional key words: mining subsidence, reclamation, hydraulic dredge pump, technical processes, reclaimed soils

Introduction

In China, most of the coal is taken from underground mines, accounting for 96% of the coal output. Underground coal mining has caused a large amount of lands to subside, which has led to farmland losses and caused severe conflicts between farming and mining. According to statistics, the lands subsiding from coal mining total more than 13,300 hectares each year, of which half is located in the plain area, which is all prime farmlands (Sun and Li 1990). It is well known that China has a very large population and the cultivatable land shortage is very serious. The mean of cultivatable and permanent farmland per person in China only equals about one-third of the average value in the world. This situation makes the reclamation of subsidence lands become an urgent task for our country.

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[1]Paper presented at the International Land Reclamation and Mine Drainage Conference and the Third International Conference on the Abatement of Acidic Drainage, Pittsburgh, PA, April 24-29, 1994.
[2]Zhenqi Hu, Associate Professor, Longqian Chen, Assistant Teacher, Jiacong Lin, Professor, and Haibin Liu, Graduate Student, Department of Mining Engineering, China University of Mining and Technology, Xuzhou, Jiangsu, People's Republic of China.

In the comprehensive treatments for subsidence lands due to coal mining, the method of reclaiming lands by hydraulic dredge pump (HDP) called "digging deep to fill shallow" is being used in our country. Since this reclamation method is easy to operate and has fewer costs, study on the technical procedure of HDP is neglected. Thus, the HDP method usually leads to poor reclaimed soils and fewer benefits. This paper tries to evaluate the technique and develop rational technical procedures for the HDP method.

Operation Principle of HDP and Its Available Condition

The HDP, in fact, is a set of machines, with a high-pressure pump, two hydraulic giants (water syringes), a slurry pump, some steel and plastic pipes, etc. The basic principle of the reclamation method is that using the HDP machine simulates the natural water erosion and turns the mechanical and electrical power into hydraulic power for digging, transporting, and filling of soils. Usually, land reclamation needs five procedures such as excavating, loading, transporting, unloading, and leveling of soils. This method can take four procedures except of the leveling by a HDP. Thus, this method has many advantages: the equipment is simple, the cost is low, the operation efficiency is high, and the operation is convenient and not affected by weather.

It is well known that the surface above a mined area will subside after the excavation of coal, which produces a larger subsidence trough. In China, the resulting subsidence is often very deep (1.5 m - 6.0 m), and water accumulates in some of the subsided areas. We can divide the subsided trough into two parts: "deep area" and "shallow area" based on the particular landscape and reclamation requirements (see figure 1). The purpose of the HDP reclamation method is to make the "deep area" deeper for fish ponds and the excavated soils are filled the "shallow area" for farmlands. Thus, if the HDP method is to be used, the following conditions should be met:

(1) The subsidence depth is not grater than 2.5 m.
(2) The area of accumulating water is not very large so that the HDP is easy to operate.
(3) The amount of soils excavated from the "deep area" is equal to or grater than the needed soils filling in the "shallow area" for farmlands with a designed elevation.
(4) The produced fish ponds meet desired requirements.
(5) Water resource is enough for HDP operation.

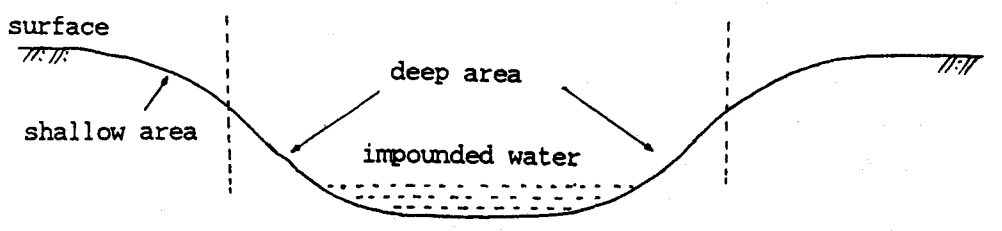

Figure 1. Profile of subsidence trough.

Evaluation of the Current HDP Method in China

Technical Procedures of the Current HDP Method in China

The procedures of the HDP operation used in Chinese coal mining areas can be summarized as follows:

(1) Producing high-pressure and high-speed water by a high pressure pump. Usually, the water speed is 50 m /h.
(2) Excavating soils by use of hydraulic giants (water syringes) with the high-pressure and high speed water, which makes the soil become slurry.
(3) Transporting the slurry to the "shallow area" to be filled through transportation pipes by use of the slurry pump.
(4) Filling and settling; the slurry can be filled in the designed "shallow area" by moving the transportation pipes, then letting the slurry settle down naturally. Generally, the filled lands need more than 5 months for stilling down after the stop of the filling work.
(5) Leveling the reclaimed land by manual work or dozers.

Evaluation of the Reclaimed Soils

Soil is a medium for plant growths (Hillel 1982). The purpose of land reclamation is to restore the soil productivity of the destroyed lands. Thus, the condition of reclaimed soils is the logical criterion for evaluating the effectiveness of reclamation. To evaluate the effectiveness of the HDP reclamation method, we choose two experimental sites (Pingdingshan and Peixian) for testing. In each site, two kinds of lands -- undamaged farmland and reclaimed land by HDP -- were chosen for comparison. The soil tests from the two sites indicated similar results as follows:

1. The soil profile examination showed that soil reclaimed by use of the HDP resulted in a massive structure soil and had no topsoil or distinct horizontal layers, which was the mixture of original topsoil and subsoil from adjacent area. It was easy to recognize that clay and moisture contents were very high, and the surface was hardened and imperious soil. The thickness of the reclaimed soil was about 60-85 cm. The underlying was the original soil profile. But the undamaged farmland had an average of 15 cm of topsoil and distinct horizontal layers. The upper layer of the farmland soil was darker than the underlying layers as well as the reclaimed soil because of the accumulation of organic matter. The granular structure dominated the topsoil of farmland, while the platy and subangular blocky structure were noted in the reclaimed soil.

2. The analysis of soil physical properties (see table 1) of the reclaimed soil indicated that the bulk densities were nearly ideal for plant growth. But the bulk density of the top layer (0-20 cm) was a little larger than that of farmland. The farmland soils had the biggest bulk densities at the depth of 20-40 cm, which revealed a compaction problem due to farming production. But the reclaimed soils had no compaction problem. However, the moisture characteristics of the reclaimed soil were the most severe influence factor to plant growth because of the high moisture content (almost close to saturate) and slow infiltration (see table 1 and figure 2). The moisture content of reclaimed soils was 1.2 - 2.5 times that of farmland soils. The infiltration rates in the reclaimed soils were about one-tenth (Peixian site) and one-fifth (Pindingshan

site) that of farmland soils. This situation may lead to losses of soils and nutrients. Therefore, the establishment of a drainage system is the key to making reclamation success and should be one of the procedures of the HDP operation.

3. The soil fertility assessment indicated that the reclaimed soil was very poor and that amelioration treatments are necessary (see table 2).

Table 1. Results of soil bulk density and moisture content

Site	Plot	depth, cm 0-20	20-40	40-60	60-80
		Bulk density (g/cm^3)			
Pingdingshan	farmland	1.16	1.55	1.56	1.49
	reclaimed	1.33	1.47	1.37	1.08
Peixian	farmland	1.32	1.64	1.42	1.42
	reclaimed	1.41	1.34	1.33	1.46
		Moisture content (%)			
Pingdingshan	farmland	13.8	14.0	15.5	14.3
	reclaimed	26.3	35.5	38.4	31.3
Peixian	farmland	26.4	19.7	22.8	25.2
	reclaimed	31.7	30.9	34.3	43.4
		Infiltration rate (cm/sec)			
Pingdingshan	farmland	0.0033			
	reclaimed	0.0006			
Peixian	farmland	0.0010			
	reclaimed	0.0001			

The Problems Existing in the Current HDP Technical Processes

Based on the above analysis, the problems existing in the current HDP operating processes are:

1. Reclaimed soil is poor. The current HDP method resulted in a massive structure soil, which was the mixture of original topsoil and subsoil and had high moisture content. Thus, the soil productivity was poor.
2. Sedimentation took a long time. Since the reclaimed soils settled down naturally, the water in the reclaimed land was difficult to drain out. Thus, the time for settling was very long (usually more than 5 months), which postponed the procedure of leveling reclaimed lands. This situation affected reclamation efficiency.

Table 2. Some macronutrient contents

type of soil	depth, cm	total N, %	total P, %	rapidly available nutrients N, ppm	P, ppm	K, ppm	O.M., %
			Pindingshan				
farmland	0-20	0.107	0.167	87	14.1	112.5	1.90
	20-40	0.066	0.130	47	3.7	95.0	1.23
reclaimed	0-20	0.042	0.091	25	2.7	97.5	0.49
	20-40	0.042	0.098	27	1.4	95.0	0.54
			Peixian				
farmland	0-20	0.106	0.157	85	8.4	104.0	1.62
	20-40	0.047	0.140	34	3.6	104.0	0.51
reclaimed	0-20	0.045	0.132	30	6.0	117.5	0.70
	20-40	0.040	0.125	26	5.9	104.0	0.41

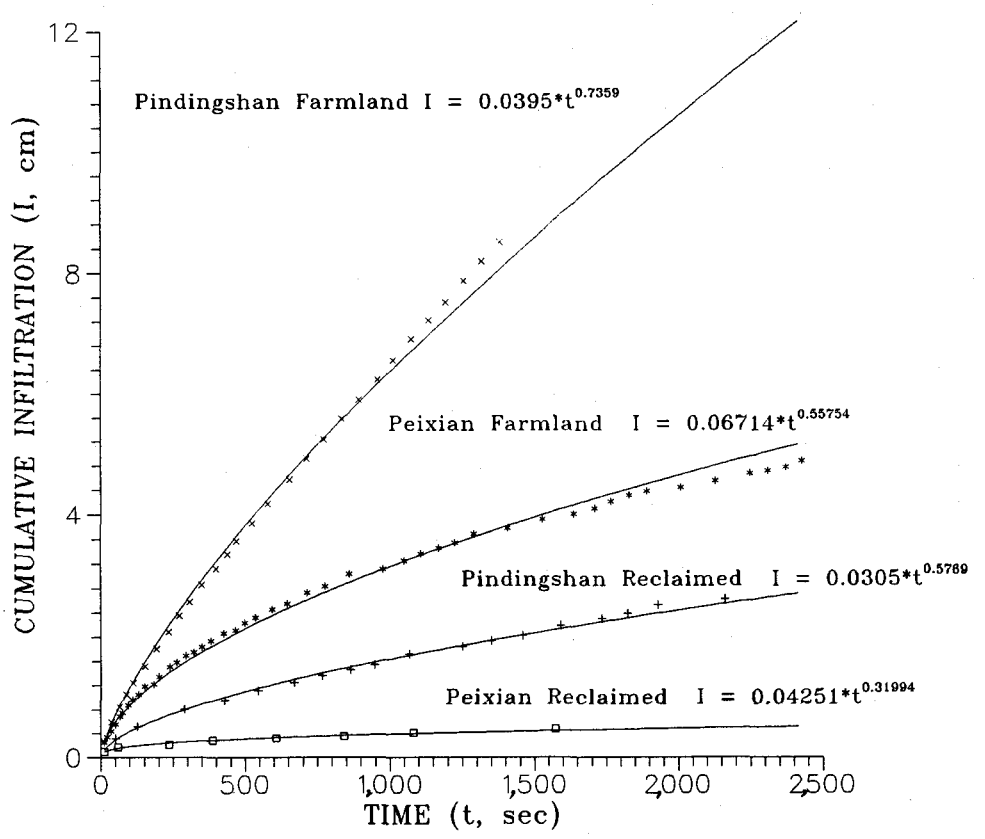

Figure 2. Comparison of infiltration (farmland and reclaimed)

Development of a New HDP Technical Process

For solving the problems in the current HDP reclamation method, a new HDP technical process was developed as follows:

(1) Removal of the topsoil of the "shallow area".
(2) Establishing drainage ditches and return ditches.
(3) Producing high-pressure and high-speed water by a high-pressure pump.
(4) Hydraulic excavating soils by use of hydraulic giants (water syringes) in a new excavation procedures.
(5) Hydraulic transporting soils through transportation pipes by use of the slurry pump.
(6) Filling and settling in a new filling orders explained as followings.
(7) Completing the drainage system.
(8) Leveling of the reclaimed land.
(9) Backfill of the topsoil.

