# PROCEEDINGS OF THE 1990 MINING AND RECLAMATION CONFERENCE AND EXHIBITION


Editors:

Jeff Skousen
John Sencindiver
Dave Samuel




This page intentionally left blank.

# PROCEEDINGS OF THE 1990 MINING AND RECLAMATION CONFERENCE AND EXHIBITION

## APRIL 23-26, 1990
## CHARLESTON, WEST VIRGINIA

Conference Chairmen
Jeff Skousen
John Sencindiver

Published by:

West Virginia University Publication Services
Communications Building
Morgantown, WV 26506
304-293-6366

Additional copies may be obtained from WVU Publication Services.

ISBN 0-925500-02-X

This proceedings may be cited as:

Skousen, J., J. Sencindiver, and D. Samuel (Eds.). 1990. Proceedings of the 1990 Mining and Reclamation Conference and Exhibition. 2 Vols. West Virginia University, Morgantown, WV.

This Proceedings was published with funds granted by the National Mine Land Reclamation Center at West Virginia University and from proceeds from registration fees from the conference.

# The 1990 Mining and Reclamation Conference and Exhibition

## Sponsors

AMERICAN SOCIETY FOR SURFACE MINING AND RECLAMATION
WEST VIRGINIA UNIVERSITY
WEST VIRGINIA MINING AND RECLAMATION ASSOCIATION
NATIONAL MINE LAND RECLAMATION CENTER
OFFICE OF SURFACE MINING RECLAMATION AND ENFORCEMENT
U.S. BUREAU OF MINES
WEST VIRGINIA SURFACE MINE DRAINAGE TASK FORCE
COAL MAGAZINE AND INDUSTRIAL PRESENTATIONS
COAL PEOPLE MAGAZINE

# The 1990 Mining and Reclamation Conference and Exhibition

## Exhibitors

American Excelsior Company
American Mining Congress
American Society for Surface Mining and Reclamation
Arctech, Inc.
Austin Powder Company
Bennett & Williams, Inc.
Bradley & Foster, Inc.
Camber Corporation
Coal Magazine/ E & MJ
Coal People Magazine
Creech Services
Cummins Cumberland, Inc.
Dynamic Graphics
Esmer & Associates, Inc.
GAI Consultants, Inc.
Georgia Duck & Cordage Mill
GeoSonics, Inc.
Gooding, Lewis, Goetz and Company, Inc.
Hausner Hard-Chrome
Hotsy Equipment Company
IMO Industries
Industrial Rubber Products Co.
Kanawha Manufacturing Company
Krupp Industries
MK - Environmental Service
McCoy & McCoy Int'l.
The Mearl Corporation
Mincom U.S.A., Inc.
Nalco Chemical Company
National Lime & Stone Co.
National Scale
Office of Surface Mining
Petroleum Products, Inc.
Postle Industries Inc.
Quantech Steel Corporation
Republic Industries
REICHdrill Inc.
Sandvik Rock Tools Inc.
Shirt Gallery
Smith International
Strickly Business Systems
Tricon Metals & Services, Inc.
U.S. Bureau of Mines
Vibra-Tech Engineers, Inc.
West Virginia University

# The 1990 Mining and Reclamation Conference and Exhibition

## EXECUTIVE COMMITTEE

Jeff Skousen
Patty Bruce
Bill Plass
Paul Ziemkiewicz
Mark Sprouls
John Sencindiver
Trish Erickson
Jim Boyer
Dave Samuel
Sam Posa

## PLANNING COMMITTEE

Chairmen:

Jeff Skousen
John Sencindiver

Program:

Jeff Skousen
John Sencindiver
Mark Sprouls

Finances:

Mary Ann Steele
Patty Bruce
Jeff Skousen

Publicity:

Sheri Harris
Jeff Skousen
Al Skinner
Mark Sprouls

Registration:

Patty Bruce

Exhibits:

Sam Posa

Tours:

Walt Davidson
Jack Muncy
Carl Zipper
Neil Robinson
Jeff Skousen
Ken Politan

Civic Center and Hotel:

Cindy Wilson
John Robertson
Jeff Skousen

Proceedings:

Jeff Skousen
John Sencindiver
Dave Samuel
Lynn Edington

Special Activities:

Cindy Wilson
Judy Ward
John Robertson
Patty Bruce

# Acknowledgement

The Editors would like to acknowledge the following individuals for serving as technical reviewers for the papers of this conference.

Tom Ammons
Clark Ashby
Richard Barnhisel
Dick Bielicki
Roger Bollinger
John Bowders
James Burger
Eric Carlson
Frank Caruccio
Tom Crandall
Lee Daniels
Walt Davidson
Robert Dunker
Patricia Erickson
Bill Evangelou
Ben Faulkner
John Freeman
Keith Garbutt
Robert Hedin
Everett Jencks
Robert Keefer
Robert Kleinmann
Bruce Leavitt
Bernie Maynard
Jim McGraw
Al Meek
Jack Muncy
Jack Nawrot
Eric Perry
John Renton
Joe Schueck
Alan Sexstone
Andy Sobek
Al Stiller
John Torbert
Al Whitehouse
Paul Ziemkiewicz
Carl Zipper

MODERATORS

| | |
|---|---|
| Acid Mine Drainage | Charlie Miller<br>Bruce Leavitt |
| Evaluating Reclamation Success: Ecological Considerations | Gary Wade<br>Jeanne Chambers |
| Minesoil Productivity and Postmining Land Uses | Robert Dunker |
| Reclamation for Improved Land Uses in Appalachia | Carl Zipper |
| Restoration of Forested Wetlands | Tim White |
| Plenary Session | John Sencindiver |
| Reclamation Policy and Regulation | Al Whitehouse |
| Mining, Reclamation, and the Media | Bill Case |
| Overburden Analysis | Steve Keen |
| Forestry | Jack Muncy |
| Wetlands - General | Jack Nawrot |
| Wetlands - U.S. Bureau of Mines | Bob Hedin |
| Abandoned Mine Lands | Dick Bielicki<br>Paul Ziemkiewicz |
| Revegetation - General | Eric Perry |
| Revegetation - AML | Jeff Skousen |

# Table of Contents

List of Sponsors . . . . . iii
List of Exhibitors . . . . . iv
Conference Committees . . . . . v
Acknowledgement . . . . . vi
Moderators . . . . . vii
List of Authors . . . . . xv

# Volume I.

## ACID MINE DRAINAGE

**Chemical Stability of Manganese and Iron in Mine Drainage Treatment Sludge: Effects of Neutralization Chemical, Iron Concentration, and Sludge Age** . . . . . 3
G. Watzlaf and L. Casson

**Engineering Evaluation of Waste Phosphatic Clay for Producing Low Permeability Barriers** . . . . . 11
J. Bowders and E. Chiado

**A Comparative Study of Tailings Analysis Using Acid/Base Accounting, Cells, Columns and Soxhlets** . . . . . 19
W. Bradham and F. Caruccio

**Long-Term Leaching of Mine Spoil With Simulated Precipitation** . . . . . 27
H. Halverson and C. Gentry

**Successful Reclamation Using Controlled Release Bactericides: Two Case Studies** . . 33
A. Sobek, D. Benedetti, and V. Rastogi

**The Use of Phosphate Refuse as a Potential AMD Ameliorant** . . . . . 43
W. Hart, A. Stiller, T. Rymer, and J. Renton

**Advances in the Prediction and Control of Acid Mine Drainage** . . . . . 51
P. Ziemkiewicz

**Neutralization and Stabilization of Combined Refuse Using Lime Kiln Dust at High Power Mountain** . . . . . 55
D. Rich and K. Hutchison

**Some Aspects of SSPE/PSM Modeling for Quantitative Assessment of Disturbed Hydrological Systems** . . . . . 61
T. Rymer, A. Stiller, W. Hart, and J. Renton

**Coal Refuse Reclamation: Effects of Topsoil Thickness on the Growth and Maintenance of Vegetation** . . . . . 69
R. Kolbash, J. Freeman, T. Romanoski, and J. Sturm

**Mine Drainage Bioremediation - The Evolution of the Technology from Microbes to Bio-Carb** . . . . . 71
J. Davison

## EVALUATING RECLAMATION SUCCESS: ECOLOGICAL CONSIDERATIONS

**Minesoil Morphology and Genesis** . . . . . 79
J. Sencindiver, D. Dollhopf, L. Daniels

**Nutrient Cycling: The Key to Reclamation Success?** . . . . . 81
J. Palmer

**Soil Microbial Processes and Dynamics: Their Importance to Effective Reclamation** . 83
J. Zak and S. Visser

**The Use of Mycorrhizal Fungi in Reclaiming a Landscape: Importance and Predictability** . . . . . 85
M. Allen and C. Friese

**Faunal Community Development on Disturbed Lands: An Indicator of Reclamation Success** . . . . . 87
R. Parmenter and J. MacMahon

**Seed and Seedbed Ecology** . . . . . 89
J. Young

**Evaluating Community Level Processes to Determine Reclamation Success** . . . . . 91
E. Allen

**Successional Processes: Importance for Obtaining and Evaluating Reclamation Success** . . . . . 93
J. MacMahon, J. Chambers, and G. Wade

**Landscape Ecology and Reclamation Success** . . . . . 95
P. Tueller

**Cumulative Effects of Mining on Hydrology, Water Quality, and Vegetation** . . . . . 97
H. Halverson and R. Sidle

**Ecological Importance of Post-reclamation Land Uses on Evaluating Reclamation Success** . . . . . 99
R. Ries and R. Hedin

## MINESOIL PRODUCTIVITY AND POSTMINING LAND USES

**Guide for Selecting an Appropriate Method to Analyze the Stability of Slopes on Reclaimed Surface Mines** . . . . . 103
J. Bowders and S. Lee

**Application of the Principles of Microbial Ecology to the Assessment of Surface Mine Reclamation** . . . . . 111
J. Harris and P. Birch

**Forage Production on a Minesoil in Southern West Virginia** . . . . . 121
B. Wolf, L. Emerson, and J. Sencindiver

**Organic Matter Dynamics on Reclaimed Coal Mines in the Canadian Rockies** . . . . . 127
P. Ziemkiewicz and S. Takyi

**Evaluation of Minesoil Quality and Capability in Alberta, Canada** . . . . . 135
T. Macyk

**Managing Restored Prime Farmland for Corn Production** . . . . . 143
R. Barnhisel and R. Gray

## RECLAMATION FOR IMPROVED LAND USE IN APPALACHIA

**Variances from "Approximate Original Contour" Requirements in Central Appalachia: History and Prospects** . . . . . 153
C. Zipper

**Mined Land Reclamation for Wood Production in the Appalachian Region** . . . . . 159
J. Burger and J. Torbert

**The Feasibility of Large-Scale Sewage Sludge/Compost Utilization on Central Appalachian Surface Mined Lands** . . . . . 165
W. Daniels and K. Haering

**Occurrence of Ground Water in Mine Spoil, A Renewable Resource: Star Fire Tract, Eastern Kentucky** . . . . . 171
J. Dinger, D. Wunsch, and J. Kemp

**Innovative Design for Improved Mining Economics and Land Use** . . . . . 181
T. Keeney and G. Hall

**The Land Use Planning Process As Applied to a Mountaintop Removal Coal Mine Site in Eastern Kentucky** . . . . . 185
T. Nieman and D. Meshako

**RECLAMATION POLICY AND REGULATORY ISSUES**

**Evaluation of Methods to Determine Plant Productivity for Bond Release** . . . . . 193
J. Skousen

**Mineral Resources Policies and the Construction Aggregate Industry** . . . . . 204
A. Bauer

**Reclamation Practice, Perceptions and Policy: The Australian Experience** . . . . . 205
D. Whitrow

**MINING, RECLAMATION AND THE MEDIA**

Panelists: A. Sanda (Coal Magazine) . . . . . 210
P. Nyden (Charleston Gazette) . . . . . 209
N. Kilpatrick (Surface Mining Research Library) . . . . . 209
E. Smith (Mine Regulation Reporter) . . . . . 210

**OVERBURDEN ANALYSIS**

**Evaluation of Acid Base Accounting Data Using Computer Spreadsheets** . . . . . 213
M. Smith and K. Brady

**Effectiveness of the Addition of Alkaline Materials at Surface Coal Mines in Preventing and Abating Acid Mine Drainage: Part 1. Geochemical Considerations** . . . . . 221
C. Cravotta III, K. Brady, M. Smith, and R. Beam

**Effectiveness of the Addition of Alkaline Materials at Surface Coal Mines in Preventing and Abating Acid Mine Drainage: Part 2. Mine Site Case Studies** . . . . 227
K. Brady, M. Smith, R. Beam, and C. Cravotta III

**The Use of $CO_2$ Coulometry in Differentiating and Quantifying the Carbonate Phases in the Coal-Bearing Strata of Western Pennsylvania: Its Applicability in Interpreting and Modifying Neutralization Potential (NP) Measurements** . . . . . 243
J. Morrison, S. Atkinson, A. Davis, and B. Scheetz

**Delineation of Potential Manganese Sources in the Coal Overburdens of Western Pennsylvania** . . . . . 249
J. Morrison, S. Atkinson, and B. Scheetz

**The Effect of Oxygen on Pyrite Oxidation** . . . . . 257
R. Hammack and G. Watzlaf

**An Application of Acid-Base Accounting for Highway Construction in East Tennessee** . 265
J. Ammons, C. Coburn, Jr., and P. Shelton

## FORESTRY

**Guidelines for Establishing Productive Forest Land on Reclaimed Surface Mines in the Central Appalachians** . . . . . 273
J. Torbert and J. Burger

**Influence of Reforestation Reclamation Treatments on Nitrogen Dynamics in Minesoils** . . . . . 279
S. Schoenholtz, J. Burger, and R. Kreh

**Factors Limiting Tree Growth in Southern Illinois Under SMCRA** . . . . . 287
W. Ashby

**Successful Oak Establishment on a Reclaimed Surface Mine** . . . . . 295
W. Davidson, A. Freeland, B. Elison

**Results of White Pine Establishment Study in Southern West Virginia After Four Years** . . . . . 299
T. Probert, R. Gallimore, J. Torbert, and J. Burger

**Using Land Reclamation Practices to Improve Tree Condition in the Sudbury Smelting Area, Ontario, Canada** . . . . . 307
P. Beckett and J. Negusanti

# Volume II.

## RESTORATION OF FORESTED WETLANDS

**Development and Summary of MiST: A Classification System for Pre-Project Mitigation Sites and Criteria for Determining Successful Replacement of Forested Wetlands** . . . . . 323
T. White, R. Lea, R. Haynes, W. Nutter, J. Nawrot, M. Brinson, and A. Clewell

## WETLANDS - GENERAL

**Forested Wetland Reclamation Success Criteria Development in North Florida** . . . . . 339
J. Wester

**Observations on the Geologic History of a Wetland Have the Potential of Assisting Efforts to Construct Wetlands on Mine Lands** . . . . . 347
R. Behling

**Impacts of Volunteer Cattail Wetlands on Drainage Quality from Reclaimed Mined Land in Northern West Virginia** . . . . . 349
E. Jamison and H. Rauch

**Simulated Typha Wetlands Applied to Removal of Iron and Manganese from Acid Mine Drainage** . . . . . 351
J. Calabrese, D. Bhumbla, A. Sexstone, G. Bissonnette, and J. Sencindiver

**Mine Drainage Treatment Through Enhanced Biological Sulfate Reduction Within Wetlands** . . . . . 353
J. Nawrot

**Diversity Patterns of Invertebrate Fauna in Cattail Wetlands Receiving Acid Mine Drainage** . . . . . 365
M. Lacki, J. Hummer, and H. Webster

**Wildlife Use of Mitigated Wetlands on Surface Mined Lands in Western Pennsylvania** . 373
F. Brenner and D. Hofius

## WETLANDS - U.S. BUREAU OF MINES


**Sizing and Performance of Constructed Wetlands: Case Studies** . . . . . 385
R. Hedin and R. Nairn

**Iron Loading, Efficiency, and Sizing in a Constructed Wetland Receiving Mine Drainage** . . . . . 393
L. Stark, E. Stevens, H. Webster, and W. Wenerick

**Wetland Sizing - Design and Treatment Effectiveness** . . . . . 403
D. Kepler

**The Use of Bacterial Sulfate Reduction in the Treatment of Drainage from Coal Mines** . . . . . 409
P. McIntire and H. Edenborn

**Metal Removal Efficiencies from Acid Mine Drainage in the Big Five Constructed Wetland** . . . . . 417
T. Wildeman, S. Machemer, R. Klusman, R. Cohen, and P. Lemke

**Tests of Organic Matter Additions for the Neutralization of Acid Mine Drainage Influenced Lakes** . . . . . 425
R. Brugam, J. Gastineau, and E. Ratcliffe


## REVEGETATION - GENERAL


**Reclamation of Soil Sterilant Contaminated Soils** . . . . . 437
S. Landsburg and A. Fedkenheuer

**Comparison of Different Mulches, Lime and Fertilizer Applications on the Establishment and Growth of Vegetation on Surface Coal Mining Land** . . . . . 445
F. Brenner

**Establishment of Native Plant Species from Forest Topsoil Seedbanks on a Borrow Area in Kentucky** . . . . . 451
G. Wade and R. Thompson

**Recolonization of Vesicular-Arbuscular Mycorrhizae on a Reclaimed Strip Mine in North Dakota** . . . . . 461
J. Cockrell

**Importance of Indigenous VAM Fungi for the Reclamation of Coal Refuse Piles** . . . . 463
D. Dove, W. Daniels, and D. Parrish

**Retardation of Pyrite Oxidation by Massive Fly Ash Application to Mine Spoils** . . . 469
D. Bhumbla, R. Singh, and R. Keefer


## ABANDONED MINE LAND - GENERAL


**Similitude Model Experiments to Detect Mine Cavities** . . . . . 473
F. Ziaie, S. Peng, and P. Lin

**Highwall Stability Analysis on a PC** . . . . . 483
P. Lin, D. Dutta, and S. Peng

**Using a Magnetometer for Investigating Underground Coal Mine Fires, Burning Coal, Refuse Banks, and for Locating AMD Source Areas on Surface Mines** . . . . . 493
J. Schueck

**Locating and Abating Sources of Acid Mine Drainage at Reclaimed Strip Mines** . . . . 503
T. Ackman

**Distribution and Characteristics of Outcrop Fires in Horizontal Strata in Eastern and Western Coal Provinces** . . . . . 505
B. Maynard

**Abandoned Mine Land Reclamation - Unique Projects in West Virginia** . . . . . 507
N. Robinson

**A Case Study of a Large Open Pit Uranium AML Project: Gas Hills, Wyoming** . . . . . 513
M. Koopman and R. Rash

**Fire Diagnostic Simulation for Burning Coal Waste Banks** . . . . . 521
J. Jones and R. Chaiken

**Mine Fire Diagnostics in Abandoned Bituminous Coal Mines** . . . . . 527
L. Dalverny, R. Chaiken, and A. Kim

**Relative Self-Heating Tendencies of Coal, Carbonaceous Shales and Coal Refuse** . . . 535
A. Kim and R. Chaiken

## REVEGETATION - AML

**Tree Species Composition, Canopy Coverage, and Importance on Several AML Sites in Northern West Virginia** . . . . . 545
C. Johnson and J. Skousen

**Revegetation of Abandoned Mine Lands in Pennsylvania with Containerized Seedlings and Soil Amendments** . . . . . 555
R. Hutnik and H. Hughes

**An Evaluation of Lime Requirement Tests on Pyritic Minesoils** . . . . . 563
P. Cagnetta and E. Jencks

**Survival and Growth of Bigtooth Aspen on Acidic Surface-Mine Soils as Influenced by *Pisolithus tinctorius* and Nitrogen and Phosphorus Fertilization** . . . . . 573
G. Demuro, E. Jencks, and D. Hindal

**Long-Term Effects of a Single Application of Municipal Sludge on Abandoned Mine Land** . . . . . 579
W. Sopper and E. Seaker

**Stimulation of Aspen Establishment on Unreclaimed Mine Spoils** . . . . . 589
R. Hedin and E. Hedin

## POSTERS

**Wildflowers for Reclamation** . . . . . 597
W. H. Davidson

**Constructing Treatment Wetlands - Maryland's Experience** . . . . . 599
F. Bagley and C. Lyons

**Mining and Reclamation at Hobet Mining, Inc.** . . . . . 601
L. Emerson

**Direct Seeding of Anthracite Refuse Using Coal Fly Ash as a Major Soil Amendment** . . 603
J. Buck, R. Houston, and W. Beimborn

**Potential Application of Wildlife Habitat Evaluation Procedures to Design Better Reclamation Habitats: Case Study from a 3400 Acre Reservoir Project** . . . . . 605
W. Beimborn, G. Reese, and L. Ealy

**Potential Application of Wildlife Habitat Evaluation Procedures to Design Reclamation Habitats: Case Study from a 5 Mile Long, 4-Lane Highway Project** . . . . . 607
W. Beimborn, G. Reese, and L. Ealy

**Electrolytic Control of AMD** . . . . . 609
A. Stiller, E. Leluika, T. Rymer, and W. Hart

**Wildlife Species Diversity on Revegetated Coal Stripmine Lands in Interior Alaska** . 611
C. Elliott

**Overview of Long-Term Effects of Surface Mining on the Ground-Water Levels and Quality in Two Small Watersheds in Eastern Ohio** . . . . . 613
W. Cunningham

**Using Laboratory Mesocosms to Evaluate the Potential Effectiveness of Constructed Wetlands for Acid Mine Drainage Treatment** . . . . . 615
R. Weider, M. Linton, and K. Heston

## LIST OF AUTHORS


Ackman, T. . . . . 503
Allen, E. . . . . 91
Allen, M. . . . . 85
Ammons, J. . . . . 265
Ashby, W. . . . . 287
Atkinson, S. . . . . 243, 249
Bagley, F. . . . . 599
Barnhisel, R. . . . . 143
Bauer, A. . . . . 204
Beam, R. . . . . 221, 227
Beckett, P. . . . . 307
Behling, R. . . . . 347
Beimborn, W. . . . . 603, 605, 607
Benedetti, D. . . . . 33
Bhumbla, D. . . . . 351, 469
Birch, P. . . . . 111
Bissonnette, G. . . . . 351
Bowders, J. . . . . 11, 103
Bradham, W. . . . . 19
Brady, K. . . . . 213, 221, 227
Brenner, F. . . . . 373, 445
Brinson, M. . . . . 323
Brugam, R. . . . . 425
Buck, J. . . . . 603
Burger, J. . . . . 159, 273, 279, 299
Cagnetta, P. . . . . 563
Calabrese, J. . . . . 351
Caruccio, F. . . . . 19
Casson, L. . . . . 3
Chaiken, R. . . . . 521, 527, 535
Chambers, J. . . . . 93
Chiado, E. . . . . 11
Clewell, A. . . . . 323
Coburn, C. . . . . 265
Cockrell, J. . . . . 461
Cohen, R. . . . . 417
Cravotta, C. . . . . 221, 227
Cunningham, W. . . . . 613
Dalverny, L. . . . . 527
Daniels, W. . . . . 79, 165, 463
Davidson, W. . . . . 295, 597
Davis, A. . . . . 243
Davison, J. . . . . 71
Demuro, G. . . . . 573
Dinger, J. . . . . 171
Dollhopf, D. . . . . 79
Dove, D. . . . . 463
Dutta, D. . . . . 483
Ealy, L. . . . . 605, 607
Edenborn, H. . . . . 409
Elison, B. . . . . 295
Elliott, C. . . . . 611
Emerson, L. . . . . 121, 601
Fedkenheuer, A. . . . . 437
Freeland, A. . . . . 295
Freeman, J. . . . . 69
Friese, C. . . . . 85
Gallimore, R. . . . . 299
Gastineau, J. . . . . 425
Gentry, C. . . . . 27
Gray, R. . . . . 143
Haering, K. . . . . 165
Hall, G. . . . . 181
Halverson, H. . . . . 27, 97
Hammack, R. . . . . 257
Harris, J. . . . . 111
Hart, W. . . . . 43, 61, 609
Haynes, R. . . . . 323
Hedin, E. . . . . 589
Hedin, R. . . . . 99, 385, 589
Heston, K. . . . . 615
Hindal, D. . . . . 573

Hofius, D. . . . . 373
Houston, R. . . . . 603
Hughes, H. . . . . 555
Hummer, J. . . . . 365
Hutchison, K. . . . . 55
Hutnik, R. . . . . 555
Jamison, E. . . . . 349
Jencks, E. . . . . 563, 573
Johnson, C. . . . . 545
Jones, J. . . . . 521
Keefer, R. . . . . 469
Keeney, T. . . . . 181
Kemp, J. . . . . 171
Kepler, D. . . . . 403
Kim, A. . . . . 527, 535
Klusman, R. . . . . 417
Kolbash, R. . . . . 69
Koopman, M. . . . . 513
Kreh, R. . . . . 279
Lacki, M. . . . . 365
Landsburg, S. . . . . 437
Lea, R. . . . . 323
Lee, S. . . . . 103
Leluika, E. . . . . 609
Lemke, P. . . . . 417
Lin, P. . . . . 473, 483
Linton, M. . . . . 615
Lyons, C. . . . . 599
Machemer, S. . . . . 417
Macyk, T. . . . . 135
Maynard, B. . . . . 505
McIntire, P. . . . . 409
McMahon, J. . . . . 87, 93
Meshako, D. . . . . 185
Morrison, J. . . . . 243, 249
Nairn, R. . . . . 385
Nawrot, J. . . . . 323, 353
Negusanti, J. . . . . 307
Nieman, T. . . . . 185
Nutter, W. . . . . 323
Palmer, J. . . . . 81
Parmenter, R. . . . . 87
Parrish, D. . . . . 463
Peng, S. . . . . 473, 483
Probert, T. . . . . 299
Rash, R. . . . . 513
Rastogi, V. . . . . 33
Ratcliffe, E. . . . . 425
Rauch, H. . . . . 349
Reese, G. . . . . 605, 607
Renton, J. . . . . 43, 61
Rich, D. . . . . 55
Ries, R. . . . . 99
Robinson, N. . . . . 507
Romanoski, T. . . . . 69
Rymer, T. . . . . 43, 61, 609
Scheetz, B. . . . . 243, 249
Schoenholtz, S. . . . . 279
Schueck, J. . . . . 493
Seaker, E. . . . . 579
Sencindiver, J. . . . . 79, 121, 351
Sexstone, A. . . . . 351
Shelton, P. . . . . 265
Sidle, R. . . . . 97
Singh, R. . . . . 469
Skousen, J. . . . . 193, 545
Smith, M. . . . . 213, 221, 227
Sobek, A. . . . . 33
Sopper, W. . . . . 579
Stark, L. . . . . 393
Stiller, A. . . . . 43, 61, 609
Stevens, E. . . . . 393
Sturm, J. . . . . 69
Takyi, S. . . . . 127

Thompson, R. . . . . 451
Torbert, J. . . . . 159, 273, 299
Tueller, P. . . . . 95
Visser, S. . . . . 83
Wade, G. . . . . 93, 451
Watzlaf, G. . . . . 3, 257
Webster, H. . . . . 365, 393
Weider, K. . . . . 615
Wenerick, W. . . . . 393
Wester, J. . . . . 339
White, T. . . . . 323
Whitrow, D. . . . . 205
Wildeman, T. . . . . 417
Wolf, B. . . . . 121
Wunsch, D. . . . . 171
Young, J. . . . . 89
Zak, J. . . . . 83
Ziaie, F. . . . . 473
Ziemkiewicz, P. . . . . 51, 127
Zipper, C. . . . . 153

This page intentionally left blank.

# Acid Mine Drainage

This page intentionally left blank.