The new HDP technical process adds some procedures such as (1), (2), (7), and (9), and improves (4) and (6), i.e., the new excavating and filling orders. Other procedures are the same as the current HDP method. The following will explain some added and improved procedures:

A. Adding the procedures of removal and backfill of the "shallow area" topsoil. The topsoil has high organic matter and nutrient contents, which takes a long time for its formation. Thus, the removal and backfill of the topsoil are needed for rapidly restoring the productivity of reclaimed soils. Although the treatment has been used in surface mined land reclamation, the current HDP method does not use it. Therefore, the added procedures (removal and backfill of the "shallow area" topsoil) are effective treatments for improving the HDP method.

B. Establishing drainage ditches and return ditches: Since the characteristics of the HDP reclamation method are hydraulic excavation and hydraulic transportation, the drainage is the key to affect the settling speed of the slurry and reclamation time. Thus, the establishment of drainage ditches and return ditches before HDP operation is needed. The return ditches are from the reclaimed land (i.e., "shallow area") to the water resource for the use of the high-pressure pumps, which makes the water reutilize. The purpose of establishing drainage ditches and return ditches is to drain the water in the reclaimed soils immediately, which turns the natural settling into accelerating settling. Thus, reclamation efficiency is high.

C. Completing the drainage system: After or close to the complement of HDP operation, the drainage system should be completed immediately so that the rest of the water in the reclaimed soils is drained out for early leveling of the land and planting crops.

D. Improving the orders of excavating and filling soils: The current HDP method resulted in the mixture of topsoil and subsoil and may sometimes lead to that subsoil covered on the topsoil. Topsoil in the "shallow area" can be removed, but the topsoil in the "deep area" is difficult to remove because of wet or water accumulation. For constructing good root medium, the order of excavation and filling of the "deep area" soil should be improved to keep similar

soil layers. Based on the principle of constructing new root medium in the surface mined land (Hu 1991), the new excavating and filling method is (see figure 3):

a. Dividing the "deep area" and "shallow area" into several blocks based on the landscape and reclamation plan. After the removal of topsoil from the "shallow area," the low banks between blocks should be built for filling.

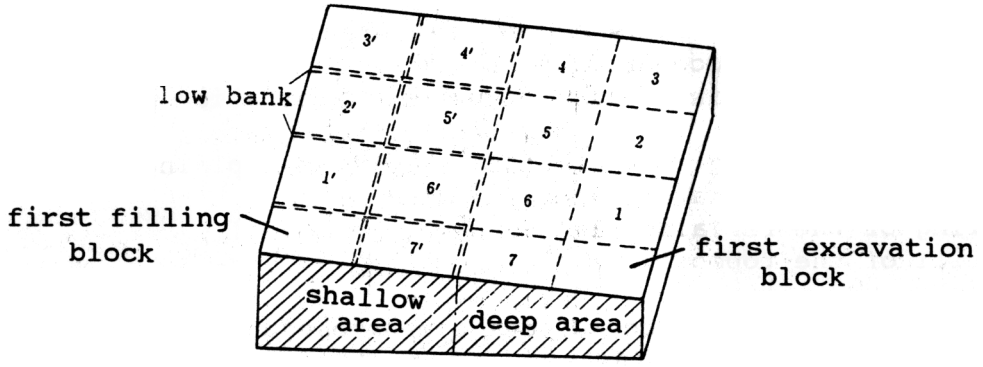

Figure 3. New order of excavating and filling soils.

b. Excavating the "first excavation block" first by use of HDP, and the excavated soils are filled in the "first filling blocks." This procedure is similar to the current HDP method, which may lead to the mixture of topsoil and subsoil in the "first filling block." Thus, the "first excavation block" and "first filling block" should be as small as possible. The purpose of this procedure is to create a condition for excavating the "deep area" soil layer by layer.

c. Dividing the "deep area" soils into two layers: upper (30-50 cm) and lower (deeper than 30-50 cm) first, then following the orders to excavate and fill soils (assume: "deep area" has n+1 blocks and "shallow area" has n'+1 blocks):

"deep area" excavating ...		"shallow area" filling ..., (to be ...)
1 block upper layer	---->	"first filling block"
1 block lower layer	---->	1' block, (1' block lower layer)
2 block upper layer	---->	1' block, (1' block upper layer)
2 block lower layer	---->	2' block, (2' block lower layer)
3 block upper layer	---->	2' block, (2' block upper layer)
	.	
	.	
	.	
n block upper layer	---->	(n-1)' block, ((n-1)' upper layer)
n block lower layer	---->	n' block, (n' block lower layer)
1 block upper layer piled on the "first filling block	---->	n' block, (n' block upper layer)

The order of excavating and filling of soils can be summarized as follow:

$$\begin{cases} \text{soil structure of i' block} = \text{"(i+1) block upper layer"} + \text{"i block lower layer"} \quad (i=1,2,\ldots n-1) & (1) \\ \text{soil structure of n' block} = \text{"1 block upper layer"} + \text{"n block lower layer"} & (2) \end{cases}$$

According to the new order of excavating and filling of soils, the new constructed soil structure keeps the similar original soil layers except for the "first filling block." After the complement of the water drainage from the new constructed land, the original "shallow area" topsoil should be backfilled. Therefore, this new method may create a new root medium with almost two topsoil layers, which is much better than the old HDP method.

Concluding Remarks

HDP reclamation method is being widely used for subsidence land reclamation in China, but the technical process has not been studied in detail. This paper is the first to explore the principle of the HDP method and the soil condition reclaimed by HDP, and developed a new HDP technical process. Although the new HDP technical process needs further reclamation practice, the exploration of the HDP method may make many experts pay grater attention to the improvement of the HDP technique.

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RESTORATION OF PYRITIC COLLIERY WASTE WITH SEWAGE SLUDGE IN THE MIDLANDS COALFIELD, ENGLAND, UNITED KINGDOM[1]

R. Neil Humphries[2], Gordon E. McQuire[3] and Martin Sly[4]

Abstract: A trial was set up in 1990 in the Midlands coalfield in the United Kingdom (UK) to evaluate the use of sewage sludge to revegetate colliery waste tips containing 1 - 2% sulfur as iron pyrites.

The rate of sewage sludge application is currently restricted by legislation and codes of practice to maximum concentrations of potentially toxic elements (copper, nickel, zinc, etc.) in the soil or waste after application. Following this guidance, an application rate of 250 mt/ha dry solids was applied at the trial site. At this rate, the colliery waste became extremely acidic pH <4.0. From experience elsewhere, much higher levels have been found to be necessary to control acidification in the absence of other measures or treatments.

In view of the restriction on the amount of sewage sludge that can be applied, it is recommended that the current practice of covering fresh colliery wastes with soil or low sulfur spoil to a minimum depth of 0.45m is continued in the UK. Where this is not possible, the sludge must always be applied with sufficient neutralising agent to control the potential acidity. If the acidity cannot be maintained above pH 5.0, the guidelines do not permit the application of sewage sludge.

Additional Key Words: pyritic wastes, acidity, sewage sludge, soil cover.

Pyritic Colliery Waste in the UK Coalfields

Current coarse washery discard colliery wastes in the UK average between 0.2% and 2.0% sulfur in the form of iron pyrites (Glover 1984), although locally much higher levels are frequently recorded in the more pyritic wastes. The most pyritic wastes occur in the Midland coalfields of Nottinghamshire and Yorkshire and in the western Scottish coalfield. Modern waste tips are formed using earth-moving plant which, through compaction, has largely eliminated the risk of spontaneous combustion and reduced the oxidation of pyrite within the tip. There remains the potential for acid generation in the final surface layers, which generally are required to be uncompacted and permeable for the purpose of revegetation.

While the modern construction methods for waste tips have much reduced the risk of acid mine drainage, there is still concern about surface acidification and subsequent pollution of watercourses and bodies by acidic runoff containing soluble salts and metals. Also, surface acidification can be of significance for restoration by causing poor vegetation growth and, in extreme cases, loss of vegetation. Both conditions can be time consuming and costly to rectify. Therefore, the prevention and treatment of acid potential and acidity has been, and continues to be, a priority in modern colliery waste restoration in the UK.

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[2]R. Neil Humphries, Technical Director, Humphries Rowell Associates, Loughborough, UK.

[3]Gordon E. McQuire, Chief Civil Engineer, British Coal Corporation, Eastwood, Nottinghamshire, UK.

[4]Martin Sly, Head of Tips and Restoration, British Coal Corporation, Midlands Group, Edwinstowe, Nottinghamshire, UK.

Practices for Treatment of Acidity

Use of Neutralizing Agents

Traditionally, the method to control acid generation and to treat existing acidity in the surface layer of colliery waste has been to apply neutralizing agents, such as limestone ($CaCO_3$) or lime (CaO), as used in the management of agricultural soils (ADAS 1981).

Where the potential for acid generation is low (i.e., on "average" pyritic sulfur <0.5%S) and where existing acidity has not been extreme (i.e., pH >/= 4.5), the traditional method has generally been successful for colliery waste when the recommended practices for application and incorporation have been properly implemented (Rowell and Humphries 1985).

However, the approach has not been consistently effective where the potential for acid generation is moderate to high (i.e., >1.0%S), and/or where extreme acidity exists (i.e., pH <3.5). This is due to the very high quantities of neutralizing agent needed and often the need for repeated annual applications (Simmons 1984, Backes et al. 1985, Rowell and Humphries 1985). Both can cause severe practical difficulties during application and incorporation, and have been a source of public nuisance owing to windblown material during application, besides inducing problems for vegetation growth. In addition this approach is often costly to implement.

Use of Soil Cover

Since the 1970's planning consents granted by the local government mineral planning authorities have normally contained a condition that requires new, and extensions to existing, colliery waste tips to be covered with a layer of soil or "soil-forming materials" (Humphries 1984).

Where soil material is available, this method has provided a means whereby the rate of acid generation can be more effectively controlled. The effectiveness is related to the depth and type of soil cover and the degree of pyrite oxidation prior to covering. The control of acidification appears to have been successful where a soil cover in excess of about 0.5 m has been applied (although no systematic field assessment appears to have been undertaken). Shallower depths of cover, and in particular soil cover of less than 0.2 to 0.3 m, does not prevent acidification of the underlying waste (Rowell and Humphries 1985). However, the application of a cover of soil has proved to be generally beneficial for vegetation, even where the waste has acidified beneath a shallow cover. Depending on depth of cover, it has been possible to establish a range of vegetation types and land uses (Humphries and McQuire 1994), which can be managed by normal agricultural, forestry, and wildlife practices.

Alternative Methods of Control and Treatment of Acidity

In the past soils were not stripped before tipping colliery spoil, and many older tips in the UK have little or no soil material to cover the final surface (Humphries 1984). Because of this, there has been considerable interest in alternative methods of controlling pyrite oxidation and acidity (Macpherson 1987, Metcalfe and Lavin 1991, Pulford 1991a/b). Also, because of economic pressure to reduce waste disposal costs, there has been recent interest in the coal mining industry in the use of organic wastes, either instead of spreading a soil layer or in conjunction with a shallower layer of soil over the colliery waste.

There is evidence from laboratory studies that the oxidation of pyrite can be inhibited by organic waste materials such as manures (Backes et al. 1987). Furthermore, the work by the water utility Yorkshire Water Ltd. has shown that acidic colliery waste can be effectively revegetated using treated human sewage effluent (Metcalfe and Lavin 1991).

The practice of applying sludge to agricultural land, mineral wastes, and derelict land has a long and successful history in the UK, where it has been carried out in a controlled manner and according to recommended practice.

Currently, about 5% of the 1 million mt dry solids of sewage sludge produced annually is used in land reclamation (Hall 1988). With an ever-increasing requirement for the utilities to dispose of sewage sludge in an environmentally safe and economic way, they see its use in colliery waste reclamation schemes as a major opportunity for disposal (Severn Trent Water 1990).

Trial Use of Sewage Sludge

Introduction

In 1990, Nottinghamshire Group (now the Midlands Group) of the British Coal Corporation and the water utility Severn Trent Water Ltd. set up a long-term trial to evaluate the use of sewage sludge to revegetate colliery waste tips as they are completed. Humphries Rowell Associates (HRA) was contracted in 1992 to monitor the trial at Thoresby Colliery for a 5 yr period.

The trial comprises some 31 plots of various treatments. These include soil or soil-forming material (sand) spread to a depth of about 0.6 to 0.7 m over untreated waste, a mixture of sand and waste to a depth of 0.5 m, and colliery waste only. The trial includes these treatments with and without the addition of undigested sewage sludge as de-watered cake at 250 mt/ha of dry solids. The sewage sludge was applied in November 1990, and was incorporated to a depth of about 0.3 m using both tines and a "spading" machine, generally resulting in a uniform mixture of waste and sludge. The plots have been either sown with a ryegrass (*Lolium perenne*) - white clover (*Trifolium repens*) pasture mixture or planted with tree species (mainly silver birch (*Betula pendula*), sessile oak (*Quercus petraea*) and Scot's pine (*Pinus sylvestris*)). The plots with a 0.6 to 0.7 m layer of soil or soil-forming material represent the current restoration treatment practised by the Group.

Various chemical data (nutrients, metals, etc.) for the "soil layer" for the period 1990-91 were collected by British Coal; thereafter HRA collected both soil and vegetation for analysis and recorded vegetation growth and development.

Results After 4 Yrs

The colliery waste plots without a soil cover show the typical rapid decline in pH from the saline fresh waste to an extremely acid waste resulting from pyrite oxidation over some 4 yrs (table 1). The addition of the sewage sludge to the waste at the permitted rates did not inhibit the development of acidity. In contrast, the "soil layer" of the soil-covered plots, with or without sludge treatment, did not become acidified (table 2).

Despite the generation of acidity within the colliery waste plots treated with sewage sludge, the ground cover vegetation has largely persisted, although at a lower level of cover than the soil- and sand-covered plots. Also, the growth of birch in 1993 on these plots was comparable to that on the plots with a cover of soil or sand (table 3). In contrast, the untreated shale plots had almost no ground cover vegetation and the few surviving trees generally exhibited "dieback" and a greater proportion of trees without expanded leaves.

Table 1. Development of acidity in untreated and sewage sludge treated waste, pH.

	1990	1991	1992	1993
Untreated colliery waste	8.3	NA[1]	3.1	2.9
Treated with sewage sludge	8.3	6.6	4.9	3.5

[1]NA = Not available.

Table 2. Soil reaction (pH) of plots in 1993.

	Waste	Waste and sludge	Sand layer	Sand layer and sludge	Soil layer	Soil layer and sludge
pH....	2.9	3.5	5.6	6.2	5.9	6.6

Table 3. Growth of birch and herbaceous ground cover on plots in 1993.

	Waste	Waste and sludge	Sand layer	Sand layer and sludge	Soil layer	Soil layer and sludge
Mean increase in birch height (cm).....	-23[1]	22	10	19	25	33
Mean increase in birch spread (cm).....	-5[1]	33	16	23	32	33
Percentage trees alive.....	4	74	68	63	65	52
Percentage herbaceous ground cover.....	3	40	80	95	80	100

[1]Represents net dieback.

Discussion

The ability of organic matter to control the oxidation of pyritic spoil has been demonstrated and is well understood (Backes et al.1987, Pulford 1991a). The failure of the sewage sludge to inhibit acidification of the pyritic colliery waste in the trial is however in accordance with experience elsewhere in the UK and in the United States, where low rates of sludge have been applied (Metcalfe, personal communication). Much higher rates, such as 750 mt/ha dry solids, appear to be necessary to control acidity.

Backes et al. (1987) suggested that the inhibition of pyrite oxidation by organic manures may be dependent on a close contact between the pyrite and the organic matter. As a good degree of mixing had been achieved in the trial, the failure can only be reasonably attributed to the low rate of application.

The rate of sewage sludge applied in the trial had been determined by the supplying utility, Severn Trent Water, which complied with current UK recommended practice and legislation designed to prevent the buildup and mobilization of potentially toxic elements (copper, nickel, zinc, etc.). The rate was limited to the maximum currently permitted for agricultural land (Department of the Environment 1992), which is the same as that recommended by the Forestry Commission and Water Research Centre (Wolstenholme et al. 1992) for woodland and public open space. Hence under current guidelines, the use of higher rates of sewage sludge necessary to control acidity at the site would not have been permitted. Furthermore, Wolstenholme et al. (1992) recommended that the maximum application rates for restoration should be restricted to 100 mt/ha dry solids.

The draft guidance issued by the Water Research Council (Hall 1988) advises that sewage sludge should not be relied upon to control acidity where the soil layer has the potential to become acidic as in the case of colliery waste. In these cases they recommend that the material should be suitably limed. Unfortunately, this advice was not included in the operational guidelines published subsequently by Severn Trent Water (1990) and implemented at Thoresby.

While the same restrictions apply to soils and soil-covered wastes, there is generally little risk of acidification and therefore generally no need for additional lime treatments or higher rates of sludge application. Hence, from both an operational point of view and one of least risk, sewage sludge application is best restricted to soil covered wastes.

Where this is not possible, it is essential that sufficient neutralizing agent (e.g., limestone) is applied to treat the potential acidity.

Guidelines for Restoration of Pyritic Colliery Wastes

For reasons of minimizing pollution and achieving successful and cost-effective restoration, the control of acidification must remain a priority for restoration of colliery waste tips in the UK.

The most effective means of controlling acidification is to cover pyritic wastes with soil material. It is recommended that the current British Coal best practice of immediately covering fresh mine waste with soil material to a minimum depth of 0.45 m (or greater depending on proposed land use) be continued. Where soil is not available, a cover of low pyritic material, which may be colliery waste, should be spread over the final layer of pyritic material to a depth of 0.45 m. Where the cover is less than 0.45 m, the current practice of adding neutralizing agents, such as limestone, to the underlying waste should be continued according to the scheme shown table 4.

Wherever sewage sludge is applied in the UK, the rates of application should accord with the current guidelines and codes of practice. These are published by the Department of the Environment (1992) for agricultural soils, and the Forestry Commission (Wolstenholme et al. 1992) for forestry and public open space. At these rates, sewage sludge is unlikely to control acidification in pyritic wastes. In the absence of soil material, sewage sludge should always be applied with neutralizing agents following the recommendations given in table 4. If there is any uncertainty about the control and maintenance of acidity above pH 5.0, sewage sludge must not be applied.

Table 4. Recommended treatment strategy for pyritic wastes.

	Soil cover			
Pyritic sulfur (%S)	<150 mm	150 to 300 mm	300 to 450 mm	>450 mm
<0.5.....	CLD[1] to 300 mm	CLD to 300 mm	½ CLD to 150 mm	Nil
0.5 to 5.....	CLD + 20 mt/ha per % S to 300 mm	CLD + 20 mt/ha per % S to 300 mm	½ CLD + 10 mt/ha per % S to 150 mm	Nil
>5.....	Cover with 450 mm of low pyritic spoil <5% S and treat as above			

[1]CLD = Calculated lime demand using ADAS (1981) methodology. If limestone addition rates > 20 mt/ha, use only calcitic limestone.

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CHEESE WHEY AS AN AMENDMENT TO DISTURBED LANDS: EFFECTS ON SOIL HYDRAULIC PROPERTIES[1]

Gary A. Lehrsch and Charles W. Robbins[2]

Abstract: Whey, the liquid byproduct of cheese production, can improve minesoils by increasing the aggregate stability of soils high in sodium or susceptible to erosion. Whey effects on soil hydraulic properties, however, are not known. In this experiment, we determined whey effects on infiltration rates (at water potentials of −30 mm or less) and unsaturated hydraulic conductivities of surface soil horizons after a winter wheat (*Triticum aestivum* L.) growing season. The experimental design was a randomized complete block with three replications of four liquid whey application treatments, totaling either 0, 202, 404, or 808 Mg/ha (control, low, medium, and high, respectively). In Fall 1992 near Kimberly, ID, a field of Portneuf silt loam (*Durixerollic Calciorthid*) was leveled, subsoiled, then roller-harrowed twice. After planting Malcolm wheat on September 15, we furrowed all plots and then constructed a berm around each. At 3-week intervals beginning on May 19, 1993, either zero, one, two, or four flood applications of 202 Mg/ha of whey were made to each plot, without subsequent tillage. After August wheat harvest, a tension infiltrometer was used to measure vadose zone, unsaturated flow characteristics in the bottom of undisturbed furrows, where most whey had infiltrated. Infiltration rates at potentials of −60 and −150 mm decreased linearly as whey applications increased from 202 to 808 Mg/ha. At a potential of −60 mm, hydraulic conductivity increased but then decreased with whey additions. In short, soil hydraulic properties were little affected by surface whey additions of 404 Mg/ha or less.

Additional Key Words: infiltration, hydraulic conductivity, tension infiltrometer, food processing wastes.

Introduction

The soils of lands disturbed by human activities often exhibit characteristics that make land reclamation difficult. Plant establishment and/or growth is often inhibited by excess acidity or alkalinity, or the presence of toxic metals or excess salts (Logan 1992). Less frequently, land may be contaminated with relatively high concentrations of sodium, radionuclides, toxic organics, or even excess plant nutrients (Hossner and Hons 1992, Logan 1992, Power et al. 1974). Physical properties, as well, may be adversely affected. Texture, bulk density, and structure are frequent impediments to seed germination or plant growth (Hossner and Hons 1992). Coarse-textured materials commonly have limited water-holding capacities, are low in organic matter (and hence in cation exchange capacity), and are prone to surface crust formation (Hossner and Hons 1992, Zabowski et al. 1993). Fine-textured materials, while they can hold more water, can have lower infiltration rates, hydraulic conductivities, or anaerobic root zones (Sopper 1992, Zabowski et al. 1993).

To correct one or more of these plant-growth limiting characteristics, inorganic and/or organic amendments are often added to the soils of disturbed landscapes. Inorganic amendments include fertilizer, lime, elemental sulfur, and gypsum (Logan 1992, Robbins and Gavlak 1989, Zabowski et al. 1993). Organic amendments, because they add nitrogen and organic material as well as improving soil physical properties such as permeability and porosity, are widely used (Logan 1992, Sopper 1992). Sewage sludge is also frequently applied (Haering and Daniels 1992, Sopper 1992). Other organic amendments include papermill sludge, cement kiln dust-stabilized sludge, municipal solid waste, compost, manure, plant residues, and wastewater

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[2]Gary A. Lehrsch and Charles W. Robbins, Soil Scientists, U.S. Department of Agriculture, Agricultural Research Service, 3793 N 3600 E, Kimberly, ID 83341, USA.

(Logan 1992, Robbins and Gavlak 1989, Zabowski et al. 1993). Cheese whey, the liquid byproduct of cheese production, has also been used as an organic amendment to help reclaim soils high in sodium (Jones et al. 1993, Lehrsch et al. 1994, Robbins and Lehrsch 1992). A sodic soil, by definition, has a saturation paste extract electrical conductivity (EC_e) ≤ 4 dS/m and a sodium adsorption ratio (SAR) >13 (Richards 1954).

Whey is readily available at cheese production facilities. In the United States in 1991, the production of 366,000 Mg of creamed and low-fat cottage cheese (U.S. Crop Reporting Board 1992) generated nearly 3 million Mg of acid whey (Jones et al. 1993). Though often fed to poultry or livestock, whey is frequently considered a waste product and disposed of in sewage treatment facilities or by land application (Watson et al. 1977), nearly always at a cost to the cheese producer. When whey is available at little or no cost, it is an attractive organic amendment.