# CHEMICAL STABILITY OF MANGANESE AND IRON IN MINE DRAINAGE TREATMENT SLUDGE: EFFECTS OF NEUTRALIZATION CHEMICAL, IRON CONCENTRATION, AND SLUDGE AGE[1]

by

George R. Watzlaf and Leonard W. Casson[2]

Abstract. The objective of this research was to determine how various factors influence the solubility of iron and manganese in mine drainage treatment sludge. Sludge samples from the field and laboratory-generated sludge samples were used for this study. The sludge collected in the field was subjected to lower pH conditions by sulfuric acid titration within one hour of precipitation and after one, two, four, and six months of aging. Two conditions were used for sludge aging: under the treated supernatant to simulate aging in a settling pond and in air to simulate aging in a drying pond or disposal on coal spoil. Additional tests were performed using untreated mine drainage collected in the field and treated in the laboratory with hydrated lime, sodium carbonate, or sodium hydroxide. Prior to alkaline treatment, iron in the form of ferric sulfate was added to portions of this water to represent various levels of iron. The results of these tests indicate: 1) sodium carbonate produced a less stable sludge than either lime or sodium hydroxide, 2) iron and manganese were less stable when iron was precipitated as ferrous as opposed to ferric hydroxide, 3) aging increased sludge stability, particularly aging in air, and 4) manganese was slightly more stable in sludge precipitated with high iron concentrations.

Additional key words: Acid mine drainage, acid mine drainage treatment sludge, metal hydroxide solubility.

## Introduction

### Chemical Treatment of Acid Mine Drainage

Federal legislation requires mine operators to adhere to specific effluent limits (U. S. Code of Federal Regulations 1985 a & b) (Table 1). To meet these limits, treatment of acid mine drainage typically involves the addition of alkaline material, commonly lime (CaO or $Ca(OH)_2$), sodium hydroxide (NaOH), or sodium carbonate ($Na_2CO_3$); natural or mechanical aeration; and settling (U.S. Environmental Protection Agency 1983). When the pH of the drainage is raised to seven or eight, and there is sufficient time allowed for settling, most mine water will meet the standards for pH, iron, and suspended solids. However, this treatment will rarely reduce manganese levels below effluent limits. Diem and Stumm (1984) found that dilute solutions of manganese nitrate ($Mn(NO_3)_2$), maintained at pH 8.4 in the presence of dissolved oxygen, showed virtually no oxidation for over four years. Owens (1963) reported that a pH of at least 9.3 was necessary to remove over 50% manganese from solution using lime. In another study, pH values above 10 were necessary to reduce manganese concentrations below effluent limits for some mine drainage waters (Watzlaf 1985). At these sites, the reduction of manganese to 2 mg/L may necessitate discharging water with a pH higher than the effluent limit of 9.0. A mine operator in this situation can be granted a variance that permits discharge of water with a pH greater than 9.0. The need to raise pH to these high levels can increase chemical treatment costs 20% to 100% over the costs to remove iron (Watzlaf 1985) and will increase the volume of acid mine drainage treatment sludge. Nicholas and Foree (1979) found that increasing the treatment pH from 8 to 10 increases the required sedimentation basin area from 158 $m^2$ to 400 $m^2$ for NaOH, and from 200 $m^2$ to 316 $m^2$ for lime.

---

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990. Publication in this proceedings does not preclude authors from publishing their manuscripts, whole or in part, in other publication outlets.

[2]George R. Watzlaf is an Environmental Engineer, U.S. Bureau of Mines, Pittsburgh, PA 15236 and Leonard W. Casson is an Assistant Professor, University of Pittsburgh, Pittsburgh, PA 15261.

Table 1. Federal Effluent Limitations

| Pollutant or Pollutant Property | Maximum for any 1 day | Average of daily values for 30 days |
|---|---|---|
| Iron, total * | 7.0 mg/L | 3.5 mg/L |
| Manganese, total | 4.0 mg/L | 2.0 mg/L |
| Total Suspended Solids | 70.0 mg/L | 35.0 mg/L |
| pH | ----between 6.0 and 9.0---- | |

* New source performance standards for total iron; daily maximum - 6.0 mg/L and 30-day average - 3.0 mg/L

The sludge generated from the treatment of acid mine drainage (AMD) is typically (1) disposed of in underground coal mines, (2) disposed of with coal refuse or coal mine spoil, or (3) permanantly retained in ponds (Ackman 1982, Moss 1971). The fate of metals in the disposed sludge has not been examined. Underground disposal practices are receiving considerable attention due to pending regulation of class V injection wells by the U. S. Environmental Protection Agency. The problem of metal solubilization from AMD treatment sludge is potentially very significant. Millions of tons of mine drainage sludge have been and are being disposed. The solubilization of metals from this sludge could lead to significant increases in treatment costs as well as future liability to treat these metals.

Iron and Manganese Precipitates

Iron exists in acid mine drainage either in the ferrous ($Fe^{2+}$) or ferric ($Fe^{3+}$) form. Ferrous iron oxidation is dependent on pH and dissolved oxygen concentration of the water. At pH 7 and with ample supply of dissolved oxygen, ferrous iron oxidation proceeds very rapidly. Once iron is converted to the +3 state, ferric iron will hydrolyze to form ferric hydroxide. If the pH of mine drainage is raised above 8.3 and there is not enough oxygen available for ferrous iron oxidation, ferrous hydroxide can form (Stumm and Morgan 1981).

The precipitation and removal of manganese is more complex than that of iron. Soluble manganese is thought to exist in AMD mainly as $Mn^{2+}$, however, it can also exist in oxidation states of +3, +4, +6, and +7 (Sienko and Plane 1966, Morgan 1967). Initially, the average valence of manganese in an air-oxidized precipitate (removed at high pH) lies between 2.67 and 3.0 ($Mn_3O_4$ - hausmannite and $\beta$-MnOOH - feitknechtite, respectively) (Hem 1981, Murray et al. 1985). Both of these minerals are eventually transformed into the more stable $\gamma$-MnOOH - manganite (Murray et al. 1985, Stumm and Giovanoli 1976). Stumm and Morgan (1981) present a schematic showing initial formation of amorphous $Mn(OH)_2$ followed by tranformation into hausmannite, feitknechtite, manganite, and pyrolusite ($MnO_2$) in the presence of oxygen.

Manganese can also be removed by adsorption onto reactive surfaces. Ferric hydroxide ($Fe(OH)_3$) as well as manganese precipitates can provide surfaces for sorption of $Mn^{2+}$, which occurs at lower pH values than necessary for manganese removal as hydroxides (Morgan and Stumm 1964, Stumm and Morgan 1981, Collins and Buol 1970). Therefore, manganese in mine waters with high iron concentrations can usually be removed at lower pH values than manganese in mine water with lower iron levels. Additionally, manganese can exist in many different complexed and chelated forms (Martell and Calvin 1952, Clark et al. 1977, Evangelou 1984). The formation of manganese complexes depends on oxidation state, pH, bicarbonate-carbonate-hydroxide equilibria, and the presence of other materials (Nalco Chemical Company 1979). Organic complexes can hold manganese in solution to higher pH levels (Clark et al. 1977). If carbonates are present and oxygen is not readily available (low Eh), divalent iron and manganese can form $FeCO_3$ and $MnCO_3$, respectively (Stumm and Morgan 1981).

As an alternative to precipitation of manganese by the high-pH method, strong oxidizers (permanganate, chlorine dioxide, hypochlorite, or ozone) can be used. These chemicals oxidize manganese to pyrolusite ($MnO_2$) (Evangelou 1984, Clark et al. 1977). The use of chemical oxidants produces a stable manganese precipitate, however, treatment costs are greatly increased (Watzlaf 1985). These chemicals also require very accurate control of dose rate. In order for hypochlorite to be effective, enough chemical must be added to produce a chlorine residual. With adequate detention time and proper chemical dosage, this residual chlorine will dissipate before discharge. However, overtreatment and/or pond short-circuiting can result in discharge of high residual chlorine levels and subsequent damage to the environment. Similarly, the dosage of permanganate needs precise control. Too much permanganate will result in excess $MnO_4^-$, which will place the effluent in violation of the manganese limit. Because of these possible problems some state regulatory agencies discourage the use of these chemicals. Knocke et al. (1987) found that hydrogen peroxide was not effective for oxidizing manganese.

Experimental Methods

This paper discusses the results of various experiments to determine the effects of sludge aging, iron concentration, and alkaline treatment chemical on the stability of iron and manganese in AMD treatment sludge. The first series of tests consisted of collecting field samples and aging these samples in the laboratory for one, two, four, and six months. The second test series investigated the effects of iron concentration and treatment chemical in laboratory-generated sludge.

Aging of Field Samples

The first series of tests involved collecting AMD treatment sludge from five mine sites. These sites were selected based on manganese and iron concentration of the raw water and type of treatment chemical used. Table 2 shows the raw water quality and treatment chemical. Sites C, 7, M, and W treated seepage from reclaimed surface mines. Site S treated water from the pit of an active surface mine. Sites C and S both used sodium hydroxide with the In-line System (ILS), which is an aeration and mixing device developed by the Bureau of Mines (Ackman and Kleinmann 1985, Ackman and Place 1987). Site 7 added hydrated lime

utilizing a screw feeder. A gravity-fed hopper loaded with sodium carbonate briquettes was used at Site M. Sodium hydroxide was directly added to the raw water at Site W to raise pH to 11.6. Due to this high treatment pH, to the ferrous form of the iron, and to very little aeration, the sludge produced at Site W exhibited the typical green color of ferrous hydroxide. At Sites C, S, and 7, the treatment systems promoted aeration; therefore, sludge produced at these sites was a brownish-orange color indicating the precipitation of iron as ferric hydroxide.

At each site, three samples were collected immediately down stream of the point where the treatment chemical was added. These samples contained both treated water and resultant sludge. One of the three samples was used to determine stability of iron and manganese upon immediate pH depression by adding sulfuric acid to the stirred sludge/water mixture in the field. At a series of predetermined pH values, the stirring was stopped and the sample was permitted to settle. After 15 minutes, a 20 ml portion was withdrawn, filtered (0.45 $\mu$m), and acidified for subsequent metal analyses. The second sludge sample was aged while submerged under the treated supernatant water. Portions of these aged samples were tested for stability one, two, four, and six months after collection, using the sulfuric acid procedure outlined above. The third sample was filtered and aged in air and tested for stability after one, two, and four months of aging. Approximately 0.25 g of air-aged sludge was ground with mortar and pestle and added to 250 ml of distilled water. The mixure was stirred and sulfuric acid was added as outlined above.

Table 2. Raw water quality* and treatment chemical for the five sites used in sludge aging tests.

| | C | S | 7 | M | W |
|---|---|---|---|---|---|
| pH | 3.4 | 3.5 | 3.3 | 3.4 | 5.0 |
| Acidity | 330. | 740. | 460. | 180. | 680. |
| Sulfate | 1830. | 4350. | 1980. | 1400. | 2980. |
| Ferrous Iron | NA | NA | 37.0 | NA | 290. |
| Total Iron | 12.6 | 208. | 54.6 | 2.7 | 290. |
| Manganese | 48.9 | 111. | 94.0 | 62.0 | 117. |
| Chemical | NaOH | NaOH | $Ca(OH)_2$ | $Na_2CO_3$ | NaOH |

* All concentrations in mg/L except pH in pH units. Acidity in mg/L as $CaCO_3$. NA = not analyzed.

## Effects of Iron Concentration and Treatment Chemical

In addition to the above tests using sludge samples collected at various treatment plants, laboratory experiments were conducted under more controlled conditions to examine the effects of iron concentration and treatment chemical on the solubility of iron and manganese precipitates. A large sample of mine water containing more manganese than iron was brought back to the laboratory. This water sample was split into three portions. Additional iron (ferric sulfate) was added to two of the portions to yield three levels of iron, which will be referred to as low, medium, and high (Table 3). Each portion was treated with an alkaline substance to raise pH to approximately 10 in order to precipitate iron and manganese. The alkaline substances tested included hydrated lime, sodium hydroxide, and sodium carbonate. As in the previous tests, sulfuric acid was used to lower pH. At pH values of 10.0, 8.5, 7.0, 5.5, and 3.0, 20 ml were withdrawn, filtered (0.45 $\mu$m), and acidified for metal analysis.

Table 3. Raw water quality* of the low, medium, and high iron water used in laboratory tests.

| | Low | Medium | High |
|---|---|---|---|
| Ferrous Iron | < 6 | < 6 | < 6 |
| Total Iron | 17.5 | 59.7 | 559 |
| Manganese | 69.3 | 70.1 | 71.7 |

* All concentrations in mg/L.

## Chemical Analyses

For the unacidified samples of raw and treated mine water, acidity or alkalinity was determined by a fixed-end-point titration to pH 8.3 using sodium hydroxide or to pH 4.5 using sulfuric acid, respectively. Sulfate was analyzed by passing the sample through a cation-exchange resin and titrating with barium chloride using thorin (*o*-[2-hydroxy-3,6-disulfo-1-naphtyl)azo] benzenearsonic acid, disodium salt) as an indicator. For ferrous iron determinations, a potassium dichromate titration with a platinum redox electrode in the first derivative mode as an end-point indicator was used. Total iron and manganese were analyzed by inductively coupled argon plasma spectroscopy.

## Results and Discussion

In all of the results, the solubilization of iron or manganese was expressed as a fraction of the total amount of the metal in the sample. For the air-aged sludge, the total metal content was calculated by dissolving a known weight of sludge in both concentrated hydrochloric and nitric acids. For all of the other tests, total metal content was based on the volume of water collected and the concentraion of metal in this raw, untreated water. The removal of 20 ml portions, acidification of these portions, and addition of sulfuric acid to the samples were acccounted for in the calculation of fraction of metal solubilized.

## Aging of Field Samples

Freshly precipitated manganese began to solubilize at pH 8.5. At pH 7.0 and 3.0, respectively, 10% to 80% and 30% to 100% of the original manganese solubilized (figure 1a). Manganese became more stable after the sludge was aged for six months under water (Figure 1b). Four of the sites exhibited very similar trends of manganese solubilization; however, site M only showed a small increase in stability with 80% solubilizing at pH 3.0. This was the only site that used sodium carbonate for treatment.

For the fresh precipitates, manganese in the sludge from Site W was the least stable (Figure 1a). At this site, 20% of the iron, which was precipitated as ferrous hydroxide, solubilized at pH 7 (Figure 2). Other sites, where iron was precipitated as ferric hydroxide, produced sludge in which iron was stable down to pH 3.5. As the sludge from Site W aged under water, its color changed from green to brownish orange indicating the coversion from ferrous to ferric hydroxide. During this aging, both iron and manganese became much more stable.

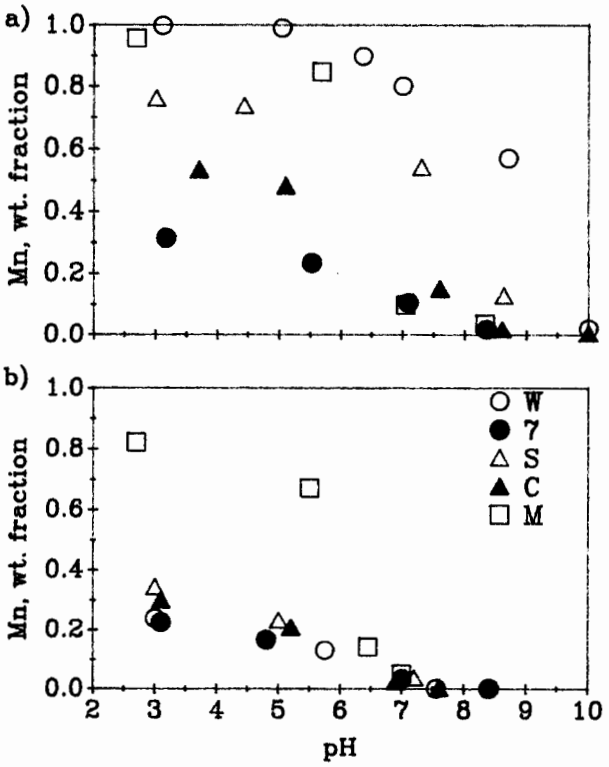



Figure 1. Effect of pH depression on the solubilization of manganese from the sludge of five mine sites, (a) immediately after metal precipitation and (b) after six months of aging under water.

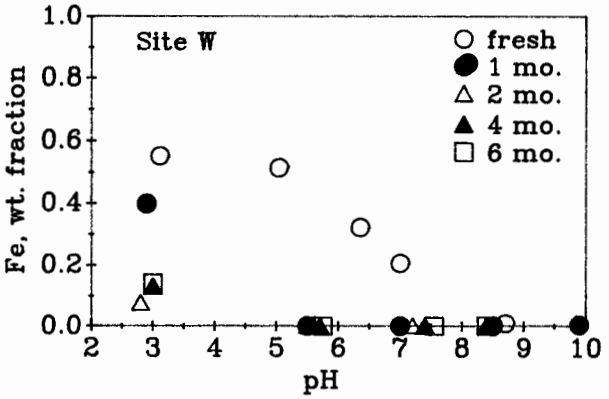



Figure 2. Effect of pH depression on the solubilization of iron from sludge of Site W as it aged under water.

The majority of the increase in manganese stability was realized within the first month of aging for Sites S, C, and 7, which used lime or sodium hydroxide and precipitated iron as ferric hydroxide (Figure 3). Sludge from Site M, which used sodium carbonate, and Site W, which precipitated ferrous hydroxide, both showed increases in manganese stability throughout the six-month test (Figure 4).

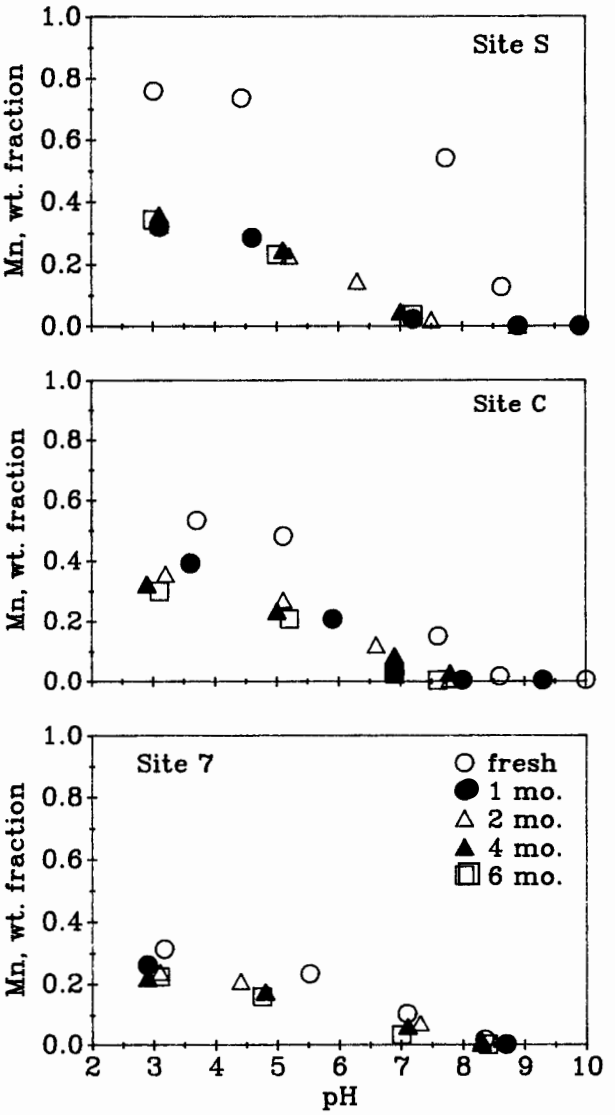



Figure 3. Effect of pH depression on the solubilization of manganese from sludge from Sites S, C, and 7 as it aged under water.

At all sites, iron and manganese became more stable when the sludge was aged in air than when it was aged under water. Table 4 shows manganese and iron solubilization after one month of aging under water and in air for Sites C, W, and M. Sites S and 7 are not shown since they were very similar to Site C. Air aging had the greatest effect on Site W. At pH 2.9 or 3.0, 40% and 69% of the iron and manganese, respectively, solubilized from the sludge aged under water, compared to 1% and 3%, respectively, solubilized from the air-aged sludge.

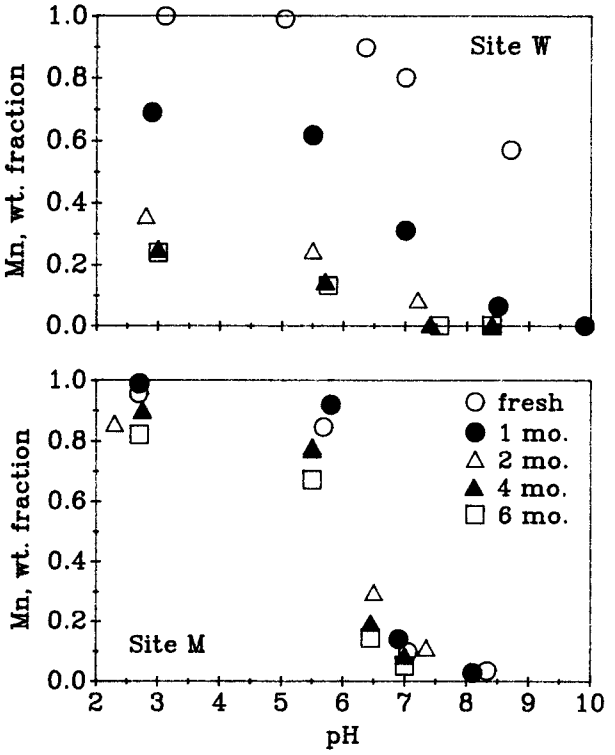



Figure 4. Effect of pH depression on the solubilization of manganese from sludge from Sites W and M as it aged under water.

Table 4. Fraction of Metal Solubilized After 1 Month of Aging in Air and Under Water.

| Site | pH | Manganese | | Iron | |
|---|---|---|---|---|---|
| | | Water | Air | Water | Air |
| C | 7.0 | --- | <0.01* | --- | <0.01 |
| | 6.9 | 0.03 | --- | <0.02 | --- |
| | 5.9 | 0.21 | --- | <0.02 | --- |
| | 5.7 | --- | <0.01 | --- | <0.01 |
| | 5.0 | --- | 0.03 | --- | <0.01 |
| | 3.6 | 0.39 | --- | 0.10 | --- |
| | 3.0 | --- | 0.45 | --- | 0.07 |
| S | 8.5 | 0.07 | --- | <0.01 | --- |
| | 7.0 | 0.31 | --- | <0.01 | --- |
| | 6.1 | --- | <0.01 | --- | <0.01 |
| | 5.5 | 0.62 | --- | <0.01 | --- |
| | 4.3 | --- | <0.01 | --- | <0.01 |
| | 3.0 | --- | 0.03 | --- | 0.01 |
| | 2.9 | 0.69 | --- | 0.40 | --- |
| M | 8.1 | 0.03 | --- | <0.09 | --- |
| | 7.1 | --- | 0.08 | --- | <0.02 |
| | 6.9 | 0.14 | --- | <0.09 | --- |
| | 5.8 | 0.92 | --- | <0.09 | --- |
| | 4.2 | --- | 0.65 | --- | <0.05 |
| | 2.7 | 1.00 | --- | 0.35 | --- |

* "<" indicates fraction solubilized was below detectable limits. These limits vary depending on dilution before analysis as well as total metal content in the sludge.

Effects of Iron Concentration and Treatment Chemical

For the laboratory-generated sludge, the chemical stability of manganese was the same for the sludges precipitated with lime or sodium hydroxide (Figure 5). However, sodium carbonate produced a sludge more prone to manganese solubilization. At almost all pH values and regardless of iron concentration, approximately twice as much manganese solubilized from the sodium carbonate-treated sludge than from the lime or sodium hydroxide sludges.

Figure 6 shows that manganese was more stable when it was precipitated with high iron concentrations with all three chemical treatments. This increase in stability was most prominent between pH 3.0 and 7.0 for the lime and sodium hydroxide treatments, and between pH 7.0 and 8.5 for the sodium carbonate treatment.

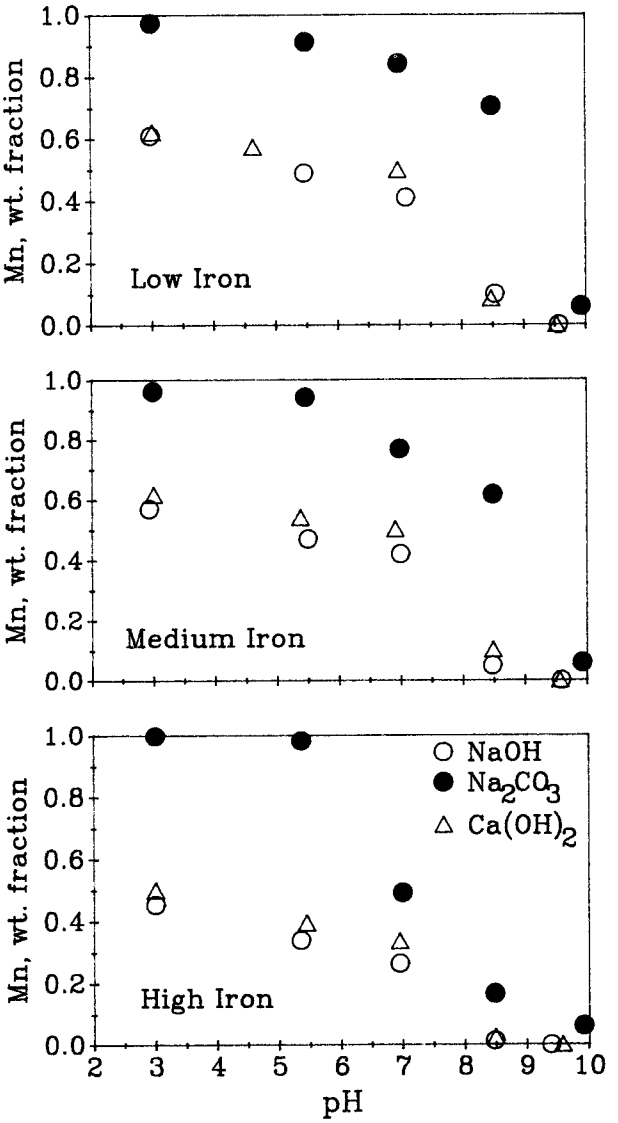



Figure 5. Effect of pH depression on the solubilization of manganese from unaged, laboratory-generated sludge. To show differences between treatment chemicals; sodium hydroxide, sodium carbonate, and lime treatments are plotted together for each concentration of iron in the raw water.