Whey's potential as an amendment for land reclamation, particularly in the western United States where soils can be affected by relatively high levels of sodium and/or lime, is great. The low pH of the whey decreases soil solution pH and thus increases Ca solubility. Microorganisms, as they decompose the lactose and proteins in the whey (Summers and Okos 1982), produce CO_2 and organic acids that also increase Ca solubility (Robbins 1985). All of these processes will speed the leaching of exchangeable Na from a sodic soil profile when sufficient water is passed through the soil. Lehrsch et al. (1993), in a study of the aggregate stability of non-sodic soils, noted that the addition of soluble salts (such as are present in cheese whey) to the soil solution should reduce the diffuse double-layer thicknesses of clay domains, resulting in clay flocculation. This improved aggregation changes the pore size distribution, usually increasing the flux density of both water and air through the soil profile (Hillel 1982). Adding and incorporating whey lactose stimulate aerobic microbes that produce polysaccharides that will stabilize aggregates (Allison 1968). In a greenhouse study, Kelling and Peterson (1981) found that an application of 25 mm of whey improved aggregation as much as did application of 22.4 Mg/ha of corn (*Zea mays* L.) residue. Lehrsch et al. (1994) found aggregate stability to increase from 25% to 80% when 808 Mg/ha of acid whey were surface-applied, then incorporated into sodium-affected soils. As aggregate stability increases, erosion decreases (Luk 1979). Watson et al. (1977) measured up to a fourfold increase in infiltration rates into a fallow, non-sodic soil about 3 months after a surface application of sweet whey. They attributed the marked infiltration increases to improved soil structure.

Cheese whey does, however, have disadvantages that may outweigh the advantages noted above. Acid whey cannot be economically dehydrated; thus, its large volume and mass make handling troublesome. If it is surface applied without incorporation, odor control may be necessary. Transportation distances, as with many potential amendments, must be relatively short for whey to be economically attractive.

Problems may also occur if too much whey is applied. High whey application rates could increase root zone salinity (Jones et al. 1993, Robbins and Lehrsch 1992, Sharratt et al. 1962). Excessive whey applications could also decrease infiltration rates in the short term owing to organic overloading (McAuliffe et al. 1982, Watson et al. 1977). Organic overloading in particular can make management and/or reclamation difficult. Two sweet whey applications, each 200 mm or more, by Watson et al. (1977) decreased ponded infiltration rates by 13% to 67%. McAuliffe et al. (1982) found saturated hydraulic conductivities to decrease by approximately 50% within 2 days after they applied only 35 mm of a dilute sweet whey. The hydraulic conductivities did increase, however, 1 to 3 weeks after the second whey application. Additional research on the use of acid whey for reclaiming both saline-sodic and sodic soils has been recommended (Robbins and Lehrsch 1992).

Despite these disadvantages, whey may still have value as an amendment for land reclamation, particularly for sites with soils high in sodium or susceptible to erosion. While the effects of whey on soil physical and chemical properties have been studied (Jones et al. 1993, Lehrsch et al. 1994, Robbins and Lehrsch 1992), its effects on soil hydraulic properties such as infiltration rates (at water potentials of –30 mm or less) or unsaturated hydraulic conductivities have received comparatively little attention. Thus, the objective

of our study was to determine the effects of surface-applied whey on the hydraulic properties of surface soil horizons after a winter wheat growing season.

Materials and Methods

This experiment was conducted 2.2 km northeast of Kimberly, ID, on a Portneuf silt loam, previously cropped to barley (*Hordeum vulgare* L.). The Portneuf has 22% sand, 12% clay, 0.72% organic C, a pH of 7.6, and cation exchange capacity of 126 $mmol_c$/kg. In the fall of 1992, the site was leveled with a grader, subsoiled to a depth of 28 to 30 cm with shanks 30 cm apart, then roller-harrowed twice. This tillage was assumed to simulate the disturbance caused by soil replacement, deep ripping, and seedbed preparation during mined-land reclamation. On September 15, with no fertilizer, Malcolm wheat at a row spacing of 18 cm was planted; then furrows were formed every 76 cm in the surfaces of all plots. Two days after planting, owing to low soil water contents, the site was furrow-irrigated for 24 h. On May 3, 1993, we constructed berms of soil around each 10.7- by 15.2-m plot. The plots were sprayed on May 13 with 2,4-D for broadleaf weed control. To satisfy the wheat's transpiration demand, we used a solid-set sprinkler system to apply 65±16 mm (mean ± SD) of water in 24 h on May 25 and 76±14 mm in 28 h on June 30 and July 1. This water commonly has a pH of 8.2, an EC of 0.5 dS/m, and an SAR of 0.65 (Lehrsch et al. 1994).

The whey used in our study was trucked from a cheese plant 63 km east of our field site. The whey, a mixture of 75% sweet (from the production of hard or cheddar-type cheeses) and 25% acid (from the production of soft or cottage cheeses), contained 0.50±0.09 g P, 16.7±2.5 mmol Ca, 3.7±0.8 mmol Mg, 31.1±10.0 mmol Na, and 17.8±2.6 mmol K per kilogram. The pH was 3.5±0.7, the EC was 7.8±1.3 dS/m, the SAR was 6.9±1.8, the chemical oxygen demand was 57,200±5,500 mg O_2/L, and the density was 1.01 g/cm^3. At each application, 202 Mg/ha of whey (equivalent to a 20-mm depth) flowed by gravity through layflat irrigation tubing from a tank truck to a plot where it flooded over the plot surface. The control plots did not receive whey. Each plot of the low treatment received a single whey application on June 8. Each plot of the medium treatment received a whey application on May 19 and June 29. Each plot of the high treatment received an application on May 19, June 8, June 29, and July 20. Wheat samples collected on July 30 were analyzed for total dry matter and other samples collected on August 2 for grain yield.

In August and September 1993, tension infiltrometers (Ankeny 1992) were used to measure unconfined (three-dimensional) infiltration rates at three locations in each plot, using a slight modification of the procedure outlined by Ankeny (1992). In the bottom of irrigation furrows, where most of the whey had infiltrated, infiltration was measured without disturbing the soil surface and from low to high water potentials (−150 to −60 to −30 mm, or from −1.47 to −0.59 to −0.29 kPa). At a water supply potential of −30 mm, flow occurs through pores with diameters of 1 mm or less, at a potential of −60 mm through diameters of 0.5 mm or less, and at a potential of −150 mm through diameters of 0.2 mm or less (Marshall and Holmes 1979). Software described by Ankeny et al. (1993) was used to determine steady-state infiltration rates, and from them, to calculate unsaturated hydraulic conductivities. From these properties, we calculated both macroscopic and microscopic capillary lengths (White and Sully 1987, White et al. 1992). The use of tension infiltrometers and their applications in the study of soil structural changes induced by tillage, precipitation, and biological activity have been reviewed by White et al. (1992).

The experimental design was a randomized complete block, with four whey application treatments, three replications, and three subsamples per replication. Whey treatment means were separated using a least-squares estimation procedure with a significance probability of 5%. The relationship between treatment means and standard deviations was used to ensure that the frequency distribution of each response variable was near normal. When necessary, a common logarithmic transformation was employed to normalize a variable's distribution.

Results and Discussion

At a water potential of −30 mm, whey additions affected neither the infiltration rate nor the hydraulic conductivity (data not shown). While trends were evident in the response of both infiltration rate and hydraulic conductivity (similar to responses at lower potentials discussed in more detail below), the differences were not significant at the 5% level. Spatial variability in large pores (equivalent diameters on the order of 1 mm, Marshall and Holmes 1979) at the soil surface likely made significance difficult to detect.

Macroscopic capillary length, essentially a flow-weighted mean soil water potential (somewhat analogous to capillary rise above a water table), and microscopic capillary length, as a representative flow-weighted mean pore radius, responded as expected. At water potentials of −60 and −150 mm, macroscopic capillary lengths decreased, in general, and microscopic capillary lengths increased, in general, with whey additions. The responses, usually significant at 10%, were not significant at 5% and thus have not been presented.

The infiltration rates, at a water potential of −60 mm, into irrigation furrows decreased linearly with whey additions (fig. 1). As the whey application increased from 404 to 808 Mg/ha, the infiltration rate decreased significantly (P=0.01), from 6.26 to 2.74 μm/s. At this highest whey application rate, the soil may have been organically overloaded so that suspended solids and/or microbiological growth clogged pores at or near the soil surface, reducing infiltration rates and hydraulic conductivities (McAuliffe et al. 1982). Alternatively, this 56% decrease could have been caused by the formation of a depositional seal, possibly due to a deterioration of soil structure, along the furrow's wetted perimeter as a result of the high whey application. The stability of macropores, as well as aggregates, influences soil hydraulic properties (Murphy et al. 1993). Lehrsch et al. (1994) found the percentage of stable aggregates in largely undisturbed soil to drop from 64% to 46% as cottage cheese whey applications to a silt loam soil in southern Idaho increased from 505 to 1,010 Mg/ha.

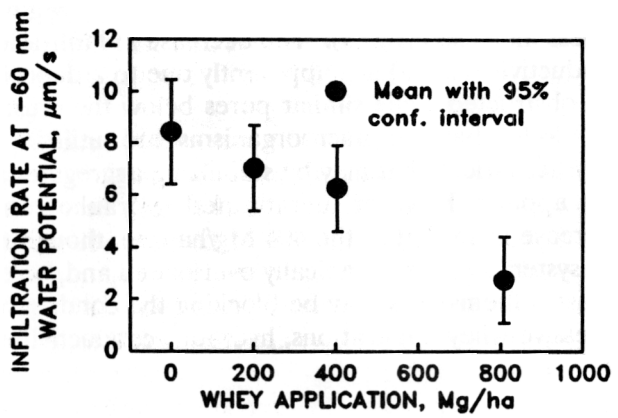

Figure 1. Infiltration rate at a water potential of −60 mm as a function of whey appplication.

In a similar manner, even at a much lower water potential of −150 mm, infiltration into the bottoms of treated furrows still decreased monotonically as more and more whey was applied (fig. 2). After the medium and high whey applications, the infiltration rates, 3.48 and 1.70 μm/s, respectively, were significantly less ($P \leq 0.015$) than the control, 6.81 μm/s. In an earlier study on a saline-sodic soil, Jones et al. (1993) found, however, that acid whey applications of up to 1,010 Mg/ha did not adversely affect ponded infiltration (water potentials ≥0).

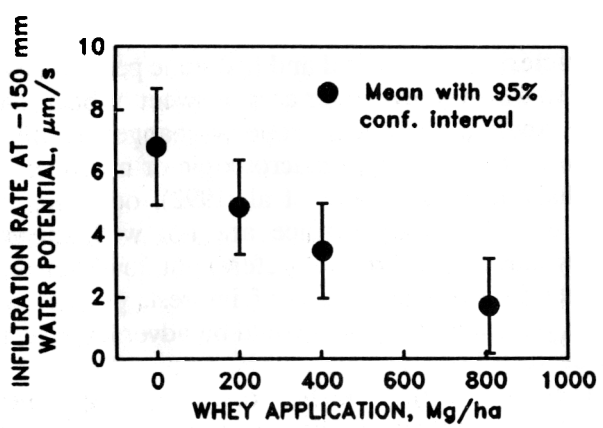

Figure 2. Infiltration rate at a water potential of −150 mm as a function of whey application.

Infiltration rates at potentials of −60 mm or less decreased with whey additions (figs. 1-2). At potentials of −60 mm or less, water flowed through pores 0.5 mm or less in diameter. This indicates that the effect of

whey additions on the pore size distribution in the soil along the furrow's wetted perimeter is consistent on pores ≤0.5 mm in diameter. That is, if the whey primarily affected pores with diameters greater than 0.2 mm, the effect of whey additions upon the infiltration rate at a water potential of −150 mm should be negligible. Such was not the case (fig. 2).

The deleterious effect of cheese whey upon furrow infiltration rates at water potentials less than −60 mm is significant from a management standpoint. If the intention is to maintain furrow infiltration, even under tension, at levels comparable to those under undisturbed conditions, cheese whey addition rates may need to be limited to rates of 404 Mg/ha or less over the summer growing season to silt loam soils in southern Idaho.

The only other hydraulic property that we studied upon which whey applications exerted a statistically significant effect was hydraulic conductivity at a water potential of −60 mm (fig. 3). The effect of whey upon unsaturated hydraulic conductivity, however, was quite different from its effect upon infiltration (figs. 1-2). As whey additions increased from 202 to 404 Mg/ha, flow through pores with equivalent diameters of 0.5 mm or less increased (fig. 3). The decrease in infiltration rate (fig. 1) and the accompanying increase in hydraulic conductivity (fig. 3) are apparently due to a depositional crust effect where pores 0.5 mm or less in the crust are obstructed while similar pores below the crust are rendered more stable. At whey application rates up to 404 Mg/ha, the microorganisms are utilizing the whey for food and, consequently, are producing polysaccharides that may be stabilizing aggregates adjacent to the flow paths below the crust. As more whey was applied, however, unsaturated hydraulic conductivity exhibited a marked (and significant, $P=0.009$) decrease from that at the 404 Mg/ha rate, though not from that of the control (fig. 3). At higher whey rates, the system may be organically overloaded and, below the depositional crust, suspended solids or the microorganisms themselves may be blocking the conducting pores, thus reducing the hydraulic conductivities. With excessive whey applications, hydraulic conductivities have decreased under certain circumstances (McAuliffe et al. 1982).

Based in part upon the findings of this study, if application rates are not too high, cheese whey may be used as an amendment to disturbed lands without affecting soil hydraulic properties excessively. Cheese whey, if added at rates less than or equal to 404 Mg/ha, may not decrease furrow infiltration rates (at potentials ≥−60 mm) markedly and may increase the hydraulic conductivity at a water potential of −60 mm.

Additional research is needed to better characterize the physical and hydraulic properties of soils to which whey, either acid or sweet or both, has been applied. Physical property changes, possibly revealed by changes in macroscopic or microscopic capillary lengths (White et al. 1992), occurring at and below the soil surface are not well known. Whey movement through preferential flow channels deeper into soil profiles is of interest, particularly since groundwater quality could be adversely affected by such flow (Kelling and Peterson 1981, Peterson et al. 1979). Further research should also examine the effects of whey incorporation by tillage on the physical and hydraulic properties of the surface soil. A likely increase in aggregate stability after tillage (Lehrsch et al. 1994) may offset the infiltration reductions with whey additions found in this study (figs. 1-2).

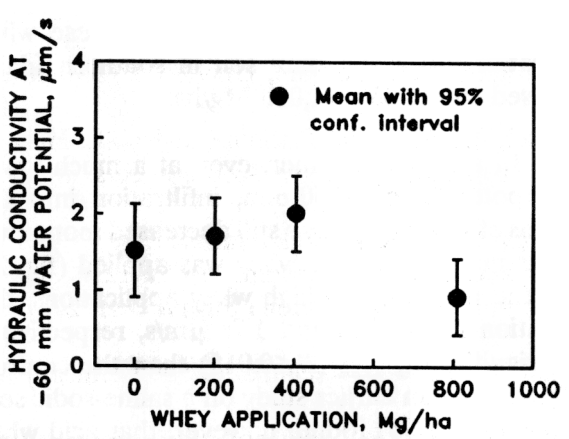

Figure 3. Hydraulic conductivity at a water potential of −60 mm as a function of whey application.

Conclusions

Cheese whey primarily affected the infiltration rate, at water potentials of −60 mm or less, through the bottoms of irrigation furrows. Infiltration rates decreased, in a linear or near linear manner, as whey applications increased from 202 to 808 Mg/ha. At water potentials of −60 and −150 mm, the infiltration rate into the 808-Mg/ha plots was significantly less than that into the control. Whey additions also affected the hydraulic conductivity at a potential of −60 mm, though in a different manner. Whey additions up to and including 404 Mg/ha increased unsaturated hydraulic conductivity. An additional doubling of the whey application rate, however, decreased the hydraulic conductivity at a potential of −60 mm.

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ESTIMATES OF NITROGEN AVAILABILITY OF POULTRY MANURE AND SEWAGE SLUDGE AMENDMENTS IN MINED PRIME FARMLANDS[1]

Qiang Zhai and Richard I. Barnhisel[2]

Abstract: The application of poultry manure and sewage sludge may speed up the return of productivity of prime farmland following surface mining, as well as for utilizing nutrients in these wastes. However, excessive application may result in nitrate contamination of ground water. This research was carried out under laboratory and field conditions to test this concern. The objective was to examine nitrogen mineralization indices used to evaluate nitrogen availability to wheat (Triticum astivum). Two field experiments were established in fall 1992 in western Kentucky. Sewage sludge was applied to both topsoil and subsoil at one site, and poultry manure was applied to only the topsoil at the second site. Three rates of organic amendments were used in these experiments. Soil available nitrogen was evaluated by both biological mineralization and chemical extraction methods. A 7-day anaerobic incubation method was well correlated with grain yield and was superior to other chemical methods in predicting nitrogen availability. Both sewage sludge and poultry manure application to the topsoil provided a high available nitrogen source for wheat growth, which resulted in a higher yield than that for the unamended control.

Additional Key Words: organic wastes, nitrogen availability indices, mineralizable nitrogen.

Introduction

Availability of nitrogen may limit revegetation of prime farmlands disturbed during surface mining. An effective reclamation method to provide nitrogen and improve soil productivity is to add organic wastes to mined soils (Barnhisel and Zhai 1992). A portion of nitrogen in organic wastes may be mineralized during the cropping season by microbial processes. This mineralization provides a certain amount of nitrogen as NH_4^+ and NO_3^-, which are easily accessible to plants. Organic waste application rates are generally based on the crop yield goals and the nitrogen availability provided by these wastes. It would be useful to evaluate nitrogen mineralization indices to guide waste loading rates because excess application would cause adverse environmental pollution through excessive nitrogen loss to water systems.

Numerous nitrogen mineralization indices, which include both chemical and biological methods for assessing nitrogen availability, have been evaluated. Examples of such indices include HCl-extractable nitrogen, acid $KMnO_4$-oxidizable nitrogen, boiling water-extractable nitrogen, and soil Kjeldahl nitrogen. These indices are derived through relatively inexpensive and rapid procedures. However, they do not simulate real nitrogen mineralization processes mediated by microorganisms. If these indices are used to predict crop production, they must be calibrated under field conditions. Biological incubation indices that simulate natural microbial processes are generally considered to be the most reliable laboratory methods for assessing plant-available nitrogen (Stanford 1982). A 7- to 14-day anaerobic incubation index is considered to be the most accurate method currently available for assessing biological nitrogen availability. It is a simple, rapid method and well-correlated with other long-term aerobic incubations (Keeney 1982).

Little is known of the reliability of these indices under field conditions, particularly for organic wastes applied to surface-mined soils. The objective here is to determine if these biological and chemical nitrogen

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[2]Qiang Zhai, Graduate Assistant, Richard I. Barnhisel, Professor, Department of Agronomy, University of Kentucky, Lexington, KY, USA.

mineralization indices can be used to predict nitrogen availability of poultry manure and sewage sludge in surface-mined prime farmland planted to wheat.

Materials and Methods

Field Experiments

The poultry manure and sewage sludge sites are located at two western Kentucky sites in Henderson and Muhlenberg counties, respectively. Two prime farmland soils were reconstructed in summer 1992. Sewage sludge was applied at the rate of 22.4 Mg/ha and was incorporated in the upper 20 cm of subsoil prior to the replacement of the topsoil. The topsoil received an additional application of sewage sludge at 0, 11.2, and 22.4 Mg/ha, which was also incorporated. Poultry manure was incorporated only in topsoil at the rates of 0, 33.6, 67.2 Mg/ha. Following organic waste incorporation, soil samples were collected and soft red winter wheat was planted at both sites. Above-ground plant tissue was sampled during anthesis period, and grain samples were also collected at harvest. Grain yield was measured with a MF8 plot combine. All plant samples were dried at 65°C and ground to pass a 0.425-mm screen, using a stainless steel Wiley Mill, and total nitrogen was determined by a standard micro-Kjeldahl procedure. Soil samples were air dried and ground to pass a 2-mm sieve. The pH, total nitrogen, NH_4-N, NO_3-N, and organic matter were determined by standard laboratory methods (Page 1982). The soil microbial mass was determined by the chloroform fumigation method (Parkinson and Paul 1982).

Biological Nitrogen Mineralization Method

Anaerobically mineralized NH_4-N was determined by a slight modification of the procedure described by Keeney (1982). Ten grams of fresh soil was incubated with 25 mL of distilled water for 7 days in a stoppered 50-mL flask at 40°C. After incubation, 13 mL of 1M KCl was added, and the flask was mechanically agitated for 10 s and allowed to stand for 30 min. The flask was reagitated for 10 s, and the mixture was vacuum filtered through Whatman #42 paper. The incubation flask was rinsed with 13 mL of distilled water, and the rinse was filtered. The filtrate was measured for NH_4^+-N by a standard micro-Kjeldahl procedure. Net nitrogen mineralized was the difference of this measurement and inorganic nitrogen in the sample.

Chemical Nitrogen Extraction Methods

1. 6M HCl Method. Release of nitrogen in HCl was determined by heating a mixture of 5 g of soil and 15 mL of 6M HCl (1:3) in sealed glass test tubes for 12 h at 100°C (Bremner 1949). The extract was analyzed for total nitrogen by the same Kjeldahl procedure.

2. $KMnO_4$ Method. The acid $KMnO_4$ oxidation method was essentially that of Stanford and Smith (1972). Two hundred milliliters of 0.5M H_2SO_4 was added to plastic tubes containing 8-g samples of air-dried soils. The tubes were sealed and shaken for 1 h and centrifuged at 3800 xg for 10 min, and the acid extracts were decanted and discarded. The residue was then resuspended with 200 mL of 0.1M $KMnO_4$ in 0.5M H_2SO_4, shaken again, centrifuged, and the supernatant solutions were decanted and analyzed for NH_4-N by the Kjeldahl procedure.

3. Boiling Soil Method. Nitrogen is extracted in boiling water was used according to Keeney (1982). Duplicate 10-g air-dried soil samples placed in glass tubes were washed twice with 25-mL increments of 0.01M $CaCl_2$ to remove mineral nitrogen initially present. The soil residues were then boiled for 16 h in 25 mL of 0.01M $CaCl_2$. The extracts were recovered by centrifuging, and then the soil was washed twice with 25-mL increments of 0.01M $CaCl_2$. An aliquot of the combined extract and washing was analyzed for NH_4-N by the same Kjeldahl procedure.

Statistical Analyses

Standard analyses of variance and of simple and multiple correlation coefficients were calculated to

indicate the relationships among wheat grain yield, above-ground tissue nitrogen concentration, mineralizable nitrogen, and chemically extractable nitrogen by the Statistical Analysis System (SAS).

Results and Discussion

Soil Properties

Selected soil properties after poultry manure and sewage sludge application are shown in table 1. Soil pH and microbial biomass were not largely altered, but total nitrogen, organic matter, and inorganic nitrogen increased as the loading rates increased.

Table 1. Effect of poultry manure and sewage sludge amendments on selected soil properties.

Rate, Mg/ha	pH	TN[1], g/kg	OM, g/kg	IN, mg/kg	MB, mg/kg
Poultry manure:					
0	5.8	0.22	4.6	15	91
33.6	6.1	0.55	9.6	66	110
67.2	6.3	0.98	16.7	72	115
Sewage Sludge:					
0	5.8	0.24	4.5	18	87
11.2	6.0	0.30	6.1	31	104
22.4	6.2	0.66	7.4	53	108

[1]TN, total nitrogen; OM, soil organic matter; IN, inorganic nitrogen; MB, microbial biomass (C).

Crop Response

Poultry manure and sewage sludge application significantly increased wheat tissue nitrogen concentration, grain yield, grain nitrogen concentration, and above-ground tissue dry matter compared with the control (table 2), but there was no significant difference between the two high loading rates. Higher yields from soils amended with poultry manure were closely related to more available nitrogen by mineralization from the poultry manure than from the sewage sludge treatments.

Biological Nitrogen Mineralization Index

Biological mineralized nitrogen is given in table 3. Poultry manure treatments released more inorganic nitrogen than did sewage sludge. High mineralization rates of poultry manure may be related to higher loading rates. However, the research indicated that the high mineralization rates resulted mostly from the more easily mineralized portion in poultry manure (Zhai and Barnhisel 1993). The amount of nitrogen mineralized increased as loading rates increased for both poultry manure and sewage sludge. Evaluation of biological mineralization and chemical extraction indices is based on the relative degree of correlation with wheat grain yield. Nitrogen biologically mineralized was well correlated with grain yield ($r=0.91^*$ for poultry manure; $r=0.92^*$ for sewage sludge).