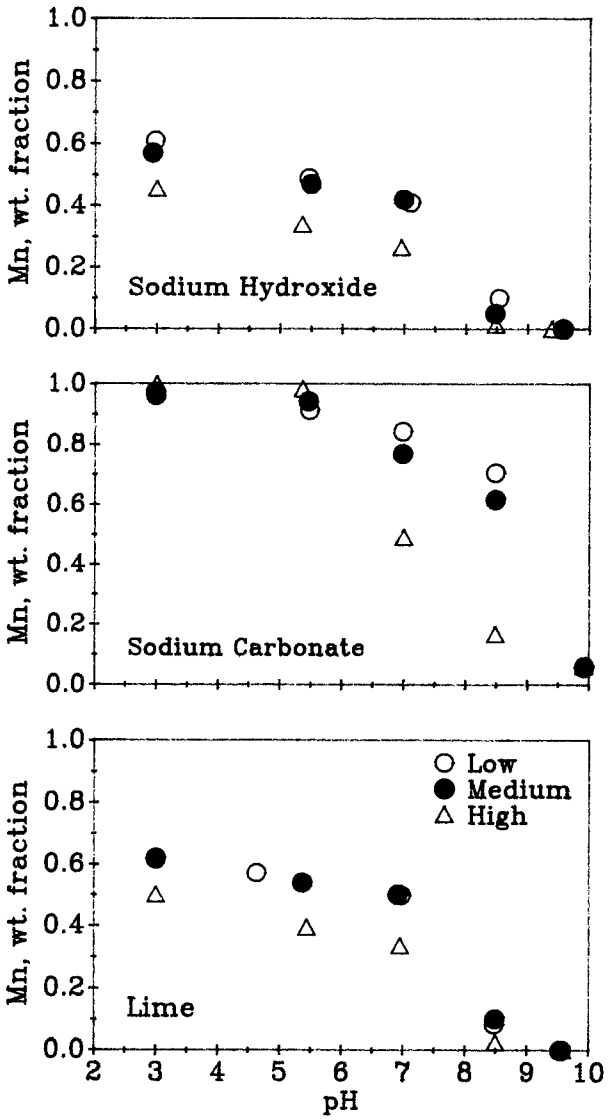



Figure 6. Effect of pH depression on the solubilization of manganese from unaged, laboratory-generated sludge. To show differences between iron concentrations; low, medium, and high iron concentrations are plotted together for each treatment chemical.

In all of the tests, iron remained insoluble at all pH values $\geq$ 5.5. At pH 3, the most iron solubilized from the sludge resulting from sodium carbonate treatment (Table 5).

Table 5. Fraction of iron solubilized at pH 3 from sludge treated with the listed chemical.

| IRON LEVEL | $Na_2CO_3$ | NaOH | $Ca(OH)_2$ |
|---|---|---|---|
| Low | 0.58 | 0.12 | 0.11 |
| Medium | 0.69 | 0.09 | 0.16 |
| High | 0.13 | 0.04 | 0.04 |

## CONCLUSIONS

Based on the results of laboratory studies and analysis of field samples the following conclusions can be made:

1. Iron and manganese were more stable in sludges precipitated with lime or sodium hydroxide than in sludge precipitated with sodium carbonate.

2. Iron and manganese were less stable if the iron was precipitated as ferrous hydroxide than if precipitated as ferric hydroxide.

3. Generally, iron and manganese were more stable in air-aged sludge than in sludge aged under water.

4. The largest increase in stability occurred in the first month of aging.

5. Manganese was more stable if precipitated with high iron concentrations.

## SUMMARY

In order to comply with effluent limits for manganese, mine operators have two main treatment options: use of alkaline materials for high-pH removal or use of chemical oxidants. Chemical oxidants are very rarely used due to their high costs and/or to their restriction by state regulatory agencies. A major concern of the high-pH method is the instability of manganese in the sludge it produces. Although we found that the manganese in the sludge becomes somewhat more stable as it ages, significant amounts of manganese may solubilize if the pH is not kept high. If hydrated lime or sodium hydroxide is used, 20% of the original manganese may solubilize at pH values between 5 to 7, even after six months of aging under water. If aged in air for four months, 10% to 20% of the manganese can solubilize at pH 4. If sodium carbonate is used, 75% and 65% of the manganese can solubilize at pH 4 from sludge aged for six months under water and sludge aged four months in air, respectively. If the sludge is disposed in or on surface mine backfill, any manganese that solubilizes may need to be retreated. The same may be true of underground disposal. This cycle of treatment, precipitation, sludge disposal, resolubilization, and retreatment may be repeated indefinitely at great expense to the operator. In fact, the high manganese concentrations currently found in some mine drainage may be due to sludge resolubilization. Additional research is needed to determine the fate of metals at actual sludge disposal sites in the field.

## Literature Cited

Ackman, T. E. 1982. Sludge disposal from acid mine drainage treatment. Bureau of Mines RI 8672, 25 p.

Ackman, T. E. and R. L. P. Kleinmann. 1985. In-line aeration and treatment of acid mine drainage: performance and preliminary design criteria. p. 53-61. In Control of Acid Mine Drainage, Bureau of Mines IC 9027.

Ackman, T. E. and J. M. Place. Acid mine water aeration and treatment system. U.S. Pat. 4,695,378, September 22, 1987.

Clark, J. W., W. Viessman, and M. J. Hammer. 1977. Water supply and pollution control. p. 444-495. Harper and Row, New York, NY.

Collins, J. F. and S. W. Buol. 1970. Effects of fluctuations in the Eh-pH environment on iron and/or manganese equilibria. Soil Science 110:111-118.

Diem, D. and W. Stumm. 1984. Is dissolved $Mn^{2+}$ being oxidized by $O_2$ in absence of Mn-bacteria or surface catalysts? Geochimica et Cosmochimica Acta 48:1571-1573.

Evangelou, V. P. 1984. Controlling iron and manganese in sediment ponds. Reclamation News and Views 2:1-6.

Hem, J. D. 1981. Rates of manganese oxidation in aqueous systems. Geochimica et Cosmochemica Acta 45:1369-1374.

Knocke ,W. R., R. C. Hoehn, and R. L. Sinsabaugh. 1987. Using alternative oxidants to remove dissolved manganese from waters laden with organics. Journal AWWA. March 1987, p. 75-79.

Martell, A. E. and M. Calvin. 1952. Chemistry of the metal chelate compounds. Prentice-Hall, New York, NY. 613 p.

Morgan, J. J. 1967. Chemical equilibria and kinetic properties of manganese in natural waters. p. 561-624. In Principles and applications of water chemistry. S. D. Faust and J. V. Hunter (Eds.). John Wiley and Sons, Inc., New York, NY.

Morgan, J. J. and W. Stumm. 1964. Colloid-chemical properties of manganese dioxide. Journal of Colloid Science 19:347-359.

Moss, E. A. 1971. Dewatering mine drainage sludge. EPA Project No. 14010 FJX, 90 p.

Murray, J. M., J. G. Dillard, R. Giovanoli, H. Moers, and W. Stumm. 1985. Oxidation of Mn(II): initial mineralogy, oxidation state and ageing. Geochemica et Cosmochimica Acta 45:463-470.

Nalco Chemical Company. 1979. The NALCO water handbook. Chapter 6 p. 15-16. McGraw-Hill, New York, NY.

Nicholas, G. D. and E. G. Foree. 1979. Chemical treatment of mine drainage for removal of manganese to permissible limits. p.181-187. In proceedings, Symposium on Surface Mining Hydrology, Sedimentology, and Reclamation. (University of Kentucky; Lexington, Kentucky, December 4-7, 1979).

Owens, L. V. 1963. Iron and manganese removal by split flow treatment. Water and Sewage Works 110:R76-R87.

Sienko, M. J. and R. A. Plane. 1966. Chemistry: principles and properties. p. 418-421. McGraw-Hill, New York, NY.

Stumm, W. and R. Giovanoli. 1976. On the nature of particulate Mn in simulated lakewaters. Chimia 30:423-425.

Stumm, W. and J. J. Morgan. 1981. Aquatic Chemistry. Wiley-Interscience, New York, NY. 780 p.

U.S. Code of Federal Regulations. 1985a. Title 30--Mineral Resources; Chapter VII--Office of Surface Mining Reclamation and Enforcement, Department of Interior; Subchapter B--Initial Program Regulations; Part 715--General Performance Standards. July 1, 1985.

U.S. Code of Federal Regulations. 1985b. Title 40--Protection of the Environment; Chapter 1--Environmental Protection Agency; part 434--Coal Mining Point Source Category; Subpart C--Acid or Ferruginous Mine Drainage; July 1, 1985.

U.S. Environmental Protection Agency. 1983. Design Manual: Neutralization of Acid Mine Drainage. EPA-600/2-83-001. U.S. Environmental Protection Agency, Cincinnati, OH. 231 p.

Watzlaf, G. R. 1985. Comparative tests to remove manganese from acid mine drainage. p. 41-47. In Control of Acid Mine Drainage, Bureau of Mines IC 9027.

This page intentionally left blank.

# ENGINEERING EVALUATION OF WASTE PHOSPHATIC CLAY FOR PRODUCING LOW PERMEABILITY BARRIERS[1]

by

John J. Bowders, Jr. and Eric D. Chiado[2]

Abstract. The findings of a laboratory study in which the feasibility of using waste phosphatic clay as a low permeability barrier was evaluated are reported herein. The ability of the waste phosphatic clay to chemically ameliorate any AMD produced was also assessed and the engineering parameters most important to achieving a successful barrier were determined. Two mine spoils, one cover soil and one phosphatic clay sample were used to construct the test specimens used in this study. Experiments consisted of performing permeability tests on different configurations of these soils, using rigid-wall, double-ring permeameters. Analyses of effluents for various water quality parameters (typical of mining requirements) were also performed. Three different methods of employing the phosphatic clay were evaluated including; placing the clay into a slurry and then applying it over top of the spoil, mixing dry phosphatic clay with the cover soil and then applying it over top of the spoil, and mixing dry clay directly with the spoil. Results indicated that the optimum method for reducing permeability and improving effluent quality was to mix the phosphatic clay directly with the spoil. Using this configuration, average coefficients of permeability were 50 to 5,400 times lower than the average coefficient of permeability of the spoil alone. Improvements in water quality were also substantial, with reductions in heavy metals and acidity averaging approximately 95 percent, relative to effluents from spoil alone. Compaction of individual layers had a dominating effect on permeability. The lowest coefficients of permeability were demonstrated in specimens in which all layers were compacted. Compaction also appeared to enhance the ameliorative characteristics of the clay by allowing increased residence times for the drainage to react with the phosphate in the spoil. Finally, 10 percent replacement by weight of the phosphatic clay in either the cover soil or the spoil layer produced the best results, relative to other replacement percentages.

Additional Key Words: Hydraulic Conductivity, Permeability, Acid Mine Drainage, Effluent Analyses, Water Quality, Slurry, Phosphatic Clay, Waste Materials

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]John J. Bowders, Jr. is an Assistant Professor, Department of Civil Engineering, West Virginia University, Morgantown, WV, 26506. Eric d. Chiado is a Geotechnical Engineer, Almes and Associates, Trafford, PA 15085.

## Introduction

In northern West Virginia and southwestern Pennsylvania, mining bituminous coal provides much of the economic base. Any threat to this industry poses a similar and severe threat to the economic lifeline of these areas. With the ever-increasing trend of environmental awareness, one outcome of the mining in these areas looms as a potential for limiting coal production. This outcome, acid mine drainage (AMD), is especially acute in these areas. As the environmental focus on water quality increases, mining companies will be increasingly regulated to reduce the amount of AMD which exits from their operations.

Although there are numerous techniques that are used to control the formation or treatment of AMD, the effectiveness and cost of these methods vary over a wide range. Thus, there remains a need for the development and implementation of methods which would maximize the effectiveness and minimize the cost. In the case of surface mines, active or abandoned, one such technique which shows significant promise is that of waste phosphatic clay barriers.

Waste phosphatic clay is produced during the preparation of fertilizer from phosphate ore. The disposal of this waste clay presents a significant problem to the phosphate producers located in the southeastern United States. It is estimated that by the year 2000, there will be over 1 billion tons of this waste. Areas used to contain the waste clay are rendered unusable until such time as the clays stabilize. This can be as long as 20 to 30 years.

To significantly reduce the environmental impact of these waste clays requires that any beneficial use must result in the utilization of a large volume of the clays. Such a use might be achieved by constructing hydraulic barriers from the waste clay, which could be applied at surface mine sites where AMD is present.

Past researchers (Renton and Stiller 1989) found that phosphatic clay has the potential to chemically ameliorate AMD. A secondary finding was that the clay reduced the volume of flow through the acidic material when they were mixed together. Thus, phosphatic clay may offer a "two-pronged" remedy to the AMD problem by physically limiting the quantity produced, and chemically ameliorating any AMD that is produced.

The objectives of the engineering evaluation reported herein were to: (1) determine the feasibility of using the waste phosphatic clay as a low permeability hydraulic barrier and (2) determine those engineering parameters most important to achieving this goal. The ability of the waste phosphatic clay to chemically ameliorate any AMD produced was also characterized. In addition, a computer model was used to simulate various arrangements of waste phosphatic clay barriers using the properties determined in the first two objectives. The results of the computer simulation are discussed in detail elsewhere (Bowders 1989).

## Experimental Program

Two acidic spoils, one cover soil, and one phosphatic clay were used in the testing program. Rigid-wall, double-ring permeameters were used to measure the permeability of the phosphatic clay barrier configurations. Effluent collected from the permeability tests was analyzed for constituents common to AMD.

Four factors were investigated to determine their effect on producing a low permeability, chemically enhanced system. These factors included: methods of employing the phosphatic clay (dry or slurry), location of phosphatic clay in the system by itself, mixed with the spoil, or the cover) and percentage by weight of phosphatic clay included in the system.

Soil and Waste Materials. The spoil materials used in this study were obtained from a surface mine and a coal refuse disposal facility in central and northern West Virginia, respectively. No testing was performed to exclusively determine the mineralogical composition of the spoils; however, the analysis of effluent from each spoil showed both to have high acid producing potential. The cover soil was also obtained from the surface mine. Test results indicated that the cover would not contribute to the chemical amelioration of AMD. The phosphatic clay was obtained from Mulberry, Florida. It is predominantly smectitic clay with calcium being the only detectable metal in effluents from the clay alone. The index properties of the spoils, cover soil and clay are presented in Table 1.

Specimen Preparation. Three different configurations were used to evaluate the phosphatic clay barriers (Fig. 1). Each configuration produced a layered system which represented the hydraulic barrier similar to what would exist at a reclaimed surface mine site. The four soils were used to construct five distinct layers: Phosphatic clay placed in slurry form (Chiado, Bowders and Scencindiver 1988), cover soil mixed with phosphatic clay, cover soil alone, spoil mixed with phosphatic clay, and spoil alone.

The spoil and cover layers were either uncompacted or compacted to standard Proctor energy (12,375 ft-lb/ft$^3$). For spoil or cover soil which contained dry phosphatic clay, the clay was added at either 5, 10, or 20 percent by weight. Placement gravimetric water contents of the slurries ranged from 150 to 375

percent. The configurations of all the tests are given in Table 2.

Permeability Testing. The major objective of this work was to determine if a suitable low permeability barrier could be developed using the phosphatic clay. Thus, permeability testing was a major emphasis of the work. Rigid-wall, double-ring permeameters (Fig. 2) were used to measure the permeability. Using these permeameters permitted the detection of any "side-wall" leakage, a sometimes common problem with rigid-wall permeameters (Daniel, Anderson and Boynton 1985).

All of the specimens were permeated with distilled water. Hydraulic gradients of 10 to 200 were used throughout the testing. The selection of a gradient was based on the need to expedite the testing process.

The permeability was calculated using Darcy's law (Eq. 1). The volume of flow into the specimen over a known period of time was used to compute permeability.

$$K = Q/(t\ i\ A) \qquad (1)$$

where: $K$ = coefficient of permeability ($L^3/T$)
$Q$ = quantity of inflow ($L^3$)
$t$ = time for flow Q to occur (T)
$i$ = hydraulic gradient (dimensionless)
$A$ = cross-sectional area of flow ($L^2$)

Constant-head permeability tests were used throughout the program. Tests were ended after two pore volumes of flow had passed through the specimen and the coefficient of permeability had achieved equilibrium. Effluent from each permeability test was collected. periodically, samples of the effluent were analyzed to determine the concentration of iron, manganese, magnesium, aluminum, calcium, and sulfate. The acidity, pH and electrical conductivity of each effluent sample were also measured. For the permeability tests in which the phosphatic clay layer was present from the beginning of the test, effluent samples were taken from near the initial and final pore volumes of flow in order to ascertain the effect of time on the chemistry of the system.

## Results and Discussion

Results of thirty-four permeability tests are reported in this article. The tests are grouped according to the phosphatic clay placement scheme. The results and ensuing discussion are in the following order: test using phosphatic clay slurry, phosphatic clay mixed with cover soil, and phosphatic clay mixed with spoil material. Permeability results are considered first followed by the effluent chemical analysis results.

Permeability Results. The results of the permeability tests in which the phosphatic clay was placed in the form of a slurry are shown in Table 3. With the exception of tests 5(1), 5(2), and 7(2), the addition of phosphatic clay slurry decreased the permeability of the layered system. The magnitude of these decreases was not consistent. The slurry appreciably decreased the permeability of the systems containing the uncompacted spoil, but were relatively ineffective when applied to compacted spoil. However, the lowest permeabilities were demonstrated in systems where the spoil was compacted. This result was attributed to the compaction and not the presence of the slurry. When applied to uncompacted samples,the low water content slurries produced the largest change in system permeability. The thickness of the slurry did not appear to be an important variable in producing low permeability systems. However, hairline cracking was found in several specimens. Thicker slurry layers may reduce the event of crack propagation through the entire slurry layer.

Results from the tests in which dry phosphatic clay was mixed with the cover soil and applied overtop of the spoil are presented in Tables 4 and 5. The spoil layer was compacted in all of these tests. The results in Table 4 are for incremental tests, i.e., the permeablility of the spoil was measured, then the cover/clay mixture was added and the permeability of the system was measured. The addition of the uncompacted clay/cover material to Spoil 1 resulted in a reduction of the permeability on average by a factor of three. This was only slightly more effective than the addition of the slurry to the acidic systems. The addition of uncompacted clay/cover to Spoil 2 resulted in an increase in the permeability of the system. No explanation exists for this behavior. Thus, the addition of uncompacted clay/cover to the acidic system is deemed ineffective.

The results presented in Table 5 show the effect of adding compacted clay/cover above the spoil layer. Permeabilities of the system were reduced by more than two orders-of-magnitude when compared to the uncompacted clay/cover system. Perhaps more important is the fact that when compared to the average permeability of the compacted spoil alone, the results of the compacted clay/cover represented a 30-fold decrease. Thus, even for a compacted spoil, the addition of the compacted clay/cover substantially reduced the system permeability.

Within the scope of clay/cover mixtures, the percentage of clay was varied from 5 to 20 percent by weight. Permeability results indicated that the 10 percent replacement resulted in the lowest permeabilities.

The final configuration of the barrier system included mixing dry phosphatic clay directly with the acidic spoil and compacting it. The results of the permeability tests are listed in Table 6. Addition of the dry phosphatic clay to the spoil material produced the lowest permeabilities of any of the systems investigated in this project. The average permeability of Spoil 1 was found to be $3.3 \times 10^{-8}$ cm/sec. It appeared that this method of using the clay provided the best means of reducing the permeability of an acidic system.

Effluent Analyses. Only effluents from two of the eleven permeability tests which included slurry were analyzed for quality. The results are shown in Table 7. The addition of the phosphatic clay slurry to the acidic system significantly improved the effluent water quality. All of the metals present in the pre-slurry effluent analysis showed drastic reductions in concentration after the slurry was added to the system. The only increase in concentration was for the case of calcium which was expected since the other metals replace the calciums on the phosphate radical. The pH of the effluents were not significantly altered by the presence of the clay slurry. Even though the slurry was placed in an individual layer, it was still able to exert a positive influence on the chemistry of the system.

Results of the effluent analyses from the tests with the clay mixed with the cover soil are shown in Tables 8 and 9. The incremental tests (Table 8) indicated large decreases in the iron and manganese effluent concentrations. The calcium concentrations increased while the pH again remained unchanged; however, the acidity levels were sharply reduced. This indicates that the effluent could be buffered to a neutral pH without much difficulty relative to that for an effluent from a system without phosphatic clay.

In the one-step tests (Table 9), the phosphatic clay was present from the beginning of permeation. The concentrations of the metals in the effluent were reduced to levels more near to those of the slurry arrangement. Thus, it appears beneficial to include the phosphatic clay from the onset of permeation. The passage of large pore volumes of water through the specimens did not appear to exhaust the capacity of the phosphatic clay to contain the metals from the effluent. Even after 18 pore volumes of flow the metals concentrations were low as illustrated by test number 11(2) in Table 9.

Results of the analyses of the effluents from the permeability tests on phosphatic clay mixed with the acidic spoil material are listed in Table 10. This system appeared to optimize the ameliorative capacity of the phosphate as evidenced by the reduced concentration of metals, sulfates, and acidity relative to tests in which the clay was used in other manners. Compaction of the layer did not reduce the capacity of the clay to improve the effluent quality. As the quantity of clay was increased, the quality of the effluent continued to improve; however, the amount of improvement decreased for increasing quantities of clay. The optimum clay addition was about 10 percent by weight.

## Conclusions

The findings of this research program have shown that waste phosphatic clay can be effective in reducing the permeability and improving the quality of the effluent water of an acidic system. The following conclusions are based on these findings:

1. Reduction in permeability varied depending on the method of application and location of the clay in the system.

2. Slurries of phosphatic clay were ineffective in significantly reducing the permeability of acidic systems.

3. The addition of dry phosphatic clay to either the cover soil or the spoil material produced systems with exceptionally low permeabilities.

4. Compaction appeared to be the most dominant engineering parameter for reducing permeability in these systems.

5. The inclusion of phosphatic clay provided further reductions in the coefficient of permeability even in those systems that were compacted.

6. A 10 percent by weight replacement of phosphatic clay in either the cover soil or spoil provided the optimum configuration for reducing the permeability.

In regard to the quality of the effluent water, the following conclusions are advanced:

1. All methods of phosphatic clay application produced significant reductions in metal, sulfate, and acidity concentrations.

2. Mass transfer of phosphate did not appear to be inhibited when dry phosphatic clay was used.

3. Compaction of the layers enhanced the clay's ameliorative capacity by creating longer residence times through reduced permeability.

4. Although the effective chemical life of the phosphatic clay was not determined, the movement of water through the different systems did not reduce the effectiveness of the clay over time, in any test.

5. The incorporation of larger percentages of clay in the system resulted in better water quality.

The most efficient use of the physical and chemical properties of the phosphatic clay occurred in systems containing a compacted layer of dry clay mixed directly with the spoil material. This configuration produced systems with average permeabilities 5,400 times lower than the average coefficient of permeability for uncompacted spoil alone, and 50 to 300 times lower than the average permeability for compacted spoil alone. These exceptionally low permeabilities were matched by substantial improvements in effluent quality. Generally, metal concentrations were as low or lower than those produced by other system configurations, and acidity concentration was greatly reduced relative to other systems.

Finally, it must be noted that these results were obtained from controlled laboratory tests. As such, the coefficients of permeability and chemical characteristics may not be exactly duplicated in a field setting. Attention to engineering detail during the construction of a barrier employing phosphatic clay could help to achieve similar results. Careful control of compactive effort, adequate mixing of the clay and spoil, obtaining the proper barrier water content and thickness, and limiting dessication, will be required to produce a phosphatic clay barrier which satisfies its intended function; namely substantially reducing the volume of flow through the backfill spoil, and improving the quality of the seepage that is generated.

## Acknowledgements

The authors are extremely indebted to their co-investigator Professor John C. Sencindiver of the Plant and Soil Sciences Div. at West Virginia University. His help and encouragement are greatfully appreciated. The work was accomplished through funds provided by the West Virginia Water Research Institute and the National Mine Land Reclamation Center. Continued interest in this work by Dr. Robert Jenkins and Dr. Paul Ziemkiewicz of those organizations, respectively, is sincerely appreciated.

## References

Bowders, J.J. 1989 Computer Simulation of Phosphatic Clay Barriers for Control of AMD. Article under preparation for publication. Dept. of Civil Engineering, West Virginia University, Morgantown, WV.

Chiado, E.D., J.J. Bowders and J.C. Sencindiver 1988 Phosphatic Clay Slurries for Reducing AMD from Reclaimed Mine Sites. Proc., Mine Drainage and Surface Mine Reclamation, Vol I, Conf. Pittsburgh, PA, April 19-21 pp. 44-51.

Daniel, D.E., D.C. Anderson and S.S. Boynton 1985 p. 44-51. Fixed-Wall Versus Flexible-Wall Permeameter. Hydraulic Barriers in Soil and Rock, ASTM STP 874, p 107-126.

Renton, J.J. and A.H. Stiller 1989 The Use of Phosphatic Materials as Ameliorants for Acid Mine Drainage, Vol. I. The Use of Rock Phosphate (apatite) for the Amelioration of Acid Mine Drainage from the Mining of Coal. Publication No. 02-050-072, Florida Institute of Phosphate Research, Bartow, FL, February, 1989.