Table 2. Effect of soil amended with poultry manure and sewage sludge on wheat response.

Rate, Mg/ha	Yield, Mg/kg	Tissue N, g/kg	Grain N, g/kg	Dry matter, g/kg
Poultry manure:				
0	2.75 a[1]	0.86 a	1.82 a	1.55 a
33.6	4.88 b	1.14 b	2.03 b	2.03 b
67.2	5.07 b	1.20 b	2.06 b	2.18 b
Sewage Sludge:				
0	3.31 a	0.86 a	1.51 a	1.19 a
11.2	4.24 b	1.04 b	1.74 b	1.42 b
22.4	4.65 b	1.10 b	1.96 b	1.69 b

[1]Comparison within column for same treatment only ($p<0.05$).

Chemical Extraction Methods

The 6M HCl method extracted the most nitrogen, followed by the 0.1M $KMnO_4$ and boiling soil methods (table 3). The 6M HCl method probably extracts greater amounts of nitrogen than are readily biologically available. The amount of nitrogen released by 6M HCl was not significantly correlated with wheat yield for sewage sludge study ($r=0.65$). Although it was correlated significantly with poultry manure treatment ($r=0.78^*$), it did not represent the actual amounts of nitrogen that crop used during growth season. This was also observed in another study by Serna and Pomares (1992). The 0.1M $KMnO_4$ method had intermediate extractibility compared with the 6M HCl and boiling soil methods. The amount of nitrogen extracted by this method was only about one-half that extracted by 6M HCl, but it was about two times higher than that extracted by boiling soil. Significant correlation between grain yield and N extracted by $KMnO_4$ was observed for poultry manure treatment ($r=0.76^*$), but not for sewage sludge study ($r=0.51$). Significant correlationships were obtained between boiling extractable nitrogen and grain yield for both poultry manure ($r=0.89^*$) and sewage sludge ($r=0.75^*$) treatments. Wheat yields from poultry manure treated plots were significantly correlated with total nitrogen, but were not sewage sludge. Therefore, we concluded that extracting methods employed to predict nitrogen availability under field condition are different for organic wastes applied to soil. Some extracting methods are suitable for poultry manure, but not for sewage sludge, although O'Keefe et al. (1986) evaluated nitrogen availability for amended sludge compost and found that chemical methods were well correlated with soil available nitrogen.

Multiple Correlation With Wheat Grain

The multiple regression analysis to select the good prediction of wheat yield is shown in table 4. Coefficients from the linear regression analysis between wheat grain yield (dependent variable) and biological and chemical nitrogen availability indices (independent variables) were improved by multiple regression. Biological mineralizable nitrogen was the major contribution to correlation coefficients both for poultry manure and sewage sludge treatments. When 6M HCl extractable nitrogen was added to the model, correlation coefficients improved markedly only for sewage sludge treatment. When $KMnO_4$ extractable nitrogen was added to the model, The improved relationship only attributed to poultry manure treatment. Correlation coefficients were not improved much when boiling extractable nitrogen combined in, although coefficients were slightly higher than those obtained in previous multiple correlation equations. Addition of soil total nitrogen parameter

into the model significantly improved correlation coefficients both for poultry manure and sewage sludge treatments. This could be explained on the bases that the nitrogen in these organic wastes can be released fairly fast compared with soil organic matter and result in meeting the wheat nitrogen requirement for entire growing season through nitrogen mineralization process.

From above evaluation of correlation between wheat yield and nitrogen availability indices, we conclude that nitrogen mineralized by a 7-day anaerobic incubation method is the best single index to predict nitrogen availability. Chemical extraction methods are not all reliable for two different organic waste treatments. They may be selectively used as the reference when biological mineralization method is used. Anaerobic mineralized nitrogen plus total soil nitrogen are well correlated with wheat yield and can be used as important parameters to predict wheat yield for mined soils amended with poultry manure and sewage sludge.

Table 3. Means of nitrogen mineralized or extracted by several nitrogen availability index methods.

Rate, Mg/ha	Biological N, mg/kg	HCl N, mg/kg	$KMnO_4$ N, mg/kg	Boiling N, mg/kg
Poultry manure:				
0	17	150	58	25
33.6	157	445	210	92
67.2	288	755	428	156
Sewage Sludge:				
0	15	129	64	26
11.2	94	244	168	75
22.4	160	442	192	124

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Table 4. Stepwise multiple correlation of wheat grain yield and nitrogen availability indices.

	Equations	R^2
Poultry manure	$Y^1 = 2.89 + 0.009X1$	0.83*
	$Y = 2.95 + 0.01X1 - 0.0005X2$	0.83*
	$Y = 2.79 + 0.01X1 + 0.002X2 - 0.005X3$	0.86*
	$Y = 2.63 + 0.007X1 + 0.002X2 - 0.006X3 + 0.007X4$	0.86*
	$Y = 2.73 + 0.007X1 + 0.005X2 - 0.005X3 + 0.01X4 - 2.64X5$	0.88*
Sewage sludge	$Y = 3.21 + 0.009X1$	0.85*
	$Y = 3.44 + 0.02X1 - 0.0027X2$	0.94*
	$Y = 3.46 + 0.02X1 - 0.002X2 - 0.003X3$	0.94*
	$Y = 3.49 + 0.017X1 - 0.003X2 + 0.0004X3 - 0.004X4$	0.95*
	$Y = 3.54 + 0.016X1 - 0.0001X2 - 0.0002X3 - 0.002X4 - 1.87X5$	0.96*

[1]Y, wheat grain yield; X1, biological mineralized nitrogen; X2, 6M HCl extracted nitrogen; X3, 0.1M $KMnO_4$ extracted nitrogen; X4, boiling water extracted nitrogen; X5, soil total nitrogen.

DEFINITION OF REHABILITATION STRATEGIES FOR PRE-STRIP TERTIARY SPOIL AT COAL MINES IN CENTRAL QUEENSLAND

L. C. Bell[1], D. R. Mulligan[1], R. J. Mitchell[1], M. W. Philp[1], and P. A. Roe[2]. ((1) Department of Agriculture and Centre for Mined Land Rehabilitation, The University of Queensland, Brisbane, Qld, Australia, (2) BHP Australia Coal Limited, Brisbane, Qld, Australia).

In the open-cut coal mines of central Queensland, the conventional dragline removal of overburden results in a spoil surface dominated by Permian sediments. In recent years, however, a number of mines have changed their coal exposure methods to include a pre-stripping operation to create a bench for the dragline up to 50 m below the surface. This approach has allowed economic coal recovery from greater depths, but, since this pre-strip material is being deposited on existing spoil piles, it has resulted in a major change in the type of spoil material on the surface. Pre-strip spoil is composed of unconsolidated high clay Tertiary sediments and is typically saline and sodic in nature, has a high tendency to crust, and is extremely deficient in N and P. In view of the shortage of quality topsoil at many of the mines for capping this spoil, it was expected that severe problems for plant establishment may exist. The aims of this project were, firstly, to define the characteristics of this pre-strip material that could cause problems for the establishment and growth of both introduced pasture grasses and native tree species, and secondly, to explore possible spoil amendments which could ameliorate some or all of these limitations. Analyses on the range of spoils included pH (3.8 to 9.9), EC (0.1 to 9.4 dS/m, 1:5 water) and sodicity (ESPs 24% to 59%). Glasshouse trials defined optimum fertilizer rates and showed gypsum rates exceeding 8 mt/ha to alleviate the surface crusting. Straw mulch also reduced the crust strength and was an effective means of improving vegetative establishment. A complementary field trial was established at Saraji mine in April 1991, and after 4 months, treatments which involved a 30 cm capping of soil on the spoil resulted in the highest densities of tree seedlings. Monitoring of numerous vegetative and substrate parameters continues. More recent initiatives are further exploring the use of organic mulches and ponding basins to reduce the surface crusting problems and improve infiltration, thus assisting establishment in the short term, and growth and sustainability of the communities in the longer term.

ESTABLISHMENT OF NATIVE SPECIES ON AMENDED AND UNAMENDED COAL MINE SPOIL IN CENTRAL QUEENSLAND

L. C. Bell[1], D. R. Mulligan[1], M. S. Orr[1], and P. Johnstone[2]. ((1) Department of Agriculture and Centre for Mined Land Rehabilitation, The University of Queensland, Brisbane, Qld, Australia, (2) Curragh Queensland Mining Ltd, Blackwater, Qld, Australia).

At the Curragh open-cut coal mine in central Queensland, tree and shrub growth on amended and unamended spoil was assessed over a 5-year period. The most successful media in terms of seedling densities were bare spoil and spoil covered by a thin layer of coarse coal reject. Where soil was replaced, very few native species established due to the intense competition from exotic grasses. Given this poor establishment on soil, the question of how well an ecosystem would develop on non-soil media was addressed by assessing the ability of each of the media to support sustainable communities in the longer term. Early results from direct-seeding tree and shrub species onto non-soil media have been quite encouraging, but analysis of the surface horizons has shown that after several years, there are still some characteristics which could either limit the re-establishment, and/or prevent the sustainability, of new communities. Due primarily to the high sulfate levels originally in the coal reject, the surface salinity in these treatments was much higher than in the uncapped spoil. Substrate analyses have shown also that there was a much lower concentration of mineralizable N in the spoil and coal reject than in the soil, revealing that the potential of these non-soil substrates to continue supplying nitrogen to the system is greatly reduced. Furthermore, the concentration of microbial biomass carbon was significantly lower in these media, and the percentage ground cover after 5 years was generally <10% (cf. 70% in the soil treatments). Further research on the longer term implications of using coarse coal reject as a spoil capping is underway. In the interim, the strategies being employed to manage the problems of poor tree establishment on soil are being addressed; firstly, by creating a landscape whereon strips of soil on spoil (into which only grass species are sown) are alternated with strips of bare spoil or coal reject on spoil (into which the tree seeds are sown), and secondly, by examining the use of herbicides and/or various cultivation practices, to control the competition afforded by the more aggressive weed and grass species.

REHABILITATION STRATEGIES AND ECOSYSTEM DEVELOPMENT FOLLOWING BAUXITE MINING AT WEIPA, NORTH QUEENSLAND

L. C. Bell (1), D. R. Mulligan (1), G. D. Schwenke (1), S. C. Roberts (1), and N. Dahl (2). ((1) The University of Queensland, Brisbane, Qld, Australia, (2) Comalco Aluminium Ltd., Weipa, Qld, Australia).

Mining of bauxite at Weipa, north Queensland, commenced in 1963, with the first land being rehabilitated in 1966. Approximately 6000 ha of mined land have been rehabilitated, at a current rate of about 280 ha/yr. While the initial program concentrated on commercial forestry, pastures and crops, the major emphasis for more than a decade has been on the establishment of a self-sustaining community of native species using direct-seeding. The objectives of this project were to assess the success of the rehabilitation strategies and to provide comparative information on alternative practices that may enhance long-term ecological stability. The first object was addressed by surveying the soils and vegetation of 50 rehabilitates sites (occupying 412 ha) ranging in age from 1 to 22 years. Unmined eucalypti woodland communities were also assessed. Organic C levels in the surface soil of rehabilitated areas have increased over time but are still below that of the undisturbed soil. Amongst the soil treatment effects on the status of the revegetation, soils stripped just prior to the wet season, non-stockpiled soils, and ploughing, all generally resulted in higher tree and shrub densities. A parallel phase of the project involved a trial in which the effects of various soil handling and seed-bed preparation techniques were examined in more detail. Three freshly-replaced soil treatments (mixed surface and subsoil, surface soil overlying subsoil, subsoil alone) and a stockpiled soil treatment were used. Other variables, such as stripping time, ripping time, and ploughing, were incorporated into the trial design. Periodic measurements of organic C, N, mineralizable N and microbial C and N were undertaken, and all analyses showed the superior fertility status of the soils which had the original topsoil returned to the surface. The trial treatments affected seedling establishment and growth to varying degrees, but in the first year after sowing, the differences were due mainly to the treatment effects on grass competition. Monitoring of the trial is continuing, and the results will assist in improving not only the quality and consistency of the revegetation, but, in the longer term, the sustainability of the reconstructed ecosystems as a whole.

LIMITATIONS OF NATURAL RECOLONISATION ON FIVE MINE SITES IN BRITISH COLUMBIA, CANADA

John Binns. (University of British Columbia, Vancouver, B.C., Canada).

Since provincial legislation was passed in B.C. in 1969, requiring active reclamation of all mine sites, little long-term study has been made of natural recolonisation, either on pre-1969 sites where no active reclamation was done, or on sites which have been actively reclaimed. The present study selected 5 sites which had all been studied scientifically during the period 1973-1978 to allow a minimum of 15 years duration over which to measure changes in vegetation community. In addition the sites were all in different biogeoclimatic locations within B.C. Three of the sites ceased production before 1969 and recolonisation was entirely natural, whilst two sites ceased production post-1969 and have been actively reclaimed. Samples selected within sites varied according to type of surface material. For each mine site, as many sample points as possible from the previous survey were relocated and measurements of surface material, nutrient analysis, and visual estimates of plant species and population were repeated. Effectively, this allowed a duration of 15-19 years between surveys of vegetation community. Results show a marked difference between mine sites, as expected. However, within site differences in limitations to recolonisation reflect competitive as well as biogeoclimatic variations. The work was carried out entirely by the author as part of an M.Sc. research project studying the limitations to natural regeneration of vegetation at past-producing mines.

SEASONAL CHANGES IN THE INVERTEBRATE FAUNA OF A RESTORATION BLASTED QUARRY FACE IN DERBYSHIRE, UK

W. Rod Cullen and C. Philip Wheater (Dept. of Environmental and Geographical Sciences, The Manchester Metropolitan University, Manchester M1 5GD, UK).

Restoration blasting attempts, by controlled blasting of modern quarry faces, to replicate daleside landform sequences and to establish self-sustaining ecosystems, similar to those on natural dalesides, within a much shorter period of time than with natural colonisation and development. Invertebrates are important components of such ecosystems and although initially the vegetation may be assisted through seeding, invertebrate colonisation and establishment is left to chance. Invertebrate colonisation does occur and the communities which develop are similar to, if initially more impoverished than, those on the natural daleside models. It is important that such communities not only develop through efficient colonisation, but are sustained over time. This may depend on suitable reservoirs of species being in close proximity to the restored areas. During the early stages, community disharmony may occur, possibly being more pronounced at some times of the year than at others as individual colonisation and localised extinction events occur. In preliminary surveys it was found that many invertebrates exhibit a summer diapause. In such habitats, sparse vegetation may reduce the buffering of external temperature on the ground surface possibly lowering the availability of surface active prey. This study examines the seasonal changes of surface active invertebrates on a natural daleside and on restoration blasts which had been produced at different times. These are discussed in relation to site management, time of year, vegetation present, and the availability of food.

STABILIZATION OF COAL MINING GOB PILES WITH AMERICAN BEACHGRASS

Samuel Depue (1) and Robert Glennon (2). ((1) Soil Conservation Service, Princeton, WV, USA; (2) Soil Conservation Service, Harrisburg, PA, USA)

American beachgrass was used to stabilize a coal mine gob pile in southern West Virginia. The gob pile was located in a steep valley that would have been difficult to stabilize using conventional methods of grading, topsoiling, and seeding due to limited space for spreading and limited sources of topsoil. The beachgrass was planted without grading or topsoiling on a 50 percent slope at two culms per hole on a 0.6 meter spacing with 30 grams of 10-10-10 fertilizer with a slow release nitrogen source under each plant. The cost of planting was $9375 per hectare, one-eighth of the estimated $75,000 to stabilize the site with conventional methods. Eighty percent of the plants survived the first year. Each plant produced 25 culms covering a 0.3 meter spread. Herbaceous weeds and light seeded woody pioneer species are beginning to colonize the site as they establish in the protected environment around the beachgrass plants.

COMPARATIVE STUDY OF GERMAN AND GREEK LIGNITE MINE RECLAMATION

Norman L. Dietrich. (Dept. of Landscape Architecture, Iowa State University, Ames, IA, USA)

The reclamation planning of three European lignite mining districts are compared: amount and extent of planning, integration with mining operations, and results. The districts are: Rheinbraun (Cologne), Leipzig-Borna ("East" Germany), and Megalopolis (Greece). Lignite mines were visited and mining and reclamation personnel interviewed. The Rheinbraun mines have the most thorough reclamation operations. The integrated mine and reclamation operations are world class in size, scope, and detail of reclamation. A comprehensive landscape and reclamation plan is required in the mine permitting process. The Leipzig-Borna district is the second largest of the districts, studied little pre-mining planning of the post-mining landscape or land uses was evident. Reclamation is not closely integrated with the mining and typically occurs many years after the mining. Reduced lignite production since German reunification has left vast areas of disturbed land with little mining; and no funding for the reclamation of the large areas of mined land reclamation. The Greek Megalopolis mines have mine operations plan, but with no integrated reclamation planning. The initial spoil pile was reclaimed according to the original "German" mining plan. No pits have been reclaimed, and spoil areas are revegetated sporadically. The Rheinbraun mining operations Cologne which include a post mining landscape/land use plan have integrated and timely reclamation operations. The other two mining operations, which do not have a comprehensive and detailed reclamation and landscape/land use plans, do not integrate reclamation operations with the mining operations. The results are large areas of mined land unreclaimed for many years.

FISH AND WILDLIFE EVALUATION OF WETLANDS CREATED BY MINING ACTIVITIES

Steven R. Kepler, Fisheries Biologist, Pennsylvania Fish and Boat Commission, Bellefonte, PA
Thomas M. Sabolcik, Wildlife Impact Review Coordinator, Pennsylvania Game Commission, Harrisburg, PA

Four lacustrine wetland systems created by mining activities, either by sand and gravel or bituminous coal removal, were sampled to determine the functional values of the fish and wildlife habitat. Most of the sampled sites were remnants of pre-act (1977) mining where minimal, if any, reclamation techniques were used. Natural succession within these impoundments have created ecosystems different in quality based on the availability of suitable habitat. Sampling techniques used to evaluate the wetland systems included initial water quality analyses, and aquatic habitat mapping using visual observations, LCD recorder, and computerized chart recorder. Fish populations were sampled using a boat mounted D.C. electrofishing unit with game fish being collected, weighed and measured and population estimates calculated as catch per unit effort (CPUE). Wildlife utilization of each site was conducted during the spring nesting season. Each site was surveyed for species utilization, nest searches determined whether nesting occurred and nesting success was noted. Wildlife utilization was determined by observation, tracks, calls, scat, etc. Whenever possible sites were monitored during the fall migration period to determine whether the sites were being utilized by migratory waterfowl. Wetland vegetative studies were also conducted at each site. Wetland species were identified and concentrations and dispersion of each wetland species were noted. Each sampled wetland data set is presented separately because of the variabilities between sampled sites based on the geology, reclamation status, and habitat.

The importance of these lacustrine wetlands created by mining tends to be a controversial issue within Pennsylvania through permit applications for active mining, remining previously affected areas and/or reclamation activities at abandoned mine sites. This study helps to document the importance of these wetland resources which were created by past mining activities and the need for protection or replacement.

FLY ASH ERODIBILITY

Gary A. Lehrsch (1) and Dale E. Baker (2). ((1) USDA-Agricultural Research Service, Kimberly, ID, USA, (2) Land Management Decisions, Inc., State College, PA, USA).

Fly ashes are the fine ash particles entrained in the flue gases from coal-burning electric power-generating stations. In the northeastern United States, fly ash is removed from the stack gases and commonly trucked to landfills for disposal. Unfortunately, the cover soil and especially the underlying fly ash of these landfills are extremely susceptible to erosion by water. The goal of this study was to estimate fly ash erodibility by collecting sediment eroded by natural rainfall in the field from standard erosion plots (1.8 m wide and 22.1 m long on a 9% slope of exposed fly ash). The universal soil loss equation was used with direct measurements on-site to obtain estimates of the erodibility factor (K factor) for fly ash. These estimates were then compared to an estimate obtained using a soil erodibility nomograph. The K factors measured in the field ranged from 0.11 to 0.13 Mg ha h (ha MJ mm)$^{-1}$ and averaged 0.122 Mg ha h (ha MJ mm)$^{-1}$ while the nomograph estimate was 0.108 Mg ha h (ha MJ mm)$^{-1}$. A K factor of 0.122 Mg ha h (ha MJ mm)$^{-1}$ was recommended for erosion control. With this K factor and the universal soil loss equation, the surface topography of vegetated fly ash disposal areas was designed to limit soil loss to a tolerance level of 4.5 Mg (ha y)$^{-1}$. Using the design K factor, erosion from vegetated demonstration plots 73 m long at a 15% slope was controlled.

PLANTING TESTS ON RECLAIMED LAND FILLED WITH COAL WASTES

Mingsong Liu. (State Land Administration, Department of Land Use Planning, Beijing, China)

This paper discusses the research of planting tests on a reclaimed subsidence trough filled with coal wastes in the Keiluan coal mining area in China. Analysis of the physical and chemical properties of coal wastes indicated that they were strongly acidic. Thus, the key of planting vegetation was the neutralization of these acidic conditions. Based on the planting tests, it was found that neutralization of the acidity required liming at a rate of 150 mt/ha. The required thickness of soil cover was found to be more than 500 mm for planting row crops, and 300 mm for planting trees. The available species of row crops were soybean, mung bean, and sweet potato. The suitable species of trees included the locust, tree of heaven, and poplar.

RESTORATION OF A CEMENT KILN DUST DEPOSIT WITH UNSCREENED COMPOSTED SEWAGE SLUDGE

John B. Lizak and Steve O'Reilly. (Lizak Geoscience & Engineering, Inc., Evansville, IN, USA).

A five hectare cement kiln dust (CKD) pile adjoining an abandoned limestone quarry in York, County, PA was reclaimed in 1992. The unusual physical and chemical characteristics of the CKD and the lack of soil precluded typical restoration approaches. Studies showed that the CKD could be most cost-effectively reclaimed by using unscreened composted sewage sludge (CSS) supplied by the Philadelphia Water Department. The CKD and the CSS were rigorously tested for toxicity and nutrients. Test plots were used to determine optimum CSS distribution rates, seedbed preparation methods, seed mixtures, and seeding, fertilizer and mulch rates. The CKD pile was graded, erosion controls were constructed, and ninety 20 Ton truckloads of CSS were delivered in June, July and August. Several unique techniques were developed to mix and distribute the CSS to simulate a natural soil horizon. The seedbed was prepared and then planted with an innovative seed mix in late September. It was then amended with fertilizer and mulch. The CKD site now contains a diverse, effective, and permanent cover capable of self generation and succession greater in extent than the natural vegetation in the area. Erosion has been nominal and the site is beginning to support diverse fauna.

BIOSOLID SOIL AMENDMENTS IN ARID MINELAND RECLAMATION: THE BUTCHER BOY MINE PILOT STUDY

David Loomis, AICP. (U.S. Bureau of Land Management, Carson City, NV, USA).

In order to achieve sustainable growth, communities in the arid western U.S. must develop new techniques to extract natural resources from public lands while protecting their unique desert environment. One promising technique is suggested by an on-going mine reclamation research project using biosolids, the solid residue of municipal wastewater treatment. The Butcher Boy Mine Pilot Study was developed because reclamation of degraded minelands in the arid west has met with only limited success due to the lack of suitable topsoil material. At the same time, communities have been searching for alternatives to disposing biosolids in municipal landfills. The Pilot Study is located near Wadsworth, 60 km north-east of Reno, Nevada. It's objectives are to determine the regulatory feasibility of applying biosolids for reclamation of degraded arid minelands and the long term effectiveness of biosolids in enhancing revegetation of those minelands. Regulatory barriers were overcome and all necessary permits were obtained in October, 1992. The next month, five .4 ha test plots were constructed on regraded tailings ponds at the Butcher Boy Mine. The Truckee Meadows Water Reclamation Facility provided the biosolids. They were applied at 33 to 67 t/ha on four of the plots. The remaining plot served as the control with no biosolids applied. After plowing, the plots were all seeded with grasses and shrubs. First year results are very encouraging. Vegetative cover amounted to as much as 51% on the treated plots compared to only 6% on the control plot. Land application of biosolids in arid mineland reclamation is feasible. Further study on its long term effectiveness is clearly warranted.