Table 1 Index Properties of Soils Used in Research

| SOIL | Gs | NAT w (%) | OPT w (%) | MAX DRY UNIT WGHT (pcf) | ATTERBERG LIMITS LL | PL | PI | GRAIN SIZE PARAMETERS Cu | Cz | CLAY SIZE FRACTION (%) | UNIFIED SOIL CLASS |
|---|---|---|---|---|---|---|---|---|---|---|---|
| PHOS CLAY | 2.80 | 31 | 10 | 75 | 137 | 47 | 90 | 8.6 | 1.0 | 95 | CH |
| SPOIL 1 | 2.62 | 4 | * | 107 | 29 | 19 | 10 | 20.8 | 1.2 | 0 | GC |
| SPOIL 2 | 2.32 | 18 | * | 97 | 36 | 26 | 10 | 7.6 | 0.9 | 3 | GC |
| COVER | 2.79 | 13 | 18 | 105 | 34 | 22 | 12 | 14.3 | 1.8 | 5 | SC |

* Optimum water content was not determined for either spoil 1 or 2 because the addition of water to these soils could have resulted in the formation of AMD

Table 2 Arrangement and nature of each test specimen

| TEST | LAYER ONE | LAYER TWO | LAYER THREE | COMMENTS |
|---|---|---|---|---|
| 1(1) | UNCOMPACTED SPOIL 1 | | | |
| 1(2) | UNCOMPACTED SPOIL 1 | | | |
| 2(1) | PHOS CLAY SLURRY | | | WATER CONTENT = 179% THICKNESS = 6.6 cm |
| 2(2) | PHOS CLAY SLURRY | | | WATER CONTENT = 194% THICKNESS = 7.2 cm |
| 2(3) | PHOS CLAY SLURRY | | | WATER CONTENT = 329% THICKNESS = 2.6 cm |
| 2(4) | PHOS CLAY SLURRY | | | WATER CONTENT = 386% THICKNESS = 4.3 cm |
| 3(1) | COMPACTED SPOIL 1 | PHOS CLAY SLURRY | | WATER CONTENT = 274% THICKNESS = 3.1 cm |
| 3(2) | COMPACTED SPOIL 1 | PHOS CLAY SLURRY | | WATER CONTENT = 353% THICKNESS = 3.7 cm |
| 4(1) | UNCOMPACTED SPOIL 1 | PHOS CLAY SLURRY | | WATER CONTENT = 172% THICKNESS = 3.4 cm |
| 4(2) | UNCOMPACTED SPOIL 1 | PHOS CLAY SLURRY | | WATER CONTENT = 255% THICKNESS = 3.9 cm |
| 5(1) | COMPACTED SPOIL 1 | PHOS CLAY SLURRY | | WATER CONTENT = 187% THICKNESS = 1.77 cm |
| 5(2) | COMPACTED SPOIL 1 | PHOS CLAY SLURRY | | WATER CONTENT = 290% THICKNESS = 2.64 cm |
| 6(1) | UNCOMPACTED SPOIL 1 | PHOS CLAY SLURRY | | WATER CONTENT = 187% THICKNESS = 1.27 cm |
| 6(2) | UNCOMPACTED SPOIL 1 | PHOS CLAY SLURRY | | WATER CONTENT = 290% THICKNESS = 2.1 cm |
| 7(1) | COMPACTED SPOIL 1 | PHOS CLAY SLURRY | LOOSE COVER SOIL | WATER CONTENT = 186% THICKNESS = 3.3 cm |
| 7(2) | COMPACTED SPOIL 1 | PHOS CLAY SLURRY | LOOSE COVER SOIL | WATER CONTENT = 276% THICKNESS = 3.8 cm |
| 7(3) | UNCOMPACTED SPOIL 1 | PHOS CLAY SLURRY | LOOSE COVER SOIL | WATER CONTENT = 272% THICKNESS = 3.4 cm |
| 8(1) | COMPACTED SPOIL 1 | UNCOMP CLAY/COVER | | 5% CLAY |
| 8(2) | COMPACTED SPOIL 1 | UNCOMP CLAY/COVER | | 10% CLAY |
| 8(3) | COMPACTED SPOIL 1 | UNCOMP CLAY/COVER | | 20% CLAY |
| 10(1) | COMPACTED SPOIL 2 | UNCOMP CLAY/COVER | | 5% CLAY |
| 10(2) | COMPACTED SPOIL 2 | UNCOMP CLAY/COVER | | 10% CLAY |
| 10(3) | COMPACTED SPOIL 2 | UNCOMP CLAY/COVER | | 20% CLAY |
| 11(1) | DESTROYED | | | |
| 11(2) | COMPACTED SPOIL 1 | COMP CLAY/COVER | | 10% CLAY |
| 11(3) | COMPACTED SPOIL 1 | COMP CLAY/COVER | | 20% CLAY |
| 12(1) | COMPACTED SPOIL 2 | COMP CLAY/COVER | | 5% CLAY |
| 12(2) | COMPACTED SPOIL 2 | COMP CLAY/COVER | | 10% CLAY |
| 12(3) | COMPACTED SPOIL 2 | COMP CLAY/COVER | | 20% CLAY |
| 13(1) | COMP CLAY/SPOIL 1 | COMP COVER SOIL | | 5% CLAY |
| 13(2) | COMP CLAY/SPOIL 1 | COMP COVER SOIL | | 10% CLAY |
| 13(3) | COMP CLAY/SPOIL 1 | COMP COVER SOIL | | 20% CLAY |
| 14(1) | COMP CLAY/SPOIL 2 | COMP COVER SOIL | | 5% CLAY |
| 14(2) | COMP CLAY/SPOIL 2 | COMP COVER SOIL | | 10% CLAY |
| 14(3) | COMP CLAY/SPOIL 2 | COMP COVER SOIL | | 20% CLAY |

Table 3 Summary of Permeability Tests With Phosphatic Clay Slurry.

| TEST (1) | SOIL LAYERS (2) | COEFFICIENT OF PERMEABILITY $k_{sp}$ (cm/s) (3) | $k_{sp-pcs}$ (cm/s) (4) | $k_{sp-pcs-cs}$ (cm/s) (5) | k(3) k(4) (6) | k(3) k(5) (7) |
|---|---|---|---|---|---|---|
| 1(1) | USP | 1.7E-4 | na | na | na | na |
| 1(2) | USP | 1.1E-4 | na | na | na | na |
| 2(1) | PCS | 1.5E-6 | na | na | na | na |
| 2(2) | PCS | 1.8E-6 | na | na | na | na |
| 2(3) | PCS | 3.7E-6 | na | na | na | na |
| 2(4) | PCS | 5.2E-6 | na | na | na | na |
| 3(1) | CSP-PCS | 2.1E-5 | 5.8E-6 | na | 3.6 | na |
| 3(2) | CSP-PCS | 2.2E-5 | 5.5E-6 | na | 4.0 | na |
| 4(1) | USP-PCS | 2.4E-4 | 1.6E-5 | na | 15.0 | na |
| 4(2) | USP-PCS | 1.4E-4 | 5.2E-5 | na | 2.7 | na |
| 5(1) | CSP-PCS | 3.2E-6 | 4.8E-6 | na | 0.7 | na |
| 5(2) | CSP-PCS | 3.6E-6 | 4.3E-6 | na | 0.8 | na |
| 6(1) | USP-PCS | 2.4E-4 | 7.9E-6 | na | 30.1 | na |
| 6(2) | USP-PCS | 1.4E-4 | 3.2E-5 | na | 4.2 | na |
| 7(1) | CSP-PCS-UCS | 7.9E-6 | 4.4E-6 | 1.5E-6 | 1.7 | 5.4 |
| 7(2) | CSP-PCS-UCS | 7.3E-6 | 7.6E-6 | 4.4E-7 | 0.9 | 16.6 |
| 7(3) | USP-PCS-UCS | 1.8E-4 | 3.7E-5 | 6.6E-7 | 4.8 | 274.4 |

Key to Soil Layers
USP: Uncompacted Spoil
CSP: Compacted Spoil
PCS: Phosphatic Clay Slurry
UCS: Uncompacted Cover Soil

Table 4 Summary of Incremental Permeability Tests With Phosphatic Clay/Cover Soil Mix.

| TEST (1) | SOIL LAYERS (2) | COEFFICIENT OF PERMEABILITY $k_{spoil}$ (cm/s) (3) | $k_{spoil/phos\ clay-cover}$ (cm/s) (4) | k(3) k(4) (5) |
|---|---|---|---|---|
| 9(1) | CSP1-UPCC | 1.2E-5 | 8.1E-6 | 1.5 |
| 9(2) | CSP1-UPCC | 1.2E-5 | 2.4E-6 | 4.8 |
| 9(3) | CSP1-UPCC | 9.2E-6 | 1.3E-6 | 2.2 |
| 10(1) | CSP2-UPCC | 4.3E-6 | 1.1E-5 | 0.4 |
| 10(2) | CSP2-UPCC | 2.7E-6 | 3.8E-5 | 0.1 |
| 10(3) | CSP2-UPCC | 1.6E-6 | 1.5E-5 | 0.1 |

Key to Soil Layers
CSP1: Compacted Spoil 1
CSP2: Compacted Spoil 2
UPCC: Uncompacted Phosphatic Clay/Cover Soil Mix

Table 5 Summary of One-Step Permeability Tests With Phosphatic Clay/Cover Soil Mix

| TEST (1) | SOIL LAYERS (2) | COEFFICIENT OF PERMEABILITY $k_{spoil/phos\ clay-cover\ mix}$ (cm/s) (3) | $\frac{USP}{k(3)}$ (4) | $\frac{CSP}{k(3)}$ (5) |
|---|---|---|---|---|
| 11(1) | DESTROYED | - | - | - |
| 11(2) | CSP1-CPCC | 7.3E-7 | 245 | 15 |
| 11(3) | CSP1-CPCC | 3.4E-8 | 5,220 | 323 |
| 12(1) | CSP2-CPCC | 3.4E-8 | na | 84 |
| 12(2) | CSP2-CPCC | 3.2E-8 | na | 90 |
| 12(3) | CSP2-CPCC | 2.2E-7 | na | 13 |

Key to Soil Layers
CSP1: Compacted Spoil 1
CSP2: Compacted Spoil 2
CPCC: Compacted Phosphatic Clay/Cover Soil Mix

Average Coefficients of Permeability Used to Obtain Ratios in Columns 4 and 5:
Uncompacted Spoil 1 = 1.8E-4 cm/sec
Compacted Spoil 1 = 1.1E-5 cm/sec
Compacted Spoil 2 = 2.9E-6 cm/sec

Note: No coefficient available for uncompacted spoil 2 because it was not tested in this state.

Table 6 Summary of One-Step Permeability Tests With Phosphatic Clay/Spoil Mix.

| TEST (1) | SOIL LAYERS (2) | COEFFICIENT OF PERMEABILITY $k_{phos\ clay-spoil\ mix/cover}$ (cm/s) (3) | $\frac{USP}{k(3)}$ (4) | $\frac{CSP}{k(3)}$ (5) |
|---|---|---|---|---|
| 13(1) | CSP1PC-CCS | 4.2E-8 | 4,238 | 262 |
| 13(2) | CPS1PC-CCS | 9.3E-9 | 19,160 | 1,188 |
| 13(3) | CSP1PC-CCS | 4.9E-8 | 3,670 | 227 |
| 14(1) | CSP2PC-CCS | 8.0E-8 | na | 36 |
| 14(2) | CSP2PC-CCS | 4.6E-8 | na | 62 |
| 14(3) | CSP2PC-CCS | 3.8E-8 | na | 76 |

Key to Soil Layers
CSP1PC: Compacted Spoil 1/Phosphatic Clay Mix
CSP2PC: Compacted Spoil 2/Phosphatic Clay Mix
CCS: Compacted Cover Soil

Average Coefficients of Permeability Used to Obtain Ratios in Columns 4 and 5:
Uncompacted Spoil 1 = 1.8E-4 cm/sec
Compacted Spoil 1 = 1.1E-5 cm/sec
Compacted Spoil 2 = 2.9E-6 cm/sec

Note: No coefficient available for uncompacted spoil 2 because it was not tested in this state.

Table 7 Summary of Effluent Analyses for Permeability Tests with Phosphatic Clay Slurry

| TEST | | pH | CONCENTRATIONS (mg/l) Fe | Mn | Mg | Al | Ca | $SO_4^{-2}$ |
|---|---|---|---|---|---|---|---|---|
| 3(1) | Before | 3.89 | 82 | 248 | 557 | 623 | 249 | 9,320 |
| | After | 5.73 | 6 | 17 | 80 | 6 | 658 | 2,145 |
| | % Change | | (93) | (86) | (86) | (99) | (164) | (77) |
| 3(2) | Before | 4.14 | 186 | 271 | 586 | 344 | 299 | 8,580 |
| | After | 4.32 | 13 | 80 | 50 | 10 | 467 | 1,833 |
| | % Change | | (93) | (70) | (91) | (97) | (156) | (79) |

Note: "Before" refers to values obtained prior to the addition of phosphatic clay slurry, while "after" refers to values obtained after the addition of phosphatic clay slurry

Table 8 Summary of Effluent Analyses for Incremental Permeability Tests With Phosphatic Clay/Cover Soil Mix

| TEST | | pH | METAL CONCENTRATIONS (mg/l) Fe | Mn | Ca | $SO_4^{-2}$ | ACIDITY pH7 (mg $CaCO_3$/l) | pH8.3 | ELEC COND |
|---|---|---|---|---|---|---|---|---|---|
| 9(1) | B | 3.60 | 540 | 406 | 430 | 15,537 | 7,587 | 8,625 | 10.0 |
| | A | - | - | - | - | - | - | - | - |
| | % | - | - | - | - | - | - | - | - |
| 9(3) | B | 3.70 | 580 | 370 | 420 | 14,106 | 7,162 | 8,212 | 9.6 |
| | A | 3.58 | 320 | 51 | 1430 | 4,100 | 620 | 700 | 3.2 |
| | % | | (45) | (86) | (340) | (71) | (91) | (91) | (67) |
| 10(2) | B | 2.83 | 9940 | 42 | 550 | 16,290 | 16,837 | 17,138 | 13.6 |
| | A | 3.06 | 860 | 11 | 1340 | 4,250 | 820 | 860 | 3.7 |
| | % | | (91) | (73) | (244) | (74) | (95) | (95) | (73) |
| 10(3) | B | 2.35 | 9730 | 43 | 490 | 13,027 | 18,825 | 19,488 | 16.4 |
| | A | 3.21 | 1010 | 10 | 1330 | 5,300 | 890 | 950 | 4.5 |
| | % | | (90) | (80) | (271) | (59) | (95) | (95) | (73) |

B: Before addition of phosphatic clay/cover layer
A: After addition of phosphatic clay/cover layer
%: Percent change in concentration with addition of layer

Electrical Conductivity units are mmhos/cm

Tests 9(1) and 9(2) contained spoil 1
Tests 10(2) and 10(3) contained spoil 2

Table 9 Summary of Effluent Analyses for One-Step Permeability Tests With Phosphatic Clay/Cover Soil Mix

| TEST | | pH | METAL CONCENTRATIONS (mg/l) Fe | Mn | Ca | $SO_4^{-2}$ | ACIDITY (mg $CaCO_3$/l) pH7 | pH8.3 | ELEC COND |
|---|---|---|---|---|---|---|---|---|---|
| 11(2) | E | 3.82 | 224 | 37 | - | - | 4,000 | 4,400 | - |
| | L | 3.12 | 20 | 6 | 545 | 830 | 1,750 | 1,950 | 1.5 |
| | % | | (91) | (83) | - | - | (56) | (56) | - |
| 11(3) | E | 3.79 | 414 | 410 | - | - | 30,400 | 34,700 | - |
| | L | 2.98 | 160 | 13 | 548 | 4,250 | 6,300 | 6,900 | 3.1 |
| | % | | (61) | (97) | - | - | (79) | (80) | - |
| 12(1) | E | 2.89 | 2245 | 43 | - | - | 25,600 | 27,400 | - |
| | L | 2.56 | 200 | 22 | 511 | 5,750 | 1,670 | 1,760 | 4.3 |
| | % | | (91) | (49) | - | - | (93) | (94) | - |
| 12(2) | E | 2.75 | 7719 | 90 | - | - | 73,000 | 76,500 | - |
| | L | 2.45 | 60 | 3 | 660 | 6,325 | 1,920 | 2,025 | 4.6 |
| | % | | (99) | (97) | - | - | (97) | (97) | - |
| 12(3) | E | 2.86 | 4583 | 134 | - | - | 34,700 | 40,700 | - |
| | L | 2.98 | 20 | 27 | 322 | 4,100 | 2,425 | 2,840 | 3.4 |
| | % | | (99) | (80) | - | - | (93) | (93) | - |

E: Early in test
L: Late in test
%: Percent change in concentration over time

Electrical Conductivity units are mmhos/cm

Tests 11(2) and 11(3) contained spoil 1
Tests 12(1), 12(2), and 12(3) contained spoil 2

Table 10 Summary of Effluent Analyses for One-Step Permeability Tests With Phosphatic Clay/Spoil Mix

| TEST | | pH | METAL CONCENTRATIONS (mg/l) Fe | Mn | Ca | $SO_4^{-2}$ | ACIDITY (mg $CaCO_3$/l) pH7 | pH8.3 | ELEC COND |
|---|---|---|---|---|---|---|---|---|---|
| 13(1) | E | 3.23 | 80 | 173 | 354 | 7,575 | 5,100 | 8,200 | 7.6 |
| | L | - | - | - | - | - | - | - | - |
| | % | | - | - | - | - | - | - | - |
| 13(2) | E | 3.91 | - | - | - | 10,000 | - | - | 8.9 |
| | L | - | - | - | - | - | - | - | - |
| | % | | - | - | - | - | - | - | - |
| 13(3) | E | 7.23 | - | - | 820 | 4,900 | 0 | 20 | 5.6 |
| | L | 6.10 | 210 | 37 | 1050 | 3,400 | 0 | 20 | 3.5 |
| | % | | - | - | (128) | (31) | NONE | NONE | (39) |
| 14(1) | E | 3.97 | 980 | 57 | 490 | 14,100 | 1,885 | 2,040 | 10.0 |
| | L | 2.20 | 320 | 23 | 1140 | 2,920 | 145 | 163 | 3.4 |
| | % | | (67) | (59) | (233) | (79) | (92) | (92) | (66) |
| 14(2) | E | 3.19 | 330 | 36 | 540 | 8,850 | 935 | 1,030 | 7.5 |
| | L | 3.10 | 20 | 18 | 1050 | 2,920 | 170 | 185 | 3.1 |
| | % | | (94) | (50) | (194) | (67) | (82) | (82) | (59) |
| 14(3) | E | 5.12 | - | 13 | 810 | 5,750 | 25 | 25 | 4.9 |
| | L | 3.10 | 23 | 11 | 1020 | 2,660 | 15 | 15 | 2.9 |
| | % | | - | (12) | (126) | (54) | (40) | (40) | (42) |

E: Early in test
L: Late in test
%: Percent change in concentration over time

Electrical Conductivity units are mmhos/cm

Tests 13(1) through 13(3) contained spoil 1
Tests 14(1) through 14(3) contained spoil 2

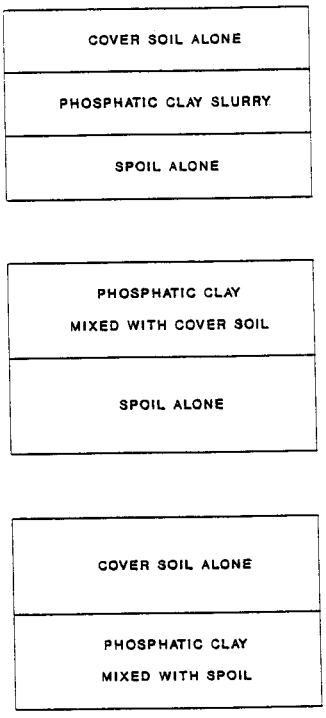



Figure 1 Physical arrangement of samples used in study

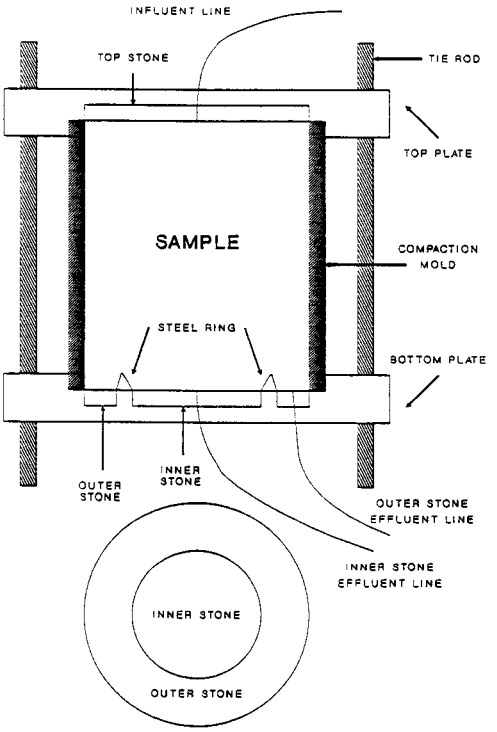



Figure 2 Schematic diagram of fixed-wall double-ring permeameter

# A COMPARATIVE STUDY OF TAILINGS ANALYSIS USING ACID/BASE ACCOUNTING, CELLS, COLUMNS AND SOXHLETS[1]

William S. Bradham and Frank T. Caruccio[2]

**Abstract**. The overburden analytical techniques involving acid/base accounting, weathering cells, columns and soxhlet reactors were tested and compared using several tailings samples having a variety of NP/S ratios. Tailings were used to normalize the effect that particle size has on leachate quality and were characterized by acid/base accounting data. The results showed that samples with high NP's and high S percentage gave similar results in all testing procedures. For the weathering cells and soxhlet tests it was shown that the leaching technique, in addition to producing a unique quality leachate, modified the sample chemistry that caused shifts in acid/alkaline production. The fine-grained nature of the tailings afforded high specific retention of fluid, creating airlocks within the columns that skewed the results. When the leachate quality predicted and obtained by the various tests was compared to the actual drainage quality emanating from the sites from which the samples were collected, the least accurate were the column tests, next were the acid/base accounting projections, then the soxhlets, with the weathering cell tests being most accurate.

Additional Key Words: Acid mine drainage, overburden analyses, mine drainage quality prediction.

## Introduction

The acid production potential of a sample is generally assessed through mine waste/overburden analyses, that can be grouped into two broad categories, static or dynamic techniques. In the static tests, whole rock analyses are used to predict mine drainage quality, the assumption being made that specific minerals comprising the mine waste/overburden will react with water to produce varying degrees of alkalinity and acidity. Alternatively, dynamic tests empirically determine leachate quality by subjecting the samples to simulated weathering conditions while monitoring the quality of the effluent produced. Within these two categories a variety of mine waste/overburden analytical techniques are available, each having advantages and disadvantages relative to the others.

Through a computer literature search (through 1986), examining 679 titles and 36 abstracts dealing with various aspects of mine drainage quality and predictive methods, we identified several mine waste/overburden analytical techniques that are used in the United States and Canada, both in the coal fields and sulfide mines. The methods include

[1]Paper presented at 1990 Mining and Reclamation Conference and Exhibition, Charleston, WV, April 23-26, 1990. Publication in this Proceedings does not preclude authors from publishing their manuscripts, whole or in part, in other publication outlets.

[2]William S. Bradham is Graduate Research Assistant and Frank T. Caruccio is Professor of Geology at the Department of Geological Sciences, University of South Carolina, Columbia, SC 29208

simulated weathering chamber tests (column and humidity cells), soxhlet reactors (infrequently used), biological confirmation tests (commonly referred to as B.C. Bacterial test), whole rock analyses (acid/base accounting, extensively used in coal fields), and B.C. Research Initial test (extensively used for base metal and gold mines in Canada and western United States). A test involving the peroxide oxidation of the sample has been recently published but lacks widespread usage. The tests that are most commonly used are described in Table 1.

Many questions exist as to the accuracy of the testing procedures and the ability of the tests to adequately project the long-term (decades) chemical weathering attributes of mine waste/overburden material based on laboratory results obtained in relatively short time periods (days to weeks). For example, fine grained particles may afford the sample with a large water holding capacity (specific retention) and create an "air lock" that will effectively inhibit the transfer of oxygen required for pyrite oxidation. In addition, the variety of environmental conditions relative to climate, mining operation, enrichment or processing methods and manner of disposal affect the oxidation of the sulfide and the manner by which the weathering products are mobilized from the site (Erickson and Hedin, 1988, Perry, 1985, Sullivan and Sobek, 1988).

In previous investigations, Caruccio and Geidel (1986) found the column tests to approximate the quality of the leachate derived from field tests using tubs. The same study showed the humidity cell tests to overestimate the acidity, while the soxhlet tests greatly overestimate the acidity. The observed variations in acid production were explained by differences in grain size mandated by a specific test. Accordingly, particle size plays a major role in acid production, with acid loads varying inversely with particle size.

To eliminate the sample size bias, the Mine Environmental Neutral Drainage (MEND of Canada Energy, Mines and Resources) tailings samples were chosen for a comparative study testing several tailings having a variety of chemical compositions. The MEND samples' relatively common particle size reduced the factors in acid production to three: percentage sulfur, neutralization potential, and leaching method.

## Materials and Methods

The samples showed much variation in chemistry (Table 2). Of interest is the ratio of acid potential (sulfur % X 31.25) to neutralization potential.

Three leaching methods were employed in the experiment: weathering cells (Caruccio 1968), columns (Hood and Oerter 1984), and soxhlet extractors (Renton et al. 1988). Ten samples were utilized and each sample was subjected in duplicate to each leaching method, for a total of 60 tests. The samples were leached on a weekly basis for approximately eight months.

The leachate samples were analyzed for pH, specific conductance, and net acidity throughout the course of the experiment. In addition, sulfate concentrations were measured for the first twelve weeks and correlated to specific conductance, in order to use conductance as a measure of sulfate load. Leachate samples for three of the samples (WEM, CUR, and BLD) were analyzed for calcium, and porosity determinations were made at the completion of the investigation.

## Results and Discussion

For each leaching method, the MEND samples produced a leachate quality reflecting the differences in controlling factors: %S, NP, and leaching method.

### Weathering Cells

Samples BLD, EQS, and NOB produced alkaline leachate through the course of the experiment, whereas samples CIN, INC, WAA, WEM, CUR, HES were acidic in nature. The HBM sample was initially acidic and became alkaline.

The first group, the alkaline producing samples, exhibited some degree of pyrite oxidation, as evidenced by moderate sulfate loads. However, the dissolution of the carbonates predominated, neutralized the acidity, and resulted in a net alkaline leachate.

The second group, the acid producing samples, exhibited the opposite behavior. These samples have low NP's and the expression of pyrite oxidation completely overwhelmed any impact the neutralization potential may have had on the leachate. Consequently, the leachate is acidic.

The HBM sample, which represents a transition between the high sulfur-low NP group and the low sulfur-high NP group, produced an interesting trend. Initially acidic despite the highest NP of any sample studied, the HBM weathering cells, after the first two weeks of leaching, gradually became a net alkaline producer. Accordingly, the HBM sample represents the transition between alkaline leachate producers and acidic leachate producers.

### Columns

The column leachings can also be grouped into distinct categories. The BLD, EQS, NOB, WEM and CUR columns were alkaline producers. The HBM, CIN, INC, WAA, and HES columns were acid producers, although the HES column did not produce as much acid as would be predicted by sample chemistry.

TABLE 1. SUMMARY OF COMMONLY USED MINE WASTE/OVERBURDEN ANALYTICAL TECHNIQUES

| Method | | Advantages | Disadvantages |
|---|---|---|---|
| Static Tests (Whole Rock Analyses) | | | |
| • Acid/Base Accounting (Sobek et al. 1978) | Whole rock analyses completed on a pulverized sample. Acid potential related to sulfur content, neutralization potential determined by hot acid digestion with HCl. | Easy to perform, quick turn-around time, useful in areas that are acid or alkaline prone. Relatively inexpensive. | Does not relate to kinetic data. Assumes parallel release of acidity and alkalinity. |
| Dynamic Tests | | | |
| • Soxhlet Reactor (Renton et al. 1988) | Leachate generated on pulverized sample which is cycled in a soxhlet reactor. During interim, sample is stored at 105° C. | Easy to perform quick turn-around time, purported kinetic data. | Expensive apparatus, extremely aggressive oxidation of sample. |
| • Humidity Cells (Caruccio 1968) | Crushed rock is placed in humidified atmosphere and leached periodically. Volume and character of leachate related to rock weight to produce alkaline/acid production potential. | Produces kinetic data, rates of acidity per unit weight of sample obtained, approximates field conditions. | Long time required, large data base generated. Relatively expensive. |
| • Column Weathering Test (Hood and Oerter 1984) | Field sample placed in large columns and leached periodically. Leachate is analyzed and related to rock weight. | Best approximator of field conditions. | Same as above, in addition, large volume of samples required, channelization problem encountered. |

TABLE 2. MEND SAMPLE CHARACTERISTICS

| Sample | % Sulfur | Neutralization Potential (p/1000) | Acid, ppt NP, ppt | CaO wt % | MgO wt % | Column Porosity (ave.) |
|---|---|---|---|---|---|---|
| HBM | 13.36 | 50.2 | 8.3 | 2.92 | 16.00 | 0.60 |
| EQS | 4.48 | 30.0 | 4.7 | 1.96 | 1.78 | 0.31 |
| NOB | 3.97 | 44.0 | 2.8 | 2.76 | 1.74 | 0.36 |
| WAA | 16.03 | 17.0 | 29 | 2.22 | 7.90 | 0.64 |
| INC | 1.14 | -5.7 | -NP | 0.52 | 4.69 | 0.63 |
| WEM | 24.58 | 7.5 | 102 | 1.20 | 2.98 | 0.38 |
| HES | 73.75 | -6.3 | -NP | 0.36 | 3.89 | 0.37 |
| CUR | 30.46 | 38.8 | 24.5 | 0.49 | 2.06 | 0.32 |
| BLD | 0.13 | 8.8 | 0.46 | 0.15 | 0.19 | 0.50 |
| CIN | 1.40 | -0.7 | -NP | 0.08 | 0.15 | 0.37 |

As in the weathering cells, the BLD, EQS, and NOB columns exhibited the predominance of NP over pyrite oxidation. Similarly, the HBM, CIN, INC, WAA, and HES column leachate indicates that dissolution of carbonates was not adequate to overcome the effects of pyrite oxidation.

Measurements of column porosity provide an explanation for the apparent contradictory behavior of the WEM and CUR columns as well as the suppression of acid production of the HES columns. The data in Table 2 reveal that the EQS, NOB, CUR, WEM, and HES columns had much lower porosities than the other columns and thus constitute a category of their own. When these columns were leached during the experiment, we noted water to pond on top of the samples and typically required seven days to drain down through the column. In addition, when the columns were dismantled at the completion of the experiment, the samples exhibited mottling and cracking, and in the case of the CUR sample, a hard rind of oxidized sample was found against the column wall, encasing the rest of the sample.

These data suggest the existence of "airlocks" within leaching column samples marked by such low porosities and permeabilities, that they effectively isolated the samples from oxygen infiltration, and thus inhibit pyrite oxidation. In samples of high neutralization potential, the dissolution of carbonates is favored by water retention, and the resulting leachate is alkaline. For samples with high sulfur and low neutralization potential, as in the case of the HES sample, the result is reduced acid loads.

Soxhlets

The soxhlet samples were stored in a drying oven at 105° C, and leached on a weekly basis in a soxhlet extractor for two hours. Following leaching, the samples were returned to the oven for storage.

The NOB and EQS samples produced alkaline leachate throughout the testing period. WAA, INC, WEM, HES, CUR, BLD, and CIN samples were acidic. Once again, the transition sample HBM, exhibited an unusual behavior by remaining alkaline for roughly twenty weeks, at which time the character of the leachate reversed itself and became acidic.

Sulfate data showed that pyrite oxidation occurred in all soxhlet leachings. Leachate quality therefore became a function of the ratio of alkaline water to acidic water produced by the leaching process. The high NP-low sulfur samples, NOB and EQS, thus exhibited alkaline leachate. Samples with high sulfur to NP ratios correspondingly produced acidic leachate.

The HBM soxhlet samples behaved in an odd manner. Initially, the leachate was alkaline to neutral. Sulfate on the order of 20,000 mg/kg, however, gave evidence that considerable pyrite oxidation was occurring. After approximately twenty weeks the leachate became acidic. There was, however, no corresponding increase in sulfate loads. In fact, sulfate loads continued along the same general decline begun after the initial leachings.

As in the other MEND samples, leachate quality is governed by the mixing of acidic water and alkaline water. Examination of the HBM soxhlet data suggests that the soxhlet extraction process actually encompasses two environments. The first, the actual extraction process, uses continuous flushings with hot water. The second environment is present in the drying oven during the period in which the sample is wet but drying.

Dissolution of calcium carbonate is enhanced during the hot extraction process. A comparison of the calcium contents of the leachates of the WEM samples for the weathering cells, columns and soxhlet tests illustrates this point (Figure 1). On the other hand, pyrite oxidation occurs minimally during extraction and is accelerated while the sample is stored in the oven. The kinetics favor the dissolution of calcium carbonate by hot condensed water while pyrite oxidation is enhanced during the time the sample is stored in the oven.

The first twenty weeks of the HBM testing data exhibit alkalinity production equalling or exceeding acid production, resulting in a net neutral or alkaline leachate. The sharp rise in acidity occurred when sufficient calcium carbonate had been leached from the system to allow the expression of the acidity reactions in the leachate. The lack of a sharp rise in sulfate corresponding to the rise in acidity indicates that the rate of acid production remains the same. What has changed, therefore, is the ratio of alkaline material to acid material.

Interestingly, after a sharp rise in acidity, the HBM sample returned to a condition in which acid production was once again balanced by alkalinity production. This suggests the possibility that the ratio of alkaline to acidic producing material has returned to one that favors neutral or alkaline leachate.

## Comparison to Field Data

Contained within the MEND report is a summary of the drainage qualities associated with each location from which the tailings samples used in this study were obtained. BLD and HBM data were solicited by telephone and the CIN data represent test pad results. The drainage quality for each of the sites is presented

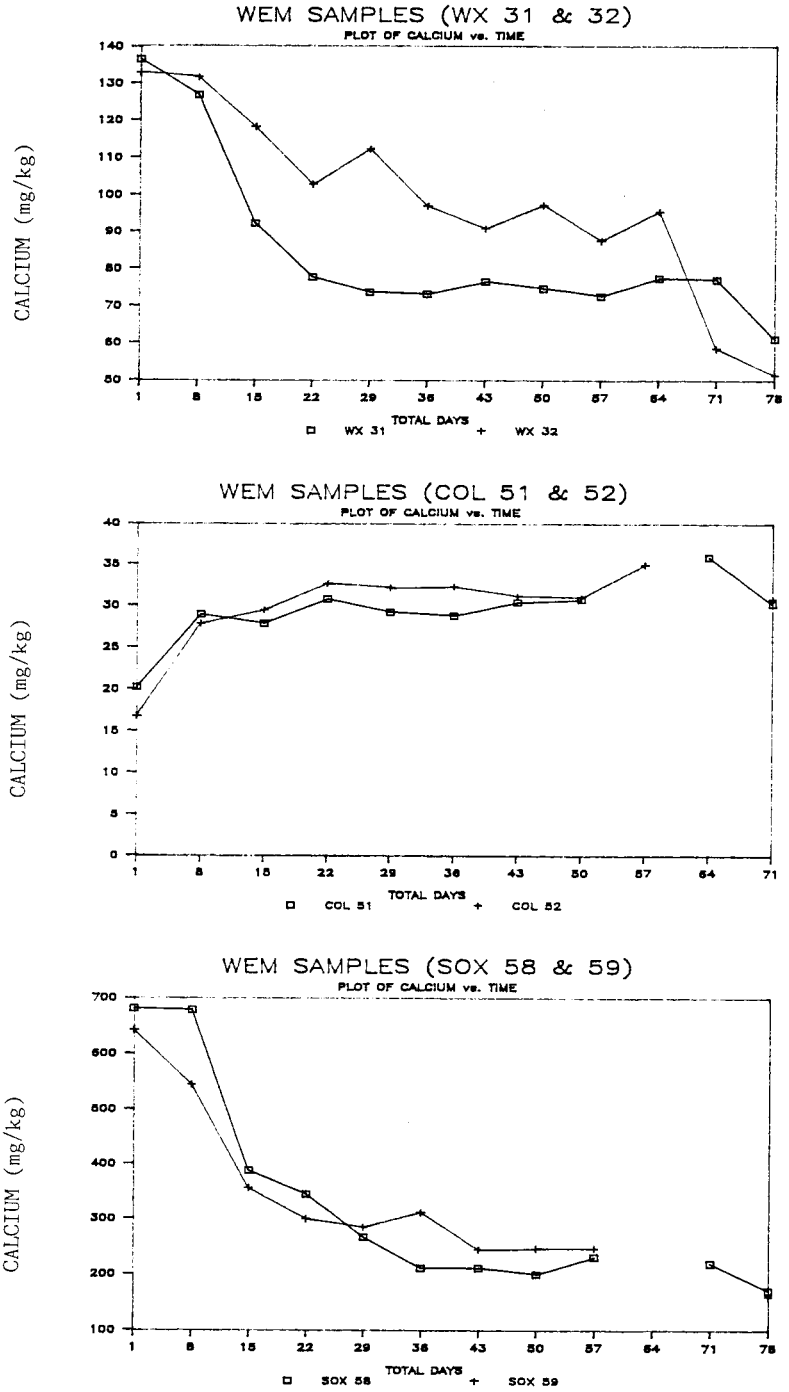



Figure 1. Relative rates of $CaCO_3$ dissolution in weathering cells (Wx), Columns (Col) and Soxhlets (Sox) for WEM sample.

in Table 3 as well as the leachate quality predicted or obtained by the various tests.

A comparison of the results shows that the acid/base accounting method correctly predicted eight out of ten, the weathering cell tests ten out of ten, the soxhlets nine out of ten and the columns seven out of ten.

## Conclusions

As evidenced by sulfate loads present even in neutral or alkaline leachates, it is apparent that leachate quality for the MEND samples varies as a function of the amount of acidic leachate produced and the amount of alkaline leachate present to neutralize. As both reactions occur in all samples, with the exception of the

Table 3. Leachate Quality Predicted/Obtained by Various Tests Compared to Field Data.

Ac = Acid Al = Alkalinity

| Sample | Acid/ Base | Cell | Column | Soxhlet | Acid/ Alk.ratio | Field Data |
|---|---|---|---|---|---|---|
| NOB | Ac | Al | Al | Al | 2.8 | Al |
| EOS | Ac | Al | Al | Al | 4.7 | Al |
| BLD | Al | Al | Al | Ac | 0.46 | Al |
| CIN | Ac | Ac | Ac | Ac | -NP | Ac* |
| INC | Ac | Ac | Ac | Ac | -NP | Ac |
| HBM | Ac | Ac/Al | Ac | Al/Ac | 8.3 | Ac |
| WAA | Ac | Ac | Ac | Ac | 29 | Ac |
| CUR | Ac | Ac | Al | Ac | 24.5 | Ac |
| WEM | Ac | Ac | Al | Ac | 102 | Ac |
| HES | Ac | Ac | Ac | Ac | -NP | Ac |

* Test Pad Results

"air locked" columns, the leachate quality depends on the balance between the amount of alkaline and acidic material present in the sample, and the rates at which both reactions occur. In turn, the leaching methods control the reaction rates.

Column tests utilizing well-sorted, coarse-grained samples (i.e., greater then 0.5 cm) most accurately resemble field test conditions. The samples are kept somewhat moist and, again excluding the "air locked" columns that develop when fine grained samples (i.e., less than 1 mm) are used, allow oxygen to permeate through most of the sample. The presence of "air locks", however, limits the transfer of oxygen required for pyrite oxidation and restricts acid production to the exterior surface of the sample.

Weathering cells, on the other hand, by providing more moisture and oxygen, as well as having a greater reactive surface area, accelerate the rate of pyrite oxidation relative to the rate of the dissolution of calcium carbonate. The result of this change in rates is a greater proportion of acidic to alkaline leachate. Further, the rate of pyrite oxidation is increased relative to the dissolution of calcium carbonate and these rate differences change sample chemistry through time. In the so called "transition" samples, such as the HBM sample, this property is manifested by the sample's migration from the field of acid producing samples to that of the alkaline producing samples.

The soxhlet extractors accelerate the dissolution of calcium carbonate relative to pyrite oxidation. The HBM soxhlet sample, therefore, migrates from the alkaline producing field to the acid producing field.

In summary, the leachate quality of the MEND sample leachings depends on the balance of the production of acid leachate to alkaline leachate. This, in turn, is a function of the amount of acid and alkaline producing materials, and of the rates of both reactions.

When compared to observed field drainage quality emanating from areas from whence the tailings samples were collected, the predictions made by the columns were least accurate (70% success), acid-base accounting next, followed by the soxhlet tests, and the weathering cell tests were most accurate (100% success).

## Literature Cited


Caruccio, Frank T., 1968. An evaluation of factors affecting acid mine drainage production and the ground water interactions in selected areas of western Pennsylvania: Proceedings of the Second Symposium on Coal Mine Drainage Research (Monroeville, Pennsylvania) pp107-151.

Caruccio, Frank T. and G. Geidel 1986. An evaluation of mine waste - overburden analytical techniques: Proceedings of the 1986 National Symposium on Mining, Hydrology, Sedimentology, and Reclamation (Lexington, Kentucky, December 8-11, 1986). University of Kentucky.

Erickson, P. M. and R. S. Hedin. 1988. Evaluation of overburden analytical methods as means to predict post-mining coal mining drainage quality. p. 11-19. Bureau of Mines Information Circular 9183. Vol. I: Mine Water and Mine Waste. (Proc., Americ. Soc. For Surf. Mining and Reclamation, Pittsburgh, PA).

Finkleman, R. and D. Giffin, 1986. Hydrogen peroxide oxidation: an improved method for rapidly assessing acid-generating potential of sediments and sedimentary rocks. Reclamation and Revegetation Research, Vol. 5, pp521-534.

Hood, W. and O. Oerter, 1984. A leaching column method for predicting effluent quality from surface mines: Proceedings of the Symposium on Surface Mining Hydrology, Sedimentology, and Reclamation, pp271-277.

Perry, E. 1985. Overburden analysis: an evaluation of methods. pp369-375. In Proc., Symposium on Surface Mining, Hydrology, Sedimentology and Reclamation. Univ. of Kentucky.

Renton, J. J., T. E. Rymer, and A. H. Stiller. 1988. A laboratory procedure to evaluate the acid producing potential of coal associated rocks. Mining Science and Technology 7:227-235.

Sobek, A. A., W. A. Schuller, J. R. Freeman, and R. M. Smith, 1978. Field and laboratory methods applicable to overburdens and minesoils. EPA-600/2-78- 54, National Technical Information Service, Springfield, Virginia.

Sullivan, P. J. and A. A. Sobek. 1982. Laboratory weathering studies of coal refuse. Minerals and the Environment 4:9-16.

This page intentionally left blank.

# LONG-TERM LEACHING OF MINE SPOIL WITH SIMULATED PRECIPITATION[1]

by

Howard G. Halverson and Claude E. Gentry[2]

Abstract. Fresh mine spoil at a pH of 3.84 was collected and transferred to leachate columns in early August 1988. The spoil was leached weekly with simulated precipitation at pH values of 5.6, 5.0, 4.6, 4.2, and 3.8. Control samples of mine spoil were simultaneously leached with distilled water at a pH of 6.47. Leachate was collected and analyzed by standard methods for cations, anions, conductivity, and pH. The cations and anions of major interest were iron, aluminum, manganese, and sulfate. The leachate initially was pH 2.1, with a conductivity of about 10,000 micromho and had concentrations of Fe, Al, Mn, and $SO_4$ greater than 4,000, 300, 400, and 24,000 mg/L, respectively. Contaminant levels in the leachate dropped rapidly early in the leaching regime. After four weeks, pH increased slightly to 2.2, conductivity declined to about 7,000, and the ionic concentrations declined to averages of 1,300 for Fe, 170 for Al, 175 for Mn, and 16,000 for $SO_4$. In the following weeks, contamination in the leachate continued to decline but at a lesser rate. At 20 weeks, pH remained near the 2.0 to 2.2 level; conductivity continued to decline to 4,500; and the ionic concentrations in the leachate were Fe at 325, Al at 55, Mn at 35, and $SO_4$ at 3,800 mg/L. After 46 consecutive weeks of leaching, the concentrations in the leachate had declined to lower levels. The leachate pH held steady at a value of about 2.1 and conductivity declined to 3,500. Ionic concentrations were Fe at 70, Al at 20, Mn at 6, and $SO_4$ at 900 mg/L. The companion control samples leached with distilled water showed a similar declining pattern, but the absolute values of contamination were always lower than those derived with acidified leaching materials. Except for the control leach material at pH 6.4, the lower pH levels--5.6 and below--all seemed to leach contaminants from the spoil with equal efficiency.

Additional key words: pH, acid mine drainage

## Introduction

Precipitation percolating through mine spoil does dissolve and transport a variety of materials in the leachate water (Doepker 1988). The dissolved constituents may remain on-site and become part of the ion balance of soil water and eventually affect plant establishment and growth. The dissolved material also can have off-site effects, either through addition to groundwater or by addition to surface runoff from the area. When the percolate enters the surface water system, it may be called mine drainage, or if acidic, acid mine drainage.

Mine drainage is often severely contaminated, but the contaminant concentration can vary widely among different sources. Watzlaf (1988) worked with several mine waters with pH values as low as 2.0 and iron concentrations ranging from 7 to 7,000 mg/L. In general, contaminant concentration increased as pH delined. Mine drainage is often contaminated, but the water chemistry tends to be consistent with time (Halverson and Wade 1988).

In each of the studies of natural mine drainage or leaching, attention has centered on leachate chemistry rather than the properties of the water used to leach the spoil. This procedure assumes the acid and contaminant potential in the spoil is so great that precipitation chemistry would not be important. In this study, we used a leaching liquid with a chemistry similar to natural precipitation and a pH adjusted to the range normally expected in

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]Howard G. Halverson is Research Forester and Project Leader, USDA Forest Service, 1835 Big Hill Road, Berea, KY 40403 and Claude E. Gentry is Professor of Agronomy, Department of Agriculture, Berea College, Berea, KY 40404.

rainfall. The objective of the work was to determine if the chemistry of the leaching water, primarily pH, affected the leachate after the water percolated through a mine spoil.

## Materials and Methods

### Leaching Materials

The leaching solutions were based on distilled water to which reagent grade chemicals were added to approximate the composition of natural rainfall. The compounds used to approximate rainfall chemistry and the mass of each chemical used are given in Table 1. Table 2 lists the ion concentration of the leaching material rather than compound concentration. Solution pH was adjusted to treatment levels by controlling the mass of sulfuric and nitric acids used in the formulation.

### Spoil Materials

Fresh spoil material was collected from an active coal mine in south central Kentucky. The material was derived from shales located immediately above the coal. The spoil was acidic and contained significant amounts of pyritic sulfur (Table 3). The high values for potential acidity are similar to values derived for other Appalachian spoils (Ammons and Shelton 1988). The comparatively high values for organic carbon and sulfur likely indicate the presence of coal wastes in the spoil material. The acid base account suggests that leachate from the spoil should be quite acidic.

### Experimental Design

The spoil material was sieved (< 2 mm) to remove coarse fragments, and the spoil material was mixed well according to accepted procedures (Sobek et al. 1978). The mixed material was placed in 16 separate polyethylene leaching cylinders (10.4 cm diameter by 20.3 cm in length) and retained by an inert fiberglass mat at the bottom of each cylinder. Leaching columns were chosen above alternative methods because the technique has been shown to be the most representative of field conditions (Caruccio and Geidel 1986).

After the columns were filled, 15 cylinders were randomly assigned to five treatments in groups of three replications per treatment. The remaining cylinder was used to check the response of the spoil material to an initial leaching with unmodified distilled water at a pH of 6.4. The spoil in the treatment columns was wet with the same water to bring the material to a consistent starting moisture content.

### Treatments

The treatments consisted of leaching the material with artificial precipitation with pH adjusted to 5.6, 5.0, 4.6, 4.2, and 3.8. Each of the three replications was irrigated at weekly intervals with 2.54 cm of water and allowed to drain freely. Leachate water from each column was collected in a new polyethylene bottle each week, usually on Thursday, and analyzed the following day. Treatments began in August of 1988 and continued until mid-July 1989. A total of 50 weeks was included in this experiment.

The treatment columns were covered with a watchglass during the six-day period between irrigations to prevent excessive drying of the spoil. Although the columns were protected, the seal was not complete so many soil pores drained and a definite drying cycle was established and air was introduced into each column between leaching treatments. Approximately 50 ml of solution was required to restore the spoil to field capacity prior to leaching. Channel formation in the sample was prevented during percolation by a second fiberglass pad used to distribute the irrigation water over the spoil surface. Treatment columns were located in a climate controlled area to remove any effects of temperature changes.

### Chemical Analyses

The leachate samples were taken to the USDA Forest Service laboratory immediately after collection, and analyses were normally performed the next day. Each leachate sample was subjected to a thorough analysis using standard methods and appro-

Table 1. Chemical Composition of Rain Simulants (mg/L)

| Compound | Formula wt | Leaching Solution 5.6 | 5.0 | 4.6 | 4.2 | 3.8 |
|---|---|---|---|---|---|---|
| NaCl | 58.42 | 0.176 | | | | |
| $CaSO_4.2H_2O$ | 172.17 | 0.709 | | | | |
| $MgSO_4.7H_2O$ | 246.47 | 0.355 | | | | |
| $K_2SO_4$ | 174.27 | 0.069 | Same for all solutions | | | |
| $NaNO_3$ | 84.99 | 0.368 | | | | |
| $(NH_4)_2SO_4$ | 132.14 | 0.746 | | | | |
| $NH_4NO_3$ | 80.04 | 0.243 | | | | |
| $H_3PO_4$ (85.2%) | 97.975 | 0.0084 | | | | |
| $H_2SO_4$ (96.5%) | 98.07 | 0 | 0.383 | 0.987 | 2.451 | 5.948 |
| $HNO_3$ (70%) | 63.00 | 0 | 0.221 | 0.554 | 1.377 | 3.460 |

Table 2. Ion Concentrations for Precipitation Simulants (mg/L)

| pH | $H^+$ | $Ca^{++}$ | $Mg^{++}$ | $K^+$ | $Na^+$ | $NH_4^+$ | $NO_3^-$ | $SO_4^=$ | $Cl^-$ | $PO_4^{-3}$ |
|---|---|---|---|---|---|---|---|---|---|---|
| 5.6 | 0.0025 | 0.165 | 0.035 | 0.031 | 0.169 | 0.258 | 0.457 | 1.114 | 0.107 | 0.007 |
| 5.0 | 0.010 | | | | | | 0.610 | 1.476 | | |
| 4.6 | 0.025 | | Same for all solutions | | | | 0.839 | 2.015 | | |
| 4.2 | 0.063 | | | | | | 1.406 | 3.422 | | |
| 3.8 | 0.158 | | | | | | 2.841 | 6.737 | | |

Table 3. Properties of Spoil Material used in Leaching Study Based on Total Sulfur and Pyritic Sulfur

| Variable | Units | Value |
|---|---|---|
| | Values based on total sulfur | |
| pH, paste method | pH units | 3.84 |
| Neutralization Potential | $CaCO_3$ equivalent per 1000 parts material | 11.73 |
| Potential Acidity | $CaCO_3$ equivalent per 1000 parts material | 41.25 |
| Net Neutralization Potential | | -29.55 |
| Total Sulfur | Percent | 1.32 |
| Total Carbon | Percent | 11.34 |
| | Values based on pyritic sulfur | |
| Neutralization Potential | $CaCO_3$ equivalent per 1000 parts material | 11.73 |
| Potential Acidity | $CaCO_3$ equivalent per 1000 parts material | 26.88 |
| Net Neutralization Potential | | -15.15 |
| Sulfate | Percent | 0.09 |
| Pyritic Sulfur | Percent | 0.86 |
| Organic Sulfur | Percent | 0.37 |

priate reference standards. Details of the methods are presented in Table 4. Data analyses were captured in a data file and formatted for statistical analysis with a computer system dedicated to laboratory control.

Statistical Analyses

Data analyses were accomplished by utilizing available statistical subroutines (Wilkinson 1988). The primary method was an analysis of covariance. Covariance techniques were used to evaluate the effect of treatment pH while removing the effect of the sequential leachings. The effect of sequential leaching was isolated by using both week number and log week number as covariates. This technique identified both linear and curvilinear week effects. The model expressed each variable as a function of a constant, the treatment, log of the week number, the interaction, and error. Interactions, when significant, suggest that coefficients for slope due to treatment are different.

Factor analysis procedures were used to identify groups of contaminants that responded to the treatments in a similar fashion. Factor analysis

Table 4. Analyses Methods Used to Determine Contaminants in Mine Spoil Leachate

| Variable | Method Type | EPA Reference Number |
|---|---|---|
| Acidity | Titrimetric | 305 |
| Alkalinity | Titrimetric | 310.1 |
| Metals and related elements including:<br>B Si Zn<br>P Fe Cu<br>Mn Mg Na<br>Co Al Ni<br>Ca K Ti<br>Cr Pb | Argon Plasma Emission Spec. | AES0029 |
| pH | Potentiometric | 150.1 |
| Specific Conductance | Potentiometric | 120.1 |
| Sulfate | Turbidometric | 300 |
| Sediment, >0.45 micron | Gravimetric | 160.2 |
| Total Dissolved Solids | Computed | - |

does not identify the factors causing the response but can be used to suggest reasons for an observed result. Factor analysis is not based on a model.

## Results and Discussion

Early results from the leaching treatments illustrate some important effects of sequential irrigations of the columns. A single column was leached with fresh distilled water at pH 6.4. The water was not allowed to equilibrate with $CO_2$ in the air. The leachate chemistry with fresh distilled water was compared periodically with the percolate from leachings with water at the lower pH values, and a few variables of interest are presented in Table 5. These results do not indicate a statistically significant difference among treatments but do illustrate a pattern of high outputs early in the leaching cycle followed by declining values in succeeding weeks. Conversely, values for pH did not change with successive leachings and were not responsive to acidity of the irrigating solution.

The statistical analyses provided more detailed information on the effects of leaching. The results are summarized by main effects, time, treatment, and the interaction (Table 6). Time was transformed to log of treatment week to normalize the data. Treatment, the pH of the leaching solution, and time, log of the sequence member of a leaching, as well as the interaction, significantly affected the concentration of each variable in the leachate in almost all instances.

There were two important exceptions to the significant impacts of treatment and time on leachate chemistry. The pH of the leachate, although treatment showed significant effects, showed little variation during the sequences of leachings, possibly due to buffering. However, less than 5 percent of the variance in pH was explained, so treatment had little total impact. Silicon, a relatively resistant mineral, showed no response to treatment, but the analyses indicated a gradual dissolution over time. Two other elements, Na and Ti, showed treatment effects but were present only in low concentrations seldom exceeding 1 ppm in the leachate. The remaining constitutents in the leachate all showed significant effects with treatment, time, and interaction.

Factor analysis was used to group the contaminants in the leachate into categories of materials with a similar response. Factor analysis isolates groups with a similar response but does not identify the factor causing the response. Factor analysis indicated that Mn, Al, Mg, Fe, Zn, Ca, Si, and $SO_4$ formed a group that seemed to be correlated with the number of leachings. The computed variables acidity, conductivity, and total dissolved solids (TDS) were also in this group. The response curve had a reverse "J" shape when plotted against leaching number. The computed variables, as expected, were in the group with the highest concentrations in the leachate. The second group included Pb, P, Cu, B, Co, Ni, and leachate pH. The second grouping seemed to be sensitive to the pH of the leaching solution implying acid-induced dissolution as the response curve was more closely related to treatment. The remaining variables, including pH, Na, Cr, K, and Ti all acted independently of the first two groups and each other. The concentration of Na, Cr, and Ti were too low and consistent to show a definite response to the treatment. The remaining variable, K, may have been released from between layers in the clay lattice structure during alternate wetting and drying cycles and actually increased with time. The leachate pH showed no response to treatment or number of leachings.

Table 5. Chemistry of Minespoil Leachate From Fresh Distilled Water at pH 6.47 Compared to the Mean Chemistry of Percolated from Water at pH Values of 5.6 and Below

| Week | Treatment pH | Leachate | | | | | |
|---|---|---|---|---|---|---|---|
| | | Conductivity micromhos | pH | Fe | Al | Mn | $SO_4$ |
| | | | | mg/L | | | |
| 1 | 6.5 | 11,100 | 2.2 | 4,999 | 370 | 378 | 30,700 |
| | < 5.6 | 10,706 | 2.1 | 4,027 | 321 | 353 | 24,880 |
| 2 | 6.5 | 6,400 | 2.2 | 1,255 | 205 | 205 | 11,600 |
| | < 5.6 | 7,006 | 2.2 | 1,354 | 172 | 177 | 10,750 |
| 20 | 6.5 | 4,100 | 2.1 | 178 | 47 | 28 | 3,150 |
| | < 5.6 | 4,733 | 2.1 | 328 | 54 | 37 | 3,841 |
| 46 | 6.5 | 2,946 | 2.2 | 50 | 18 | 3 | 572 |
| | < 5.6 | 3,176 | 2.1 | 70 | 22 | 6 | 881 |

Table 6. Significance of Treatment and Number of Consecutive Leachings on Leachate Chemistry of Mine Spoil

| Variable | Treatment (T) | Week (W) | Logweek (LW) | TxW | TxLW | $R^2$ |
|---|---|---|---|---|---|---|
| Conductivity | ***[1/] | *** | *** | ** | ns | .933 |
| pH | ns | ns | ns | ns | ns | .021 |
| $SO_4$ | *** | *** | *** | *** | *** | .942 |
| Acid | *** | *** | *** | *** | *** | .942 |
| B | *** | *** | *** | ns | ns | .873 |
| Si | ns | *** | *** | ns | ns | .655 |
| Zn | *** | *** | *** | *** | *** | .926 |
| P | *** | *** | *** | *** | *** | .910 |
| Fe | *** | *** | *** | *** | *** | .915 |
| Cu | *** | *** | *** | *** | *** | .926 |
| Mn | *** | *** | *** | ns | *** | .976 |
| Mg | *** | *** | *** | *** | *** | .944 |
| Na | ns | *** | *** | ** | ns | .249 |
| Co | *** | *** | *** | ** | *** | .933 |
| Al | *** | *** | *** | ns | *** | .948 |
| Ni | *** | *** | *** | ** | *** | .937 |
| Ca | *** | *** | *** | ns | ** | .906 |
| K | *** | *** | *** | ** | *** | .081 |
| Ti | *** | *** | *** | ns | ns | .176 |
| Cr | *** | *** | *** | ** | *** | .900 |
| Pb | *** | *** | *** | ns | *** | .856 |
| TDS | *** | *** | *** | *** | *** | .949 |

[1/] ***-Significant at .001 level, **-Significant at .01, ns-Nonsignificant.