A REVEGETATION STRATEGY FOR TACONITE IRON ORE TAILING USING VA MYCORRHIZAE AND NATIVE GRASSES

Robert K. Noyd (1), Frank L. Pfleger (1), Michael P. Russelle (1), and Michael R. Norland (2). ((1) University of Minnesota, St. Paul, MN, USA; (2) U.S. Bureau of Mines, Minneapolis, MN, USA).

Interactions between potentially successful plant species and vesicular-arbuscular mycorrhizal (VAM) fungi influence successful dynamics and long-term sustainability of revegetated ecosystems. Revegetation of coarse taconite iron ore tailing is difficult because tailing has an alkaline pH (8.2), low water retention (1 % by wt.), and low levels of available P (1-3 mg/kg), N (< 1 mg/kg), and organic matter (0 %). Current revegetation practices include planting a mixture of agronomic species and fertilizing annually with 448 kg/ha of diammonium phosphate (18-46-0). The University of Minnesota and the U.S. Bureau of Mines' goal is to establish self-sustaining plant communities with a minimum of fertilization and labor using native grass species. A revegetation strategy was formulated based on information from plant selection experiments in the field and P response curves in the greenhouse. The cool-season grass, Canada wild rye (Elymus canadensis L.), was selected to initially stabilize the site, followed by the warm-season grass big bluestem (Andropogon gerardi Vitm.). This seeding schedule was based on their site suitability and growth characteristics in association with a locally-occurring species of VAM mycorrhizal fungi, Glomus claroideum Schenck & Smith. Canada wild rye shoot yields are less responsive to VAM, but the plant's abundant root system is highly colonized by these fungi and thus it increases the inoculum in the tailing for the more VAM-responsive big bluestem. Phosphorus response curves indicate that inoculation with VAM fungi increases shoot mass and P uptake and maximum shoot mass can be obtained with about 190 kg/ha of KH_2PO_4 fertilizer. Natural immigration of VAM fungal propagules is inadequate and an outside source of inoculum is necessary. The fine tailing basins and their restored vegetation offer a potential source of Glomus claroideum and other adapted VAM fungal species that live under similar chemical stresses as those posed by coarse tailing. It is expected that natural dispersal of plant seeds will slowly diversify and further stabilize the established plant populations.

HIGH GYPSUM COAL COMBUSTION BY-PRODUCT USE IN AMELIORATION OF ACID SOILS

K. Dale Ritchey, Ralph B. Clark, Sheila K. Zeto, and Virupax C. Baligar. (Appalachian Soil and Water Conservation Research Laboratory, Agricultural Research Service-USDA, Beckley, WV 25802-0867, USA)

Clean Air legislation has resulted in coal-combustion power plant production of large volumes of high-calcium by-products. A beneficial alternative to disposal would be utilization as an amendment for acid soils. One promising by-product for agricultural use is wallboard quality gypsum (calcium sulfate). Many of the power plants which burn high S coal are located in the eastern United States, where surface and subsurface soil acidity is sometimes a constraint to productivity. High acidity in or below the plow layer often precludes plant deep root penetration, and yields are reduced during droughts because roots are not able to exploit water stored in subsurface layers. Because of its solubility, surface-applied gypsum is able to leach into soil profiles and significantly reduce effects of toxic levels of aluminum. In Appalachian soils, gypsum has been shown to increase the Ca status below the zone of application, reduce aluminum saturation, increase rooting density and increase the amount of water extracted by orchardgrass roots. In more oxidic soils where long-term field experiments have been conducted, field crops have been shown to have deeper and more efficient root systems, improved yields and greater uptake of water and nitrate-nitrogen.

RECLAMATION OF OHIO ABANDONED MINED LANDS USING SOIL AMENDMENTS

Paul Sutton. (The Ohio State University, Columbus, OH, USA)

A study was initiated on an abandoned mine site in southeastern Ohio in 1979 to determine the long-term effects of composted papermill sludge, power plant fly ash, sewage sludge and topsoil on the chemical properties of acidic spoil (pH 3.4) and vegetative cover. The minesoil is classified as an Enoch soil (loamy-skeletal, siliceous, acid, mesic Typic Udorthents). The soil pH values immediately after the treatments had been applied and after eleven growing seasons were 3.4 and 3.8, respectively, for the control; 4.9 and 5.2, respectively, 67 Mg/ha; 5.2 and 5.2, respectively, 90 Mg/ha; 5.4 and 6.2, respectively, 112 Mg/ha of papermill sludge; 3.9 and 4.9, respectively, 448 Mg/ha of fly ash; 5.4 and 6.3, respectively, 224 Mg/ha sewage sludge; and 7.3 and 7.0, respectively, for the 20 cm cover of limed topsoil. The application of 4.5 Mg/ha of agricultural limestone to one-half of the papermill sludge, fly ash and sewage sludge amended plots had little influence on soil pH after 11 years (mean of amended plots 5.9 and mean of unamended plots 5.6). The mean pH of the spoil below the treated zone ranged from 3.8 to 4.5. Except for the control, a 100% vegetative cover was established in 1979 as a result of a grass-legume seeding. Yields have declined and in 1990 the mean dry matter collected from the limed part of the plots was 79% of that collected in 1980 and 57% from the unlimed part. This is due primarily to a lack of nitrogen because there are too few legumes in the stand and no fertilizer nitrogen is being applied. With the addition of lime and fertilizer forages could be produced. Without any management of the plots, however, the vegetative cover will largely revert to blackberry briars (*Rubus* species). If the objective of reclamation is soil protection, then primary indications are that in either case a sufficient vegetative cover will be maintained to greatly reduce soil erosion.

GEOSTATISTICAL MODELING OF COAL MINESOILS: A THEORETICAL PERSPECTIVE

William J. Tarutis, Jr. (Wilkes University, Wilkes-Barre, PA, USA).

Classical application of statistics to soil science assumes that data to be analyzed are independent and identically distributed. However, it is becoming increasingly apparent that many soil properties are not independent, but are correlated either spatially or temporally. Such soil properties are said to be regionalized. This renders the use of traditional statistical methods invalid. Originally developed in mining engineering, geostatistics is becoming an important statistical tool for the description and comparison of spatially-dependent data in many scientific disciplines. Geostatistics focuses on the detection, modeling, and estimation of spatial patterns. The basic tool of geostatistics, the semivariogram, shows the distance at which samples become independent (the range) for the soil property of concern. At separation distances within the range, samples are spatially correlated with each other. This information can then be used in conjunction with another geostatistical method, kriging, which provides estimates of unknown values at locations where measurements have not been made. Semivariograms and kriging may suggest sample groupings for minesoil chemical characterization (e.g., soil pH, electrical conductivity, metals, nutrients) and minesoil management practices (e.g., lime additions and fertilizer recommendations). The purpose of this poster is to show, using a hypothetical example, the theoretical basis of geostatistics and to discuss how geostatistics can be utilized to describe the spatial variability of coal minesoils for characterization and management.

CORN AND FORAGE PRODUCTIVITY OF OHIO MINESOILS

John F. Underwood and Paul Sutton. (The Ohio State University, Columbus, OH, USA).

Four non-prime minesoils have been evaluated for corn (*Zea mays* L.) grain production and three for cool-season grass production in southeastern Ohio. All sites had a minimum of 15 cm of silty-clay loam topsoil placed over the graded spoil. Corn experiments were conducted on Bethesda (loamy skeletal, mixed, acid, mesic), Fairpoint (loamy skeletal, mixed, nonacid, mesic), Farmerstown (fine loamy, mixed acid, mesic) and Morristown (loamy skeletal, mixed (calcareous), mesic) minesoils. All are Typic Udorthents. Comparisons were made to nearby unmined reference plots representative of soils at sites before mining. Corn grain yields from 41 site-years that included all four minesoils ranged from 1.13 to 8.97 Mg/ha. The mean (4.64 Mg/ha) was 62% of nearby unmined soil and 80% of county average yields. Mined plots averaged 8% lower plant stands, and 2% higher grain moisture at harvest, and plants were 48 cm shorter in height—three indications of plant stress. Chisel tillage (36 to 38 cm depth with 76 cm spacing) increased corn yields only one time in 13 site-years. All but the Bethesda are considered satisfactory for corn production. Field studies with forages were conducted for 38 site-years on three minesoils (Bethesda, Fairpoint and Morristown) to determine the influence of fertilizer additions on productivity. Rates of fertilizer applied were for N (112 and 224 kg/ha), for P (24 and 48 kg/ha), and for K (46 and 92 kg/ha). The fertilizer was applied annually in eight treatment combinations to predominantly tall fescue (*Festuca arundinacea* Schreb). The mean dry matter yield on the unfertilized minesoil was 2.18 Mg/ha or 55% of the yield on the nearby unmined unfertilized plots. The highest forage yield of 6.66 Mg/ha was from the 224-48-92 (N-P-K) kg/ha treatment combination which was 90% of the yield from the nearby unmined soil with the same treatment. This yield was 161% of the county average. Maximum forage yields from minesoils were closely related to available water in the A horizons ($r = 0.98^{**}$). Bethesda, Fairpoint, and Morristown all have proven suitable for cool-season grass production, but greatest seasonal yield variability occurred with the Bethesda minesoil.

THE CARABID COMMUNITIES OF A RESTORATION BLASTED LIMESTONE QUARRY FACE IN DERBYSHIRE, UK

C. Philip Wheater and W. Rod Cullen (Dept. of Environmental and Geographical Sciences, The Manchester Metropolitan University, Manchester M1 5GD, UK).

Since abandoned limestone quarries are often unstable, nutrient deficient and subject to excessive drainage, they are difficult to reclaim for amenity, recreation or conservation. One method has recently been developed in Derbyshire, UK. Restoration Blasting involves the controlled blasting of worked faces to replicate daleside landform sequences, which, following management for the establishment of plants, will support self-sustaining ecosystems similar to those on natural dalesides. The establishment of invertebrates on such landforms may be slow, relying on the natural colonisation and establishment of herbivores and scavengers in advance of predators, parasites and parasitoids. It is expected that predator communities will only become established once suitable prey are present. During the initial stages of invertebrate community development, a large turnover of predator species may be predicted as individual colonisation and extinction events take place. Once the predator community begins to establish and stabilise then it may be that the invertebrate community as a whole is approaching a sustainable system. In order to examine whether this has occurred, one group of invertebrates has been examined in some detail. The Carabidae (Coleoptera) are an important group of surface active predators which have been well studied in other environments. Comparisons are made between the communities found on natural dalesides and on restoration blast sites produced at different times, as well as in relation to the time since restoration blasting and the developing vegetation communities.

EARTHWORM INTRODUCTION ON CALCAREOUS MINESOILS

Wiryono, John P. Vimmerstedt, and David A. Kost (The Ohio State University, Ohio Agricultural Research and Development Center, Wooster, OH, USA)

Burrowing activity of the nightcrawler, *Lumbricus terrestris* (*L.t.*), incorporates organic matter into mineral soil while creating long-lasting macropores. Thus *L.t.* has potential as a biological means of improving physical and chemical properties of surface mined areas. Our efforts to establish *L.t.* Lpopulation on forested acidic or calcareous minesoils have been successful, but thus far have not been able to establish *L.t.* in grassland ecosystems on calcareous minesoils. In May , 1989, we put 11 clitellate *L.t.* under sphagnum moss on calcareous gray cast overburden (an alternative resoiling material), on standard graded topsoil, or on ripped and disked topsoil. All soils had cover of agronomic grasses and legumes. We found no *L.t.* at the 24 points of inoculation during sampling in fall of 1990 with formalin extractant, although smaller species, *Lumbricus rubellus* and *Dendrobaena spp.* , were found. At another location, in May, 1990, we put 25 clitellate *L.t.* at 16 points in grasslands growing on gray cast overburden. Using formalin extraction, we found no *L.t.* in May 1992 at these locations. Working in this same area in November, 1992, we released 10 clitellate *L.t.* at 16 points under 10 cm of moist *Alnus glutinosa* leaf litter. Careful examination of the surface inoculation points in spring and fall of 1993 did not show obvious signs of earthworm activity. Our next step will be to use Earthworm Inoculation Units (earthworm-minesoil microcosms containing *L.t.* adults, immatures, and cocoons) as the source of the new populations.

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