The analyses indicate that precipitation chemistry and the number of leaching events are important factors in determining spoil leachate chemistry. However, the concentration of materials in leachate declines rapidly during the first few leachings of fresh spoil indicating some resistance to weathering. The leachate chemistry tended to stabilize after about 20 leaching events and continued at the same levels for the next 30 weekly leachings. Leaching materials with pH values 5.6 and below apparently leached contaminants with equal efficiency.

## Literature Cited

Ammons, J. T. and P. A. Shelton. 1988. A comparison of results from acid-base accounting versus potential acidity measured by the peroxide oxidation of weathered and unweathered soil containing pyrite. p. 206-209. In Mine Drainage and Surface Mine Reclamation, Vol. 1: Mine Water and Mine Waste. (Pittsburgh, Pa., April 19-21, 1988). Bureau of Mines Information Circular IC 9183.

Caruccio, Frank T. and Gwendelyn Geidel. 1986. An evaluation of mine waste overburden analytical techniques. p. 147-153. In 1986 National Symposium on Mining, Hydrology, Sedimentology, and Reclamation. (Lexington, Ky., December 8-11, 1986). University of Kentucky.

Doepker, Richard D. 1988. The interrelation of factors influencing the dissolution of metals in columns of mine tailings. p. 210-219. In Mine Drainage and Surface Mine Reclamation, Vol. 1: Mine Water and Mine Waste. (Pittsburgh, Pa., April 19-21, 1988). Bureau of Mines Information Circular IC 9183.

Halverson, Howard G. and Gary L. Wade. 1988. Chemical variation in acid mine drainage in southern Kentucky. p. 95-104. In 1988 Symposium on Mining, Hydrology, Sedimentology, and Reclamation. (Reno, Nev., December 5-9, 1988). University of Kentucky.

Sobek, A. A., W. A. Schuller, J. R. Freeman, and R. M. Smith. 1978. Field and laboratory methods applicable to overburden and minesoils. EPA-600/2-78-OS4. Cincinnati, Oh. pp. 204.

Watzlaf, George R. 1988. Chemical stability of manganese and other metals in acid mine drainage sludge. p. 83-90. In Mine Drainage and Surface Mine Reclamation, Vol. 1: Mine Water and Mine Waste. (Pittsburgh, Pa., April 19-21, 1988). Bureau of Mines Information Circular IC 9183.

Wilkinson, L. 1988. SYSTAT: The system for statistics. Systat, Inc., Evanston, Ill.

# SUCCESSFUL RECLAMATION USING CONTROLLED RELEASE BACTERICIDES: TWO CASE STUDIES[1]

by

Andrew A. Sobek, Donald A. Benedetti, and Vijay Rastogi[2]

Abstract. Controlled release bactericides affect reclamation success and provide assurance against post-reclamation water quality problems. They inhibit Thiobacillus ferrooxidans and aid in the establishment of beneficial heterotrophic bacteria necessary to support revegetation of the site. These conditions persist after the bactericide is depleted from the controlled release systems. Two coal refuse disposal areas, one in Ohio and one in West Virginia, were reclaimed using two different generations of controlled release bactericides. Case Study #1, located in Ohio, was reclaimed using first generation products with a release life of two years. Yet, six years after application, the treated area continues to have a dense vegetative cover while the untreated control area has only sparse vegetation. Water quality data from the treated area continue to show a significant improvement versus that from the control area. Case Study #2, located in West Virginia, was reclaimed using third generation products with a controlled release life in excess of seven years. In its third year, the vegetation is lush and healthy except for the control area where vegetation is becoming sparse due to acid toxicity. The water quality data from the treated area corroborates these improvements and justifies the use of bactericides in reclamation.

Additional Key Words: bactericide, reclamation, water quality, Thiobacillus ferrooxidans.

## Introduction

Pyritic materials associated with coal refuse oxidize to form mineral acids when exposed to the atmosphere. This oxidation process is catalyzed by the bacteria Thiobacillus ferrooxidans (Beck and Brown 1968; Baker 1975) causing acid generation to overwhelm inherent neutralizers in the reject materials. Kleinmann and others (1981) proposed a pH dependent three-stage mechanism of acid generation in pyritic materials. The bacteria, T. ferrooxidans, play a very important role in this process, making bacteria control one method of breaking the acid-generation cycle.

Inhibiting or destroying thiobacilli can significantly slow the rate of acid production. Anionic surfactants, organic acids and food preservatives (Onysko et al. 1984) act as bactericides and kill these bacteria; however; bactericides degrade over time and are lost because of leaching and runoff. To overcome the inherent short duration effectiveness of spray applications, controlled release systems to provide the bactericide slowly over a long time period were developed (Sobek et al. 1985).

Control of acid generation for prolonged periods greatly enhances reclamation efforts and can reduce reclamation costs by reducing the amount of topsoil needed to establish vegetation. Three natural processes resulting from strong vegetative cover for three years or more can break the acid production cycle. These processes are: (1) a healthy root system that competes for both oxygen and moisture with acid-producing bacteria; (2) populations of beneficial heterotrophic soil bacteria and fungi that are reestablished, resulting in the formation of organic acids that are inhibitory to T. ferrooxidans (Tuttle et al. 1977); (3) the action of plant root respiration and heterotrophic bacteria activity increase $CO_2$ levels in the spoil, resulting in an unfavorable microenvironment for growth of T. ferrooxidans.

The objectives of the two case studies are: (1) to demonstrate that bactericides in a combination of liquid spray and controlled release pellets will substantially reduce acid

---

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]Andrew A. Sobek is Technical Manager, Donald A. Benedetti is Environmental Engineer, and Vijay Rastogi is Business Manager of ProMac Systems, The BFGoodrich Company, Akron, OH 44313.

formation; (2) to demonstrate that after controlled release systems are exhausted, the inhibition of acid production continues for prolonged time periods; (3) to demonstrate the feasibility of incorporating these materials into normal reclamation practices.

Case Study #1: Route 43, East Springfield, Ohio

The paste pH of the coal refuse was less than 3.0 across this site while 10:1::water:refuse extracts had pH values ranging from 2.5 to 3.5. A composite sample of refuse was analyzed and found to be 50.3% total carbon, 1.32% total sulfur, 0.11% sulfate sulfur, 0.43% pyritic sulfur, and 0.78% organic sulfur. No neutralization potential measurement was made.

Site Treatment

No amendments were added to the regraded refuse pile before ProMac Systems was applied. The 0.9 hectare at the southern end of the site was left as a control plot while the 1.0 hectare adjacent to the northwest side of the control was treated with ProMac Systems. ProMac Systems used for this site had the following formula: (1) ProMac 500SN - 386 kg, (2) ProMac 500PBL - 408 kg, and (3) ProMac 600PBL - 417 kg. This site received 465 ppm of bactericide in controlled release form.

The site, both control and treated areas, was covered with 15.2 to 20.3 cm of topsoil. The topsoil was fertilized with 336 kg/ha of a 16-16-16 fertilizer and 6.9 m.t./ha of lime. The area was seeded with 61.6 kg/ha of seed mixture followed by an application of 4.5 m.t./ha of hay mulch.

Site Monitoring

Methods. Eight soil moisture samplers (pressure-vacuum lysimeters) were installed at different locations on the site. It was expected that the effects of the ProMac Systems application would be noticeable first in the vadose zone, and then in the drains or the diversion ditch. A rain gauge and an evaporation pan were also installed on the site.

Monitoring included periodic site inspections to ascertain surface vegetation cover and acid "burnout." The site was photographed periodically and sampled for bacterial activity during the past five years.

Soil samples were taken from both treated and untreated areas of the site with a soil corer. Each soil core had all the topsoil layer and part of the refuse material. The areas of the pile, treated and untreated, were sampled according to a grid pattern. Each area was divided into four blocks, avoiding the border zone between treated and untreated. Peripheral areas were also avoided. Five samples were taken from each block, totalling 20 samples per area. Each block was combined to yield a total of four composite samples per area. This reduced the variability between samples that originated from the same area, reduced the number of dilutions needed for testing and enhanced the statistical significance of the test.

Two separate experiments were conducted to measure the most probable numbers (MPN) of heterotrophs and Thiobacillus ferrooxidans. The first was the growth of heterotrophic microorganisms in cover soil of the treated section versus cover soil from the untreated section using a semi-rich medium (TSB). This comparison was conducted for composite samples of the cover soil. The second experiment was the growth of T. ferrooxidans in treated versus untreated sections using a specific medium (9K). This comparison was conducted for composite samples of the refuse layer. Each composite sample consisted of five individual cores. This study was conducted once each year for five years starting in June 1985.

Microbiology. The thickness of the cover soil in the treated and untreated areas of the refuse pile was the same statistically; however, the actual thickness of cover soil was greater in the untreated area (20.3 cm average) than in the treated area (17.3 cm average). The presence and amounts of heterotrophic microorganisms determines the status of the vegetation in a given area. A strong and varied heterotrophic population supports a good vegetative cover. These microorganisms proliferate in areas that contain enough preformed organic matter to sustain their growth. The organic matter in the refuse pile is mostly coal. Coal, especially when present in large particles, is not a carbon source readily utilized by most heterotrophic organisms. Thus, heterotrophs were enumerated only in the cover soil and not in the refuse. Enumeration was done using a general heterotrophic medium amended with sterile soil extract to supplement unknown required growth factors.

The distribution of the heterotrophic organisms in the study site is shown in Figure 1. The untreated area displayed a suppressed number of heterotrophs when compared with treated area. The difference between treated and untreated areas is close to two orders of magnitude except for the drought year of 1988. A very good and healthy heterotrophic microbial population is shown in the treated area except for 1988, when the untreated area was almost equal to the treated.

Thiobacilli are a special group of bacteria that live in a low pH environment, deriving their energy from oxidation of pyritic material and their carbon source from carbon dioxide. In an environment rich in organic matter they are surpassed in growth by heterotrophs. Thus, thiobacilli counts were performed only for the zone starting from the base of the cover soil extending into the upper part of the refuse itself.

The refuse (Figure 2) shows significant differences in numbers of thiobacilli in the treated area compared with the untreated area except for 1986 and during the drought year of 1988. The number of T. ferrooxidans were found to be roughly the same throughout the 12.7 cm thickness of refuse tested in the treated area before the data were composited. This same trend was found in the control area.

Normally, it is expected that as one goes lower in the soil system, lack of oxygen and

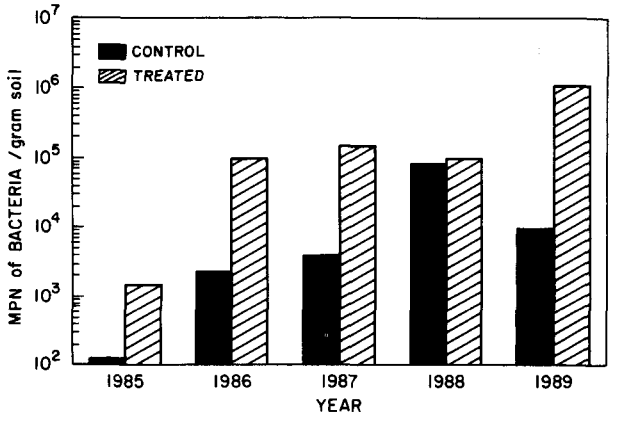



Figure 1. Heterotrophs in the treated and untreated areas at the R. 43 Site

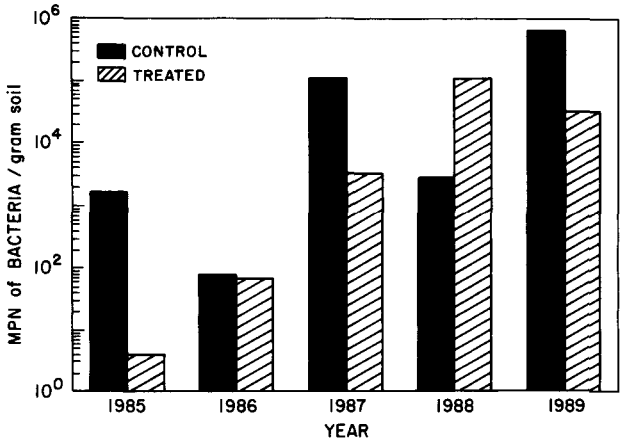



Figure 2. T. ferrooxidans in the treated and untreated areas at the Rt. 43 site

limiting amounts of nutrients will inhibit or reduce growth. These data from the control area for 1985 and 1986 may indicate a trend of downward movement of thiobacilli to lower depths of the pile. However, the data from 1987 and 1989 would tend to refute that supposition. The data for 1986 may have resulted from the microenvironmental conditions at the time of sampling. This hypothesis will have to be verified in future studies.

It has been found that specific classes of microorganisms are better presented as a ratio of the total population (Horowitz and Atlas 1976; Walker and Colwell 1976). Specific classes of bacteria will increase in number as a result of a total increase in the bacterial population. Thus, if the total number of bacteria increases in a specific environment, it is expected that a general increase in every specific group of bacteria will be seen. This situation is well presented in this study. In spite of the fact that higher numbers of thiobacilli were found in

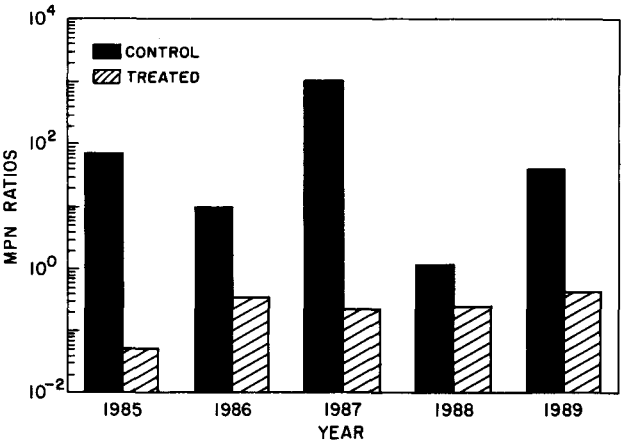



Figure 3. T. ferrooxidans-heterotrophs ratio at the Rt. 43 site

1988 in the treated than in the control areas, the ratio of thiobacilli to heterotrophs was higher in the control area (Figure 3).

Results of the last five years (Figure 1) show an increase in the number of heterotrophs each year, except for the drought year of 1988, in the treated area. However, in the control area, heterotrophs had a large increase in 1988, but decreased in 1989. The number of thiobacilli in the treated area (Figure 2) increased by more than ten-fold each year through the drought year of 1988, but decreased in 1989. In the control area, the thiobacilli decreased by more than ten-fold from 1985 to 1986, then increased by a factor of 1000 from 1986 to 1987, followed by another ten-fold decrease in 1988 and a 100-fold increase from 1988 to 1989. The increases in thiobacilli in the control area are much higher and exhibit an up-and-down effect, while the increases in the treated areas have been consistent until 1988 and 1989.

The ratios of thiobacilli to heterotrophs for a five-year period (Figure 3) make the differences between the treated and control areas very clear. The data indicate a fairly stable and low ratio of thiobacilli to heterotrophs in the treated area. In the control area, the ratios are much higher and follow the same trend exhibited by the thiobacilli populations over the past five years (Figure 2).

Water Quality. Background water samples were taken at points in the ditch that collected drainage and flowed from south to north along the western boundary of the site. The data (Table 1) indicate that a near-neutral to slightly alkaline water enters the site at 0.6 L/s and flows past the gob pile. By the time the water reaches the sampling point on the western edge of the control plot, the flow has slowed and has become very acidic.

Water samples were to be collected throughout the project on a monthly basis; however, the previous water flow pattern was not reestablished after the site was regraded and recontoured. The

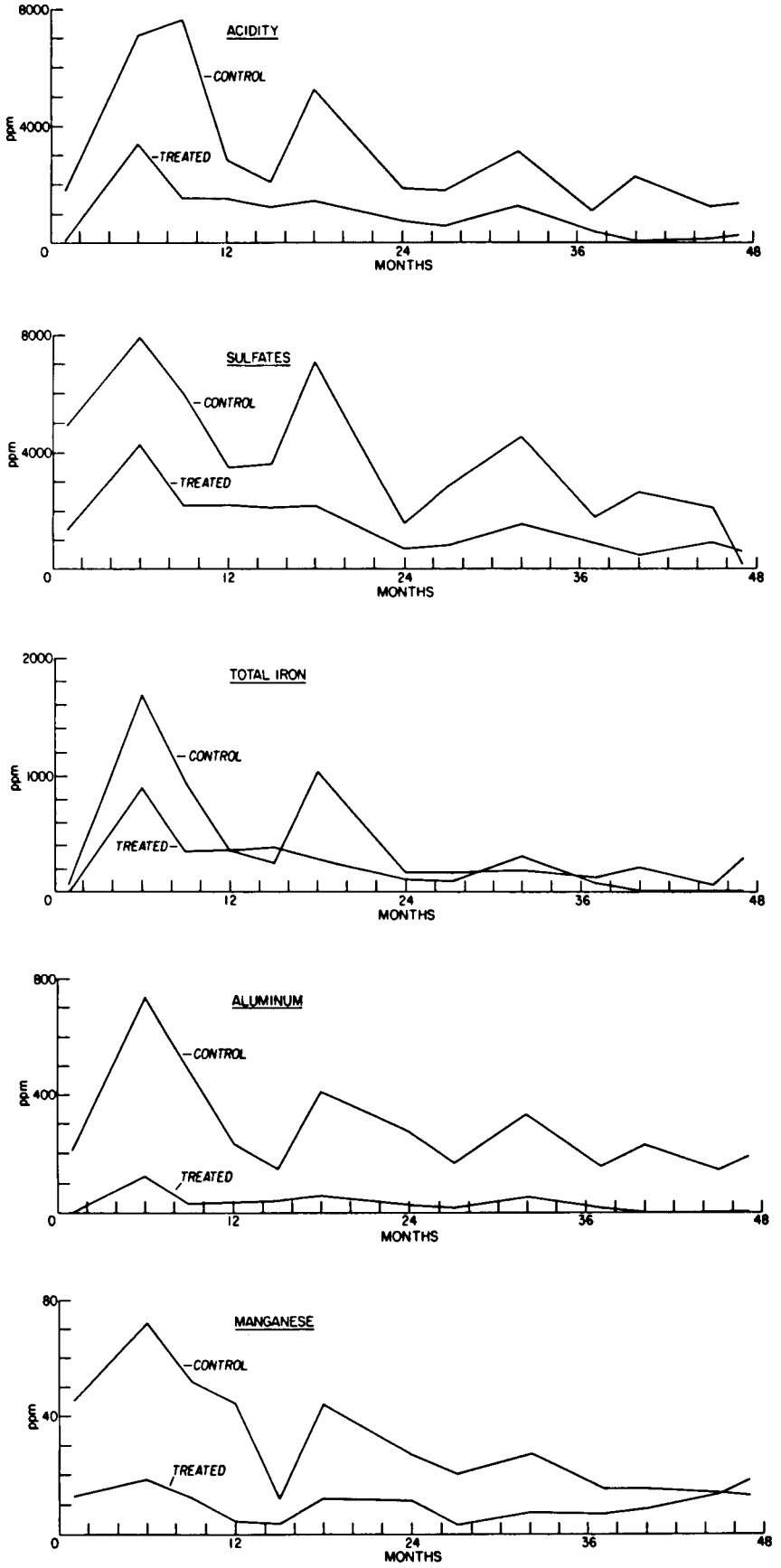



Figure 4. Refuse water quality data from the Rt. 43 site

Table 1. Background Water Quality Data

| Parameter | Sample Locations: Exit | | Treated Plot | | Control Plot | | Entrance | |
|---|---|---|---|---|---|---|---|---|
| | * | ** | * | ** | * | ** | * | ** |
| pH (field) | 2.7 | - | 2.7 | - | 2.4 | - | 6.4 | - |
| pH (lab) | 2.6 | 2.7 | 2.6 | 2.7 | 2.4 | 2.4 | 7.0 | 7.3 |
| Acidity (mg/L) | 1034 | 716 | 1503 | 1104 | 1391 | 941 | - | - |
| Total Iron (mg/L) | - | 72 | - | 60 | - | 96 | - | 0 |
| Temperature ($^{o}$C) | 15 | 22 | 15 | 22 | 13.5 | 22 | 11.5 | 22 |
| Conductivity (umhos/cm) | 2650 | 2150 | 3750 | 2950 | 3500 | 2850 | 650 | 700 |
| Flow (L/s)*** | 3.1 | - | 0.06 | - | 0.06 | - | 0.63 | - |

* Water samples taken on 4/1/83
** Water samples taken on 5/18/83
*** Flow was estimated on 4/1/83 only
(-) Indicates no data

only consistent water flow on-site was at the exit from the site. This sample location is northwest of the treated area and is indicative of the untouched portion of the site that was not included in the study.

In June 1984, a soil moisture sampling system (pressure-vacuum lysimeters) was installed for monitoring the water quality of the vadose zone nearest the plant root system, and as a "backup" system for the water quality monitoring of runoff and seepage water collected by the drainage ditch. The installation and sampling procedures of Parizek and Lane (1970) were followed.

The monthly data for acidity, sulfates, aluminum, manganese and total iron are found in Figure 4. Acidity, sulfates, aluminum and manganese all exhibit a downward trend in both the treated area and the control area; however, the treated area is much lower in all parameters until month 47. Sulfates in the control area decreased below the sulfate concentration in the treated area and manganese in the treated area increased to a higher level than in the control area at month 47. Total iron concentration in the control area is erratic and dips below the total iron concentration in the treated area in months 12, 15, and 32. However, total iron in the control area always increases sharply after reaching these lows.

The data summarized in Table 2 are the percent reduction in each parameter when the average of all water quality measurements from samples taken out of the four lysimeters in the treated portion of the site are compared with the average of all water quality measurements from samples taken out of the four lysimeters in the non-ProMac treated portion of the site. The data in Table 2 indicate the ProMac treatment has been very successful in reducing all parameters, except for manganese in 1987 and 1988. In the second year, except for total iron, the reductions are not as large as in year one.

Table 2. Reduction in Specific Parameters Comparing Treated to Untreated Areas at Route 43

| Parameter | Year: 1985 | 1986 | 1987 | 1988 |
|---|---|---|---|---|
| Acidity | 72% | 58% | 97.2% | 89.0% |
| Specific Conductivity | 35% | 32% | 67.2% | 86.0% |
| Total Iron | 52% | 73% | 92.1% | 99.6% |
| Manganese | 94% | 91% | 44.0% | 15.6% |
| Aluminum | 93% | 89% | 99.5% | 98.9% |
| Sulfates | 68% | 58% | 81.4% | 57.0% |

Total iron shows a 21% increase in reduction in 1986 and a 40.1% increase in reduction in 1987 when compared to 1985.

Vegetative Ground Cover Evaluation. Biomass production was measured by running transect lines across the plots. Four quadrats (0.37 $m^2$ per quadrat) were clipped at ground level where the transect lines intersected. Total biomass was measured (dry weight basis) by drying all clippings at 105 degrees Celsius in a forced-air oven until a constant weight was reached. The treated area had a total biomass of 2,915 kg/ha in 1989 while the control area only had a total biomass of 315 kg/ha.

## Case Study #2: Dawmont Refuse Area, Dawmont, West Virginia

The Dawmont reclamation project, an abandoned coal refuse disposal area covering approximately 14.2 hectares, was polluting offsite areas with acid mine drainage (AMD) at a flow rate of 0.6 to 1.0 L/s.

Site Treatment

The regraded refuse received a blanket of lime applied uniformly at a rate of 22.4 m.t./ha before the following ProMac treatment was applied: (1) ProMac 2000SB - 3,175 kg, (2) ProMac 2000PN - 2,381 kg, (3) ProMac 2000PB - 3,969 kg, and (4) ProMac 2000PY - 3,969 kg. The site received 201 ppm of bactericide in controlled release form.

After the lime and ProMac products were applied, the entire site was covered with 30.5 cm of topsoiling materials. Fertilizer was applied to the topsoil at a rate of 1,121 kg/ha of a 10-20-20 commercial fertilizer plus 56 kg/ha of urea (46-0-0). Before the area was seeded and mulched with 4.5 m.t./ha of straw, 13.5 m.t./ha of lime was disked into the cover material.

Site Monitoring

Twelve pressure-vacuum lysimeters were installed at different locations on the site to measure water quality near the plant root zone. Six samplers were put in the 0.4 hectare control area and six samplers were put in the treated area. Vandals destroyed six lysimeters, causing a small gap in the water quality database.

Cover soil depth is measured once a year when the microbiological studies are done. Although it varies across the site, there is very little difference between the amount of cover on the control when compared to the treated portion of the site. The control area has an average of 24.4 cm of cover material compared to 23.6 cm of cover material in the treated area.

Microbiology. The microbiological studies conducted at this site were the same as the studies conducted at the Route 43 site in Ohio. When the treated and untreated areas were compared, the heterotrophic bacterial counts were significantly greater for the treated area (Figure 5). There was a decrease in heterotrophic counts each year in the untreated area while the heterotrophic counts increased in the treated area in 1988 and sharply decreased in 1989. Next year's data will be needed in order to determine if this is a trend or just a result of climatic conditions in 1988 and 1989.

The data illustrated in Figure 6 showed a trend to have fewer thiobacilli in the treated area than in the untreated area. However, there was a sharp increase in the thiobacilli in the treated area for 1988. It was expected that lower numbers of thiobacilli would be observed as in 1987 and 1989. Possibly the effect of ProMac products on the site was altered by the overall effect of the hot, dry summer of 1988. It seems that thiobacilli flourish better than heterotrophs at lower soil moisture contents or else there was not enough soil moisture to continually activate the control release mechanisms of ProMac pellets.

The ratio of thiobacilli to heterotrophs at the interface zone serves as a strong indicator for the overall microbial picture of the site. It balances situations such as high or low carbon

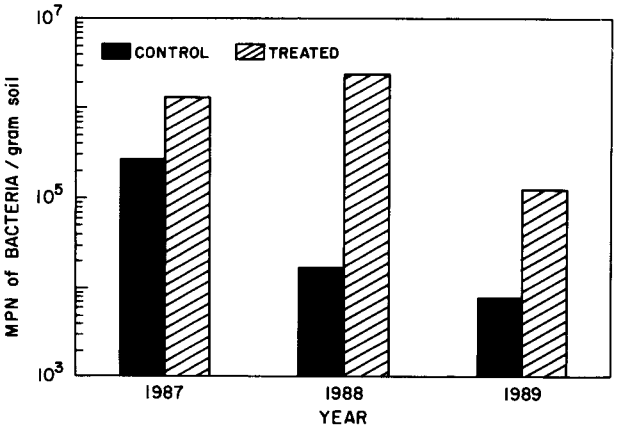



Figure 5. Heterotrophs in the treated and untreated areas at the Dawmont site

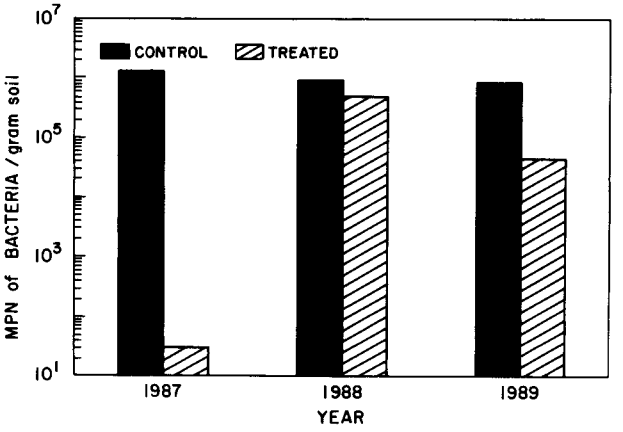



Figure 6. T. ferrooxidans in the treated and untreated areas at the Dawmont site

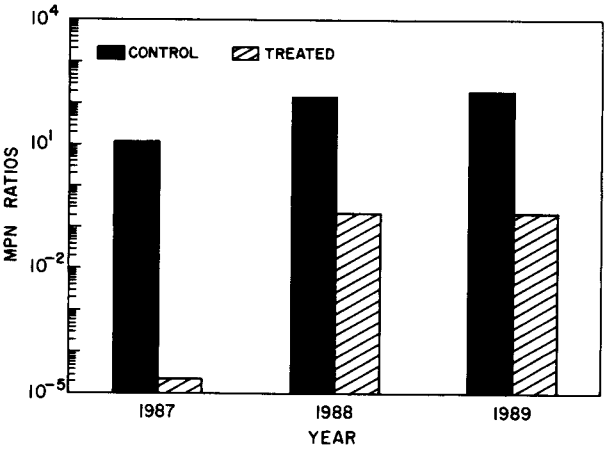



Figure 7. T. ferrooxidans-heterotroph ratio at the Dawmont site

availability or many other non-specific environmental factors that may affect the overall numbers. In most cases, it can be expected that the ratios of a specific bacterial group to the overall population will stay roughly constant.

The ratios of thiobacilli to heterotrophs (Figure 7) in the control area increased 10-fold from 1987 to 1988, and very slightly in 1989. In the treated area, the ratios were very low in 1987, increased 10,000-fold in 1988 and remained the same in 1989. A much lower ratio was expected to be obtained for the treated area in these years. Climatic conditions could be responsible for these unexpected results.

Water Quality. Background water quality data (Table 3) were obtained from a sample of standing water at the base of the unreclaimed refuse site (DM-A) and from a seep that flows from the toe of the regraded refuse area (DM-B). Sample DM-A is indicative of the surface runoff water quality while DM-B is most representative of water infiltrating the refuse, picking up a load of acid salts, and polluting offsite land and water resources. While both water samples are very acidic and contain large amounts of pollutants, sample DM-B would be the most costly to neutralize.

In October 1987, a soil moisture sampling system (pressure-vacuum lysimeters) was installed to monitor water quality in the deep root zone area. The monthly data for acidity, sulfates, aluminum, manganese and total iron are graphically displayed in Figure 8. The trend for all parameters except manganese is that the control area contains higher concentrations than the treated area. Manganese follows the same trend except for month 11 in 1988; the treated area contained more soluble manganese than the control. The drought during the summer months may be responsible for altering this trend.

When viewed from a different perspective (Table 4), the data show a significant reduction in measured parameters. In all parameters, significant reductions are realized when the treated area is compared to the control area.

Vegetative Ground Cover Evaluation. Background data from the refuse (Table 5) indicate why the site was barren. All refuse samples were composites of several smaller samples. Sample DM-1 was from the huge gob pile before it was regraded. Sample DM-2 was taken from the regraded area on the northern part of the site farthest from the railroad tracks while sample DM-3 was taken from the large regraded equipment storage area near the railroad tracks.

The acid-base account data (Table 5) showed that acid generation had occurred as evidenced by

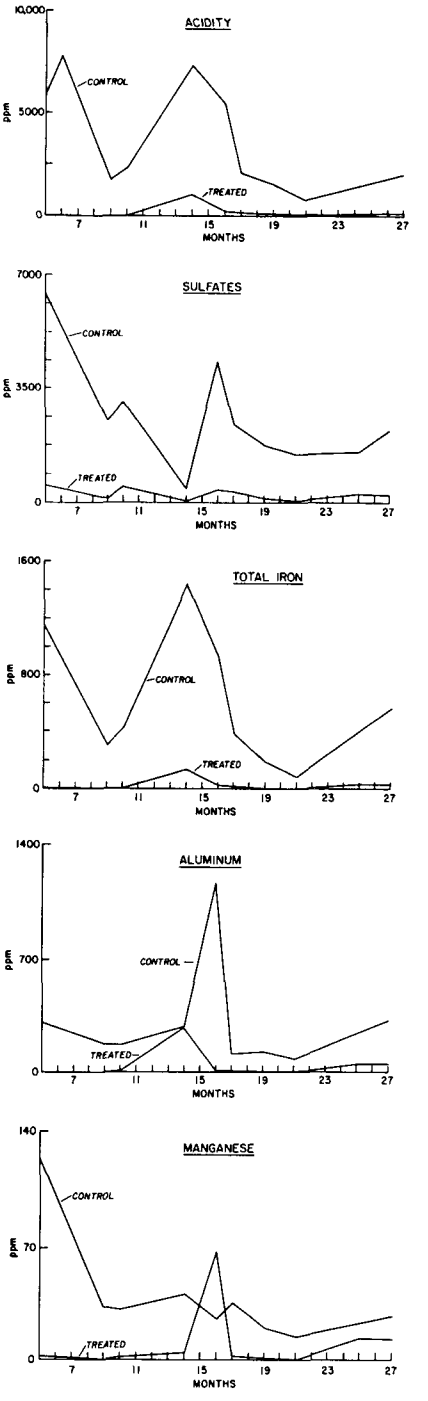



Figure 8. Refuse water quality data from the Dawmont site

Table 3. Background Water Quality Data from Dawmont Site

| Sample ID | pH | Conductivity (umhos/cm) | Acidity (mg/L) | SO4 (mg/L) | Fe (mg/L) | Mn (mg/L) | Al (mg/L) |
|---|---|---|---|---|---|---|---|
| DM-A | 2.5 | 4650 | 2876 | 2438 | 161 | 8.5 | 418 |
| DM-B | 2.1 | 18500 | 20610 | 3125 | 3600 | 290.0 | 1303 |

Table 4. Reduction in Specific Parameters at the Dawmont Reclamation Site

| Parameter | Year 1988 | Year 1989 | Overall |
|---|---|---|---|
| Acidity | 95.8% | 95.7% | 95.8% |
| Specific Conductivity | 89.9% | 86.5% | 91.2% |
| Sulfates | 90.5% | 90.1% | 90.8% |
| Total Iron | 96.1% | 93.8% | 97.3% |
| Manganese | 74.8% | 61.1% | 85.6% |
| Aluminum | 85.4% | 82.2% | 88.5% |

pH values ranging from a low of 2.0 to a high of 3.9. There were no neutralizers present in the refuse to counteract the effects of the acid being generated. The neutralization potential values ranged from a high of +1.79 m.t. $CaCO_3$ equivalent/1000 m.t. of refuse to a low of -10.37 m.t. $CaCO_3$ equivalent/1000 m.t. of refuse. The area (sample DM-3) on the western part of the site seemed to be the least toxic; however, the chemistry of this refuse would kill most vegetation.

The refuse was weathered as indicated by the amount of sulfate sulfur in samples DM-1 and DM-2 (Table 5). Approximately 50% of the total sulfur in these samples was in the sulfate form. Only sample DM-2 had more than 50% of its total sulfur as pyritic sulfur; however, this area was a regraded refuse pile. Moving the refuse around during regrading exposed slightly weathered material that was high in pyritic sulfur. Regardless of the state of weathering, more than enough pyritic sulfur was left to cause acid generation to continue for a long time. Also, there was a large store of acid salts that could be solubilized and leached from the site in the future.

Biomass production was measured in 1989 in the same manner as it was measured at the Route 43 site. Total biomass was measured on a dry weight basis and after a constant weight was reached, the treated area had 1,604 kg/ha while the control area had only 1,033 kg/ha. Considering the materials that make up the deep root zone, one might expect next year's data to show a greater difference between the two areas.

## Conclusions

The following conclusions can be drawn from these case studies:

1. Bactericides inhibit the proliferation of T. ferrooxidans.
2. A system that combines a liquid spray with controlled release pellets substantially reduces acid formation, allows heterotrophic microorganisms to proliferate and promotes good vegetative growth.
3. The bactericide has residual effect on acid generation long after the controlled release pellets have been exhausted. Generation one pellets used at the Route 43 site were made of natural rubber and had decomposed by the end of the second year of the study. No pellets were found and no trace of the bactericide was found in the surface and refuse water, but effective control of acid generation continues into the sixth year.
4. An integrated system of bactericides can be used as a standard reclamation practice without adding problems to the operation.
5. Without liming the refuse and by adding only minimal cover material, an integrated system of bactericides can help establish and maintain a healthy growth of vegetation.
6. The areas treated with ProMac bactericides show a better water quality in the vadose zone than the control area, providing a non-toxic source of moisture for plant growth.

## Literature Cited


Baker, M. 1975. Inactive and abandoned underground mines - water pollution prevention and control. U.S. Environmental Protection Agency. EPA 440/9-75-007. Washington DC.

Beck, J.V. and D.C. Brown. 1968. Direct sulfide oxidation in the solubilization of sulfide ores by Thiobacillus ferrooxidans. Journal of Bacteriology 96:1433-1434.

Horowitz, A. and R. M. Atlas. 1976. Oil biodegradation in Arctic coastal waters. p. 357-365. In Science in Alaska: Vol. 2. Proceedings of the 27th Alaska Science Conference. (Fairbanks, AK).


Table 5. Sulfur Forms and Acid-Base Account of Samples Taken at Dawmont Refuse Pile

| Sample Number | Paste pH | Total % S | Sulfate % S | Organic % S | Pyritic % S | Calcium Carbonate Equivalent m.t./1000 m.t. of Material: Acid Potential from Pyritic % S | Neutralization Potential | Deficiency(-) or Excess(+) Neutralizers |
|---|---|---|---|---|---|---|---|---|
| DM-1 | 2.5 | 1.79 | 0.882 | 0.685 | 0.223 | 6.97 | - 4.15 | - 11.12 |
| DM-2 | 2.0 | 18.83 | 2.200 | 1.030 | 15.600 | 487.50 | -10.37 | -497.87 |
| DM-3 | 3.9 | 3.90 | 2.160 | 1.010 | 0.730 | 22.81 | 1.79 | - 21.02 |

Kleinmann, R.L.P., D.A. Crerar and R.R. Pacelli. 1981. Biogeochemistry of acid mine drainage and a method to control acid formation. Mining Engineering p. 300-304.

Onysko, S.J., R.L.P. Kleinmann and P.M. Erickson. 1984. Ferrous iron oxidation by Thiobacillus ferrooxidans: inhibition with benzoic acid, sorbic acid and sodium lauryl sulfate. Applied and Environmental Microbiology 28(1).

Sobek, A.A., M.A. Shellhorn and V. Rastogi. 1985. Use of controlled release bactericides for reclamation and abatement of acid mine drainage. International Mine Water Congress. (Granada, Spain. September 17-21, 1985).

Tuttle, J.H., P.R. Dugan and W.A. Apel. 1977. Leakage of cellular material from Thiobacillus ferrooxidans in the presence of organic acids. Applied and Environmental Microbiology 33:459-469.

Walker, J.D. and R.R. Colwell. 1976. Enumeration of petroleum degrading microorganisms. Applied and Environmental Microbiology 31:198-507.

This page intentionally left blank.

# The Use of Phosphate Refuse as a Potential AMD Ameliorant

Walter Hart, Alfred Stiller, Thomas Rymer, John Renton[2]

## Abstract

The use of a variety of apatite rock called Code 31 has been evaluated in the laboratory as being a very effective acid mine drainage (AMD) ameliorant. Presently, field studies are being conducted to determine the economic feasibility of using this material to inhibit acid production from 100,000 ton refuse test piles. One drawback to the use of Code 31 is its associated cost and overall expense. A phosphate waste material which contains approximately 25% apatite and is high in calcareous material has been evaluated in the laboratory as either a potential ameliorative material. This material is available at only 10% of the cost of the CODE 31. Functionalization of the cumulative acid load with respect to time and application schedule can be performed to evaluate the cost-economic engineering analysis for the material. It has been shown in laboratory studies that this material is, within certain ranges, an effective ameliorative material and exhibits a synergistic acid neutralization property due to the calcareous material present. In some instances, at higher application rates, however, this material has variable ameliorative and synergistic properties due to the ratio of apatite to calcite present in the phosphate waste tailings.

Additional Key Words: soxhlet extractor.

[1] Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23 -26, 1990.

[2] Walter Hart, Alfred Stiller, and Thomas Rymer are Graduate Student, Associate Professor, and Sr. Research Associate, respectively, in the Department of Chemical Engineering at West Virginia University, Morgantown, WV 26506. John Renton is a Professor in the Department of Geology and Geography at West Virginia University.

## Introduction

The potential use of phosphates as AMD ameliorants has been previously reported by Renton (1988). The primary phosphates considered were from North Carolina (Texas Gulf Code 31) and Florida (Florida Institute for Phosphate Research). A Commercial grade product, Texas Gulf Code 31, was most effective in inhibiting the oxidation of pyrite and improving leachate quality with respect to total acidity. The economics of the use of the Code 31 phosphate are presently under investigation. As a contingency, another phosphate-bearing material (provided by Texas Gulf) referred to as "waste phosphate tailings" was evaluated for potential amelioration. The economics associated with this material are extremely favorable ($5.00/ton). Therefore, in order for this material to be considered it must exhibit the same consistency and ameliorative properties as the Code 31 material.

## General Geology and Mineralogy

Large Miocene phosphorite deposits occur in parts of four states and extend along the Atlantic coastal plain from North Carolina into the center of the Florida Peninsula. The reserves and resources of this large deposit are estimated at several billion tons of phosphate rock. The bulk of the sediments consists of phosphate particles (pellets and other grain types), quartz, and clay minerals. There are locally important and variable amounts of calcite and dolomite with minor amounts of silicates, opal, iron-aluminum phosphates (in weathering zones) and heavy minerals (McClellan 1985).

The observed variations in mineralogy over the region are the result of primary compositional differences as well as post-depositional alteration. The latter includes diagenetic changes that resulted in the formation of authigenic minerals as well as weathering of the deposits to alter the compositions and phases to those that are seen today (McClellan 1985).

Carbonate fluorapatite (francolite) and clay minerals are the two most important constituents from an economic and environmental point of view. Francolite is the only economically important phosphate mineral present; the clays represent a significant source of impurities in the ore and present significant by-product disposal problems. Thus, this brief summary will concentrate on these two groups of minerals (McClellan 1985).

Reactive phosphate rock is an unaltered or low weathered fluorapatite of sedimentary origin in which a high degree of anion and cation substitution, mainly carbonate, has taken place in the lattice structure of the crystal. The predominant mineral of a highly reactive rock is francolite, a carbonate fluorapatite. In francolite, sodium, magnesium, potassium, strontium, zinc, uranium, etc. are substituted for calcium, while carbonate, sulfate, silicate, aluminate, etc. are substituted for phosphates in the crystal lattice (Texas Gulf 1989).

Replacement of calcium by sodium and magnesium in francolite is systematic but limited in the crystal structure. Sodium and magnesium are an integral part of the francolite structure and cannot be removed by benefication (Texas Gulf 1989).

The waste phosphate tailings, due to the variability of size distribution and general character, were theorized to have a mineralogy different from the materials presented above. The waste phosphate tailings are highly fossiliferous (containing, among other creatures, sharks' teeth and gastropods) and should contain high amounts of carbonaceous minerals with an assortment of cationic replacement. The major mineral, in addition to apatite and calcite, is Kutnahorite (an iron-manganese dolomite). The presence of a weatherable manganese mineral is of primary environmental concern.

The waste phosphate tailing material received by the laboratory was sized and each size fraction was subjected to x-ray diffraction analysis. The resulting mineralogy can be seen in Tables 1 and 2.

Table 1.

Size Fractions of Waste Phosphate Tailing and Associated Mineralogy

| size (mesh) | openings (mm) | mineralogical (table 2) sample id cross-reference | | |
|---|---|---|---|---|
| < 7 | > 2.830 | 1 | | |
| 7- 14 | 1.410 - 2.830 | 2 | | |
| 14 - 25 | 0.710 - 1.410 | 3 | | |
| 25 - 70 | 0.210 - 0.710 | 4 | | |
| 70 - 100 | 0.149 - 0.210 | 5 | | |
| 100 - 140 | 0.105 - 0.149 | 6 | | |
| > 140 | < 0.105 | 7 | | |
| whole rock batch I waste phosphate tailings | | 22 | 23 | 24 |
| whole rock batch II waste phosphate tailings | | 25 | 26 | 27 |
| Texas Gulf code 31 | | 28 | 29 | 30 |

Table 2.

Pertinent Mineralogy of Waste Phosphate Tailings.

| sample id | % apatite | %Kutnahorite | % calcite | %$SiO_2$ |
|---|---|---|---|---|
| 22 | 38.2 | 8.4 | 42.1 | 11.3 |
| 23 | 44.7 | 7.6 | 34.5 | 13.2 |
| 24 | 38.0 | 7.1 | 42.7 | 12.2 |
| 25 | 25.1 | 6.6 | 44.8 | 23.5 |
| 26 | 26.8 | 6.3 | 40.7 | 26.2 |
| 27 | 26.8 | 6.0 | 43.2 | 24.0 |
| 28 | 94.7 | - | 0.8 | 4.5 |
| 29 | 95.6 | - | - | 4.4 |
| 30 | 94.5 | - | - | 5.1 |
| 7 | 5.7 | 43.8 | 32.3 | 18.2 |
| 6 | 5.2 | 33.0 | 30.5 | 31.2 |
| 5 | 11.0 | 22.0 | 30.9 | 36.1 |
| 4 | 10.8 | 22.0 | 35.5 | 31.7 |
| 3 | 10.6 | 16.3 | 48.7 | 24.5 |
| 2 | 24.6 | 1.2 | 30.1 | 44.1 |
| 1 | 28.0 | 2.3 | 43.9 | 25.8 |

## Materials and Methods

### Experimental Design

Evaluating the effectiveness of the various forms of phosphate materials in ameliorating acid mine drainage was done by making use of the soxhlet extraction procedure developed by Renton et al. (1988). The soxhlet extractors are designed to simulate the weathering of mine spoil in the field. A representative sample of mine waste rock was selected from the Island Creek mine in Grant County, West Virginia. Samples were dried at room temperature and ground to pass through a seven mesh (2.83 mm opening) ASTM standard sieve tray. This particle size distribution was chosen to ensure that diffusive effects would be present as they are in the field. Phosphate rock samples were similarly selected and dried, but were ground to pass through a 140 mesh (0.105 mm opening) ASTM standard sieve tray. This particle size distribution was chosen to avoid diffusive effects in the phosphate rocks while keeping particle sizes large enough to reduce respiratory irritation. A set of four replicate samples of overburden and phosphate rocks were weighed to 0.1 gram sensitivity. The rocks were combined at a predetermined application schedule and mixed thoroughly. Rock mixture samples were contained in cellulose extraction thimbles and were alternately extracted and oxidized per Renton et al. (1988).

Code 31 phosphate is known to be an effective ameliorant in laboratory studies at application rates of two percent and above (Renton et al., 1988). Due to the differing chemical compositions of the phosphate tailings wastes, ameliorative effectiveness was expected to vary somewhat from the Code 31 product (as indicated above). The extreme cost advantage afforded by the phosphate tailings wastes would make much greater application rates of these materials economically feasible, if necessary. Consequently, it was decided to apply both the phosphate waste tailings (and the Code 31, for the sake of reference) at percentages up to double the determined optimum Code 31 application rate of two percent in increments of one-half percent. In addition to the Code 31, two different batches of phosphate waste tailings from the TexasGulf Chemicals Aurora, North Carolina mine were applied to the Grant County, WV mine spoil.

Distilled, de-ionized water was used as the solvent in the soxhlet extractors. Leachates were analyzed for pH, neutralization by base to pH's of 7.0 and 8.3, respectively, sulfate content, iron content and manganese content by standard procedures (Standard Methods for the Examination of Water and Wastewater) . Sulfate content was used by Renton (1988) to indicate acid produced by various overburden types. Analysis of waste tailings compositions showed the presence of sulfates in varying quantities which could lead to inflated estimations of sulfate attributed to the waste rock. Consequently, pH and neutralization by base to pH 7.0 and pH 8.3 were more reliable indicators of ameliorative effectiveness. Iron content was tested against the control to indicate any contribution of the phosphate waste tailings. Manganese is known to be detrimental to human health if present in certain quantities. Manganese content was analyzed to indicate the mobility of this ion which was found to be present in the phosphate waste tailings in the form of Kutnahorite.

A data spreadsheet was set up for each type of waste phosphate and each ion as shown in Table 3.

Table 3 Spreadsheet for data entry.

Type of Phosphate Material:
Replicate number:

| Percent Phosphate (wt. phosphate/100 g) | weeks 2 | 4 | 6 | 8 | 10 |
|---|---|---|---|---|---|
| 0 (control) | | | | | |
| 0.5 | | | | | |
| 1.0 | | | | | |
| 1.5 | | | | | |
| 2.0 | | | | | |
| 2.5 | | | | | |
| 3.0 | | | | | |
| 3.5 | | | | | |
| 4.0 | | | | | |

## Results and Conclusions

The data collected consists of approximately two megabytes and resides on the VAX 780 computer at the WVU College of Engineering. The data were processed on a MICROVAX 3500 Workstation, on a system freely programmed into working architecture by Mr. Paul Sutter, Apple Computer software engineer. All data analyses and graphics were performed by accessing the single database from various subprograms written and developed by the authors.

### Data Analysis

The three important aspects of the data analysis involve: (1) Examination of the consistency of the three phosphate types' ameliorative effects with respect to time and application rate (wt phosphate/100 wt refuse material), (2) Functionalization of ameliorative effects with respect to time and application rate, and (3) Evaluation of the functionalized ameliorative effectiveness and cost analysis.

The consistency with which the material alters the acid production with respect to time and application rate is an important parameter in evaluation of phosphate amelioration. A good ameliorative material is characterized by three properties: (1) the acid load will be inversely proportional to the application rate (2) the cumulative acid load will become asymptotic with time, and (3) the cumulative acid load will not show fluctuations or variability with respect to time and application rate. These properties tend to make curves of acidity versus time and application rate smooth and continuous for an effective ameliorant. A cost effective ameliorative material will show significantly decreased acid loads with low application rates. To illustrate the first of the properties cited above it is helpful to examine the ameliorative properties of Texas Gulf Code 31 phosphate material with respect to time and application rate. Averaged soxhlet cumulative acidity data are plotted versus time and phosphate application rate in Figure 1. Figure 2 is a grid version of the same data to show trends. Code 31 shows all three properties of a good ameliorative material in that it has an immediate ameliorative effect at low application rates with an increasing ameliorative trend at higher application rates, asymptotic behavior through time, and almost no fluctuations or variation through increased application rates or time. The ameliorative effects are easily functionalized. A plot of data generated from the resulting function can be seen in Figure 3.

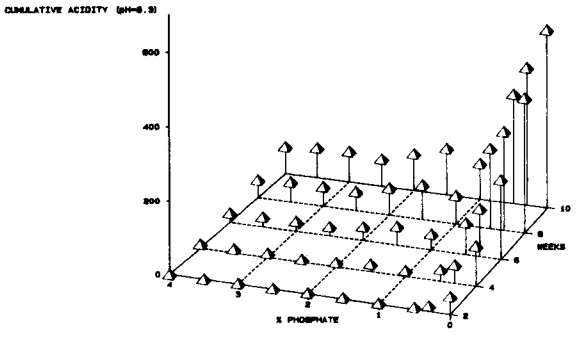



Figure 1.

Ameliorative effectiveness of Texas Gulf Code 31 as it relates to weeks in the soxhlet extractors and the percentage of material added to the toxic waste refuse material.

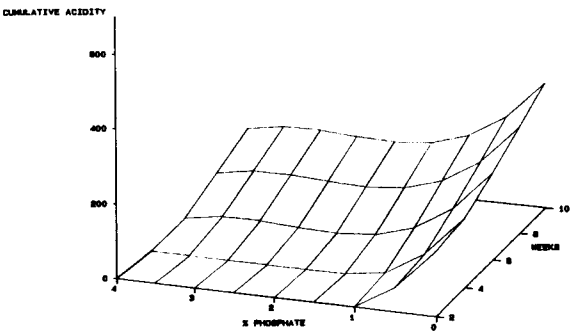



Figure 2.

Gridded version from Figure 1 data to show contour relationships between ameliorative effectiveness of Texas Gulf Code 31 with respect to weeks and application rate.

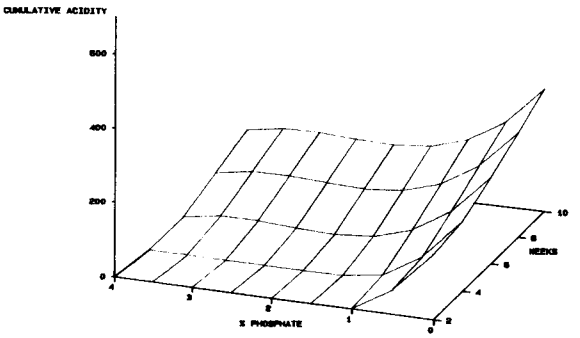



Figure 3.

Functionalized 3-dimensional relationship between ameliorative effectiveness of Texas Gulf Code 31 with respect to weeks and phosphate application rate. The function is that given by equation 1.

Similar treatment can be given to the data for the Batch I and Batch II waste phosphate materials. Batch I plots are seen in Figures 4 to 6 while Batch II plots can be seen in Figures 7 to 9, respectively. Figures 1, 4, and 7 represent actual cumulative acidity data as it relates to time (weeks) and phosphate application rates of each rock type. Figures 2, 5, and 8 show the 3-dimensional grids of the averaged experimental data. Figures 3, 6, and, 9 represent the results of functionalization (i.e. cumulative acidity deterministic equation as a function of time (weeks) and phosphate application rates [%phosphate]). Note that the functionalized equations are mathematically smooth.

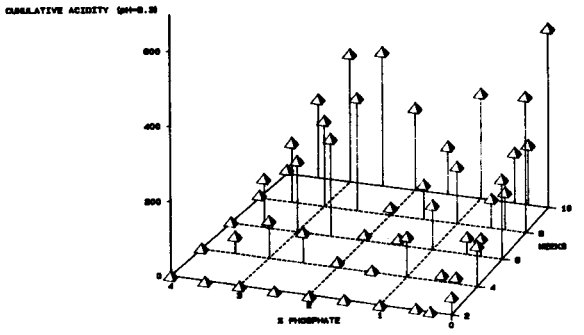


Figure 4.

Ameliorative effectiveness of Texas Gulf Waste Tailings (batch I) as it relates to weeks in the soxhlet extractors and the percentage of material added to the toxic waste refuse material.

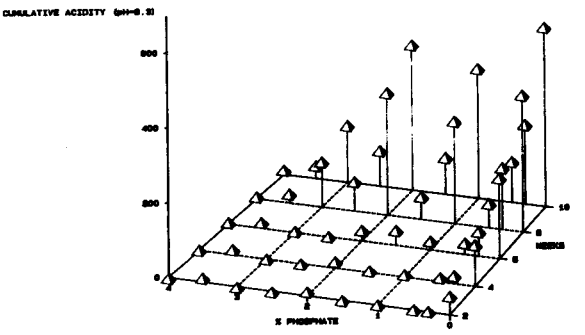


Figure 7.

Ameliorative effectiveness of Texas Gulf Waste Tailings (batch II) as it relates to weeks in the soxhlet extractors and the percentage of material added to the toxic waste refuse material.

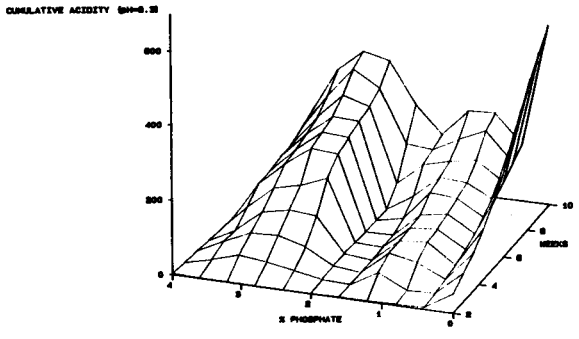


Figure 5.

Gridded version from Figure 4 data to show contour relationships between ameliorative effectiveness of Texas Gulf Waste Tailings (batch I) with respect to weeks and application rate.

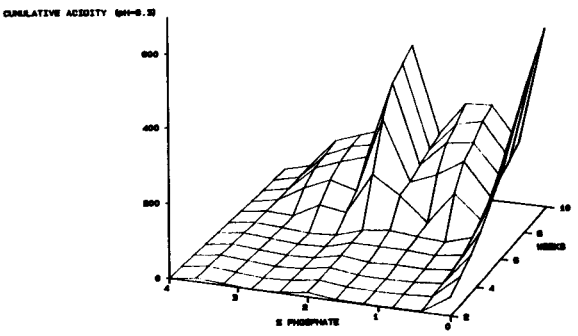


Figure 8.

Gridded version from Figure 7 data to show contour relationships between ameliorative effectiveness of Texas Gulf Waste Tailings (batch II) with respect to weeks and application rate.

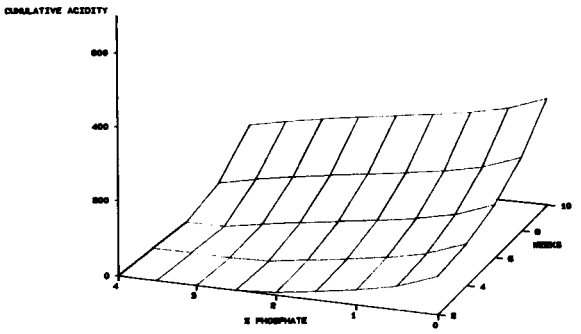


Figure 6.

Functionalized 3-dimensional relationship between ameliorative effectiveness of Texas Gulf Waste Tailings (Batch I) with respect to weeks and phosphate application rate. The function is that given by equation 2

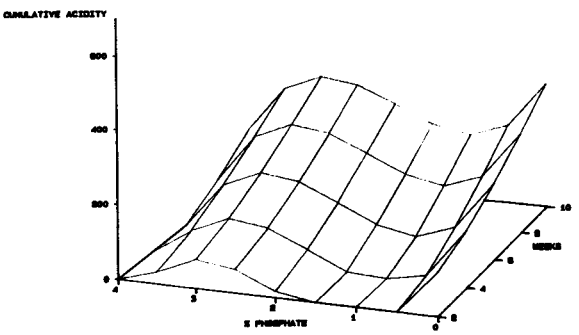


Figure 9.

Functionalized 3-dimensional relationship between ameliorative effectiveness of Texas Gulf Waste Tailings (Batch II) with respect to weeks and phosphate application rate. The function is that given by equation 3.

Manipulation of the three deterministic equations relating the cumulative acidity (for each phosphate type) yields information not obvious from the graphical representations of Figures 1 through 9. The deterministic equation for each phosphate type is given below:

Code 31:

$$A_{31} = -0.0095 \times t^3 + 0.065 \times t^2 + 33.26 \times t - 13.24 \times p^3 + 104.77 \times p^2 - 261.27\,p + 206.26\,(1/t) \quad (1)$$

Waste Phosphate Batch I:

$$A_1 = -0.10 \times t^3 + 3.49 \times t^2 - 136.60 \times p^3 + 207.48 \times p^2 - 273.39\,p + 102.32\,e^{-p} \quad (2)$$

Waste Phosphate Batch II:

$$A_2 = 0.21 \times t^3 - 0.65 \times p^3 - 11.20\,p + 98.67\,(1/p) \quad (3)$$

where:

$A_i$ = cumulative acidity for phosphate material, i
p = phosphate application rate (parts/100 parts overburden)
t = weeks in soxhlet reactor simulator

A more general function can alternatively be developed using

$$A(r) = A_0\, e^{-wr} \quad (4)$$

where:

A(r) = the cumulative acid load as a function of application rate,r
r = application rate for phosphate addition
$A_0$ = the maximum potential acid load
w = constant

## Economics of Phosphate Amelioration

The generation of functions to estimate the milligrams of calcium carbonate required to neutralize the sulfate effluent from the soxhlets makes it possible to compare the cost effectiveness of the phosphate rock types based on costs of materials and to estimate material costs on a field scale. The total materials cost associated with phosphate amelioration includes both the cost of the phosphate rock and the cost of the neutralizing material. The phosphate is applied only once and thus represents an initial cost. Neutralizing material must be applied until the acid load is below some specified value which may be determined by regulation. The cost for each material at any time can be found by multiplying the amount of material used by the unit price of the material. Thus, the total cost at time t can be written

$$\$_{T,i}\,(p_i, t) = U_{P,i}\, p_i + U_B A_i\,(p_i, t) \quad (5)$$

where $\$_{T,i}$ = Total cost (dollars)
$U_{P,i}$ = Unit price of phosphate i (dollars per gram)
$p_i$ = Phosphate amount (grams per hundred grams overburden)
$U_B$ = Unit price of base neutralizer (dollars per milligram)
$A_i$ = Base neutralizer amount (milligrams)

This function for total cost is valid over the same range as $A_i$ , but extrapolation is only as good as the function used to fit the curve.

With Equation 5, costs can be generated for an infinite number of combinations of phosphate types, phosphate application rates, and times. Therefore, a time, $t_0$, which produces a fair comparison among phosphate types must be used. Since the maximum allowable acidity may vary in different areas, the time it takes for a system to approach any such value may be of limited general interest, and it is useful to find a less arbitrary time period. Unameliorated systems will eventually approach a non-zero, but nearly constant level of acid production. Any useful ameliorant should stabilize the system long before this time is reached. The appropriate time is found when the change in $A_i$ with respect to time is very small, or

$$dA / dt\,|_{p=0} \;\Longrightarrow\; 0 \quad (6)$$

Note that the "i" subscript is unnecessary since zero percent phosphate is the control for all cases. For practical purposes the time, $t_0$, can be taken when the derivative function attains some predetermined tolerance limit about zero. The derivatives can be calculated numerically from finite differences between values generated from the function A ( p=0, t).

With the value of $t_0$ determined, values of total cost, $\$_{T,i}$ , can be generated for the various phosphate types at different application schedules. For each phosphate type, the total cost can be minimized with respect to percent phosphate applied by satisfying the following conditions:

$$d\,\$_{T,i} / d\,p_i\,|_{t=t0} \;\Longrightarrow\; 0 \quad (7)$$

$$d^2\,\$_{T,i} / d\,p_i^2\,|_{t=t0} > 0 \quad (8)$$

This can be done numerically analogously to the acidity ($A_i$) derivative above. With the minimum total cost known for each phosphate type, the optimum choice among the three types is simply the low-cost option.

## Determination of Long-Term System Stability

As mentioned above, good ameliorants should make cumulative acid load asymptotic with time without much fluctuation or variability. Laboratory soxhlet testing is an excellent predictor of acid producing potential for the times corresponding to the actual times the soxhlets were run (Renton 1988). Extrapolation is not difficult since the systems are well-behaved and stabilize quickly. For ameliorated systems,

behavior of the system depends on the type of ameliorant. Some materials tested as ameliorants have actually had an accelerant effect on acid production (McConaghy 1985). As indicated above, the waste phosphates tested here are not pure apatite, and other components present may diminish effectiveness. Accelerants increase acid production and might be expected to delay or inhibit stabilization of the system since the change in acidity with time may not approach zero or a constant value. Exhibition of this behavior may indicate that an accelerant is present in the waste phosphate to a significant extent. At a minimum, it indicates that the system cannot be accurately predicted without a longer testing period.The apatite/calcite ratios shown in Tables 1 and 2 could demonstrate this.

The final result of the economic analysis can be seen in Figure 10. This shows the total cost of ameliorant (including treatment), the cost of water treatment chemicals, and the cost of phosphate ameliorant material. The vertical line indicates the optimum (minimized) total cost. In this case the phosphate application rate corresponds to 2.5 wt percent phosphate. The toxic rock in question contains 2% total sulfur and the graphical output corresponds to a mass of 200 million tons of this toxic material for which treatment is demanded.

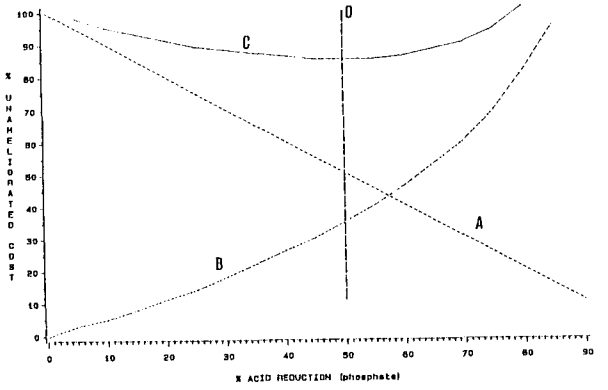



Figure 10.

Cost curves showing the total treatment cost using a combination of water effluent treatment and phosphate amelioration. Curve "A" represents the cost of effluent treatment using standard treatment chemicals (NaOH or CaO). Curve "B" represents the cost of phosphate amelioration material (Code 31). Curve "C" represents total cost of the combination phosphate amelioration material and effluent treatment chemical. The line "D" represents the optimum level of phosphate treatment with respect to total cost. All costs are expressed as a percentage of the cost of water effluent treatment using no phosphate amelioration.

REFERENCES


Renton, J., Rymer, T., Stiller, A., A Laboratory Procedure to Evaluate the Acid Producing Potential of Coal Associated Rocks, Mining Science and Technology, Elsevier Sci. Publ., 7, 1988, p. 227.

McClellan, G., Kauwenbergh, V., Ispording, W., An Overview of the Mineralogy of the Phosphorite Deposits of the Southeastern United States, International Geological Correlation Program, Project 156, 8th International Field Workshop and Symposium (Southeastern U.S.), Guidebook, Scott Snyder, ED., 1985,

Texas Gulf Chemicals, Reactive Phosphate Rock, Unpublished Manuscript, 1989.

Standard Methods for the Examination of Water and Wastewater, ed. by Greenberg, A, Conners, J., and Jenkins, D., American Public Health Association, 1980.

McConaghy, B., Analysis of Novel Acid Mine Drainage Experiments, M. S. Thesis, Department of Chemical Engineering, West Virginia University, 1985.

This page intentionally left blank.

# ADVANCES IN THE PREDICTION AND CONTROL OF ACID MINE DRAINAGE [1]

By P.F. Ziemkiewicz [2]

ABSTRACT. Over the past five years a research team at West Virginia University has developed a mathematical model: the SSPE/PSM which appears to yield significant improvement over current methods of predicting the production of acidity from pyritic coal spoils. The team has also developed a technology which, at the laboratory and small scale field level controls the formation of acid mine drainage (AMD). Another technology may significantly reduce the cost and upgrade the performance of AMD water treatment systems. The objective of these projects is to move laboratory technologies to the field test stage and to identify whether they have the potential for development into practical, effective systems for the coal industry.

This paper discusses the status of these technologies and the strategy being used to test them and, if proven practical, to bring them to application.

Additional key words: Acid mine drainage, phosphate, prediction model.

## Introduction

Acid mine drainage (AMD) is the most significant environmental impact resulting from both surface and underground mining in the northern appalachian coal basin. AMD ruins fisheries and recreational lakes, damages structures, increases the cost of municipal water treatment, degrades the value of lands for the citizens of the state and lowers its potential for tourism.

Current AMD control technologies are expensive and they require an indefinite period of maintenance. On active mining sites costs are borne by the company. On abandoned mine lands the polluted water generally flows directly into adjacent streams. Some appalachian coal reserves are simply too prone to AMD production to consider mining. Companies are reluctant to assume indefinite liabilities associated with development of the coal fields and governments are reluctant to approve permit applications which are likely to result in bond forfeiture. Such reserves are effectively sterilized until successful AMD control technologies are developed and implemented.

AMD results from the weathering of pyrites in mine spoil and refuse. Weathering results in production of acid forming salts which, over time, combine with water to yield AMD. Current AMD treatment technology for active mine sites involves the continual addition of alkaline materials such as hydrated lime, lye or ammonia to drainage waters in an effort to neutralize their acidity. This current technology allows the acid forming process to run its course and then, as rainwater and snow melt percolate through the dump, it treats the resulting discharge. Many products have

[1] Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990

[2] Paul F. Ziemkiewicz is the Director of the National Mine Land Reclamation Center, West Virginia University, Morgantown, WV 26505.

been evaluated over the years to control and several have shown promise. While several products are under evaluation, there is presently no widely accepted technology which controls AMD production at the source without indefinite maintenance.

The AMD problem is subject to increasing public scrutiny and controversy. In fact, a regulatory issue exists because there is no technical solution to the AMD problem. Regulatory solutions to the problem will tend to be ineffective and punitive as long as there is no practical control technology.

The objective of this discussion is to describe development and field testing of two promising AMD control technologies: Phosphate application and a low-voltage electrolytic cell technique.

## Phosphate Technology

A team of researchers at West Virginia University has been studying the AMD problem for several years. The team, consisting of J.J. Renton, A.H. Stiller and T.E. Rymer, has recently completed several breakthroughs which may radically change the AMD picture in the future.

1. The team has developed a means of significantly restricting the reaction in mine refuse. The method involves addition of small amounts (2-3%) of rock phosphate to the refuse. The effectiveness of rock phosphate has been demonstrated in laboratory and small scale field trials.

2. A simulation model has been developed which predicts the rate at which given rock units will produce AMD. This model further has the capability to predict the rate at which AMD will leach from a given dump. Used in conjunction with the ability to restrict AMD production through rock phosphate addition, this model estimates how much phosphate will be necessary to control AMD production on a given site. This model would permit a reliable estimate of: a) the AMD hazard for any new development and b) the costs associated with controlling AMD.

Status of Phosphate Technology: The effectiveness of the AMD chemical model has been verified against historical AMD field sites. Predictions made by the model regarding the status of AMD at a given time after dump construction have been highly accurate. To date, however, most of these sites have been dormant for some years so it has not been possible to determine the early pattern of AMD discharge.

Before recommending acceptance of phosphate, however, five major issues need to be resolved:

1. Source of phosphate-in addition to commercial rock phosphate, the potential of low grade (low cost) phosphate rejects or ores is of great interest;

2. Application methods under mining conditions;

3. Impacts on prep plant and material handling systems at the mine;

4. Costs in terms of capital and operating outlays and in terms of any reduced efficiencies at the mine;

5. Effectiveness in controlling AMD production under operating conditions.

Evaluation of these factors can only be made by applying phosphate in conjunction with an operating mine/prep plant. It should also be stressed that phosphate is not being used as an acid neutralizing agent, like limestone. Rather, it coats pyrite particles and their air channels with insoluble iron phosphate salts physically isolating pyrite from water and oxygen. Neutralizing agents such as limestone, on the other hand permit the oxidation of pyrite to occur, then react with the products. This is why neutralizing agents are usually required in large amounts.

Pyrite oxidation in coal wastes tends to be a rapid process (Rymer et al. 1988 reported 65% oxidation within 200 days after initial exposure to air). So, it is unlikely that addition of low rates of phosphate to already oxidized rock will have much beneficial effect.

## The Phosphate Study

Several companies and government agencies are participating with West Virginia University in testing this technology. This is an important point because it is nearly impossible to develop a practical reclamation technology without the advice, guidance and financial support of the regulatory agencies and the industry.

Major financial support has been provided by the West Virginia Department of Energy, Mr. George E. Dials, Commissioner. The Department's participation will ensure that the program addresses the key issues facing the regulatory agencies. Two coal companies are participating in the phosphate evaluation program: Island Creek Coal Co. and Leckie Smokeless Coal Co. In addition, Texasgulf Inc. the supplier of the phosphate is a major contributor to the program. Analabs Inc., of Beckley, West Virginia, is contributing through its sampling and analytical support to the project. This group of companies will ensure that project technology will be transferred directly to each of the key elements within the industry.

Refuse piles will be constructed at each mine. At one site a series of 4000 Ton piles of run of plant refuse are being constructed on prepared, plastic lined

foundations. Each pile has a single drainage point for collecting and monitoring of effluent. These piles will be treated with three levels of phosphate: 0%, 2% and 4% commercial rock phosphate added per ton of refuse. At the other site, two 100 000 Ton piles will be constructed in similar manner. One will be treated with 3% phosphate and the other will be the untreated control. Both mining operations are primarily underground operations mining high to mid-sulfur coals.

The refuse will consist of a blend of refuse materials as produced by each company's preparation plant.

Application of the phosphate will be tailored to the company's operations. At one site phosphate will be applied as a slurry and at the other site it will be applied dry to the prep plant's refuse bin via a pneumatic feed.

In addition to testing the effectiveness of phosphate application, the project will allow us to verify model's estimates against field observations. If successful, this package will allow site-specific estimates of refuse pile behavior during the permitting phase of the development.

This project will generate a tremendous volume of data. Without sophisticated data handling and treatment systems the project would generate clouds of incomprehensible numbers. The value of a systematic approach to data collection, storage, retrieval, reduction and analysis is recognized by treating data processing as a distinct task. Five database management programs will be applied to the project. In addition, ten data processing software packages will be used, seven of which will either be developed specifically for the project or modified for the project.

## Electrolytic Control of AMD

Many abandoned underground coal mines continue to discharge large volumes of AMD decades after abandonment. Once an underground mine is completed the company is left with few options in the event that AMD becomes a problem. The general solution is to institute an AMD neutralization program consisting of the addition of an alkaline agent such as soda ash briquettes, caustic, ammonia or quicklime coupled with a sedimentation pond to capture the resulting iron flocs (yellowboy).

These systems are expensive to operate and the duration of treatment is indefinite. The less expensive neutralizing agents such as quicklime require large, capital intensive treatment plants while the more expensive agents such as caustic and ammonia can operate on inexpensive capital facilities. Costs in the range of $5,000 to $20,000 per month are not uncommon for mid scale AMD neutralization systems.

## AMD Control for Abandoned Underground Mines

Nearly 80% of West Virginia's AMD comes from underground mines. Many were abandoned decades ago and continue to produce toxic discharges. While the previous discussion on phosphate described a method for preventing AMD production from fresh refuse, this discussion focuses on an inexpensive method for treating AMD both for active and abandoned operations. If successful, it would have application to AMD environments where oxidation had already occurred or where the acid producing rock was inaccessible.

After completion of mining, the portals are usually sealed and a pool of groundwater forms in the downstream section of the old workings.

This water often does not meet state water quality specifications with respect to acidity, iron and manganese. Typically, a neutralizing agent is added to the water upstream of a settling pond which removes the flocculated metal sludge. The neutralizing agents are expensive but they serve two purposes when admixed to AMD streams: (1) They complex the soluble iron which drives ferrous oxidation to the ferric ion. (2) They increase the pH in the near vicinity of the addition site very dramatically. The total effect is to force the precipitation of ferric oxide and further neutralize the water. The sediment quality of the precipitate is strongly dependent upon the availability of dissolved oxygen. At the addition site ammonia replaces the dissolved oxygen so the rate of floc consolidation is slow. This can be shortened by rapid aeration immediately after ammonia addition.

An electrolytic process has been developed at the Chemical Engineering Department of West Virginia University as an alternative to chemical neutralization. It is known as the AMD-CELL SYSTEM. Electrolytic processes have been attempted in the past but have usually failed to operate economically due to their high current consumption. The proposed technology operates on a low current (<2 amps) format and has been developed and run successfully in a lab bench demonstration unit. The objective of this project is to evaluate, in a joint effort with the mining industry, the scale-up potential of this technology, its effectiveness and practicality under operational field conditions. As of this writing a large scale field trial is being developed with a West Virginia coal mining company.

A local AMD water which is both acidic and high in manganese and iron has been used to evaluate the prototype unit. The prototype is a short unit and it

compensates by recirculating the AMD. Results indicate that within two minutes of treatment both iron and manganese ions have been precipitated below State water quality specifications and the pH has been raised from about 2.5 to 4.2. This alone would lower by two thirds the amount of neutralizing agent required to bring the water into compliance. More significantly, it removes the iron and manganese.

There remain several issues which need to be evaluated. The first is ion concentration. This is important because it determines the voltage requirement for the cell. Ideal voltages at standard conditions can be found in tables; however, these voltages change according to ion concentrations. After these are known a simple calculation can be used to determine the voltage required to activate the AMD-CELL SYSTEM to a specific type of AMD (e.g. simultaneously oxidize ferrous iron and reduce hydrogen ion).

The total conductivity and specific ion conductivity must also be determined. These values can be obtained from solution analysis and standard tables. The conductances are important because they set the current requirement, which, in turn will be coupled to the AMD flow rate. This should make sense, since current and flow rate are time dependent functions.

Once the voltage and current parameters are determined the cell can be tailored to the specific AMD discharge point. A circuit can be made which prevents the voltage from exceeding the required potential. This unit will be analogous to a voltage regulator. It is important to prevent side reactions, for example, reduction of ferric ion back to ferrous ion.

The next step tests the specific AMD water in the bench scale unit. The unit is used to design the voltage and residence time required of the field system.

## Experimental Design

The electrolysis system should precipitate iron from the water as ferric oxide. The other electrode reaction is the reduction of hydrogen ion. In other words the hydrogen ion, which is responsible for the low pH would be removed from the stream as insoluble gaseous elemental hydrogen. The end result would be a stream which is low in soluble iron, ferrous ion, and high in pH.

The AMD stream will be passed into one or the other of two tubes. One of the tubes is a electrolysis cell, the other is a control tube. This will ensure accurate evaluations of the effects of electrolysis of the water. Each tube is constructed from 10 inch or 8 inch internal diameter ABS or PVC tubing. The tubes are 10 feet long. This will provide a working space of about 8 feet for the electrodes. The reactors will be constructed in a manner identical to that of the laboratory prototype.

Electronic circuits may have to be designed and constructed so the ideal lab parameters can be duplicated in the field units. At the present time it is not possible to define the circuit without any lab data.

## Laboratory Experimentation

A series of tests at different voltages, residence times, and frequencies will be run to determine the most effective conditions for metal and hydrogen removal. These experiments will demand a complete analysis of iron ion, sulfate ion and sodium hydroxide equivalents to overcome buffering and to achieve neutrality for each sample. From these data the most effective configuration for the particular company's AMD system can be developed.

## Field Work

The field work will involve the construction of a scaled up field unit. The circuitry which would provide the proper voltage at the ideal frequency and permit current flow would be built. The AMD stream would be allowed to flow through the cell. The flow rate would be recorded. These data would be compared to the expected values calculated from the lab bench scale experiments. Samples of mine effluent would be obtained for comparison to demonstrate the effectiveness of the system. The flow rate could be varied by splitting the incoming stream and the effectiveness of the treatment could be validated at different flows. These again could be used to test the validity of the laboratory bench tests.

## Literature Cited

Rymer, T.E., Renton, J.J. and Stiller, A.H. 1988. A computer simulation probability model for geochemical parameters associated with coal mining operations. USDI Bureau of Mines Information Circular 9183. p 76-82.

# NEUTRALIZATION AND STABILIZATION OF COMBINED REFUSE USING LIME KILN DUST AT HIGH POWER MOUNTAIN[1]

by

Douglas H. Rich and Kenneth R. Hutchison[2]

Abstract. High Power Mountain Corporation (HPM), a large coal surface mine in central West Virginia, disposes of filter cake and coarse refuse by means of "refuse cells". These cells are designed to encapsulate the combined refuse, isolating the material from the hydrological system, and provide an environmentally sound method of refuse disposal.

Lime kiln dust, a by-product of the lime industry, is admixed with the refuse to:

- o Increase pH and alkalinity,
- o Inhibit bacterial growth, and
- o Limit formation of acid water.

In addition, the lime kiln dust has the added benefit of stabilizing the combined refuse. The requirement of "rock bridges" needed to support 85 ton refuse trucks in the cell is no longer necessary. Therefore, the volume of the refuse cell is totally utilized by refuse and not a combination of the refuse and the rock bridge material.

## Introduction

High Power Mountain (HPM), located in the western portion of Nicholas County, is approximately sixty five miles east of Charleston, West Virginia. Summersville, the county seat of Nicholas County, is located approximately twelve miles east of the mine site.

HPM is a mountaintop removal operation mining seven seams. Also, a contour/highwall operation mines a lower seam for a total of eight seams. The mining is performed for HPM by a contract miner, High Power Energy (HPE), and the preparation, shipment and sales are performed by HPM. In 1990 approximately 2.3 million tons of coal will be shipped from the property. At this shipment level, upwards of 900,000 tons of combined refuse will be generated.

## Mining

Surface mining is performed by front-end loaders, shovel, and trucks. The overburden is removed by a 19 cubic yard Demag hydraulic shovel and 13.5 and 22 cubic yard front-end loaders. Trucks consist of 85 ton Caterpillars and 200 ton LeTourneaus. The majority of the overburden is a very fine to fine grain, buff colored, micaceous sandstone. A minor amount of shale is present. Cast blasting is performed in selected mining areas.

The seven coal seams mined by the mountaintop removal method are the:

- o Upper Kittanning
- o Middle Kittanning
- o Lower Kittanning (5-Block)
- o Clarion
- o Stockton A
- o Stockton
- o Coalburg

The lowest seam mined at the HPM is the Winifrede seam. Mining of this seam is by the contour method followed by highwall mining using the Metec mining system.

The coal seams are in the upper portion of the Pottsville Series and the lower portion of the Allegheny Series of the Pennsylvanian Era. Most seams are multi-benched, ranging from two to five benches, with intercalated shale partings. Because the quality of the coal varies both horizontally and vertically, the coal is mined by split loading of the benches. Prior to coal loading, the shale partings are removed. Typically, the loading operations consist of twenty to twenty-five

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, April 23-26, 1990, Charleston, West Virginia.

[2]Douglas H. Rich, Geologist, Supervisor, Planning and Development, High Power Mountain Corporation, Drennen, WV 26667; Kenneth R. Hutchison, Geologist, Senior Sales Representative, APG Lime Corp., Ripplemead, VA 24150.

different loading sequences for all eight seams. Coal loading is accomplished by a 26 cubic yard coal bucket attached to a Caterpillar 992 front-end loader, loading into 85 ton Dart coal haulers.

## Preparation

The coal is mined, weighed and delivered to the HPM preparation plant, crushed to minus eight inches, and transported via two 42 inch belts to one of three raw coal stockpiles. Under the stockpiles a 42 inch reclaim belt delivers the coal to the plant crushing and screening station. The coal is crushed to minus two inches (-2") before being washed.

The plant is computer controlled and consists of three parallel washing circuits. The heavy media cyclones clean the coarse size fraction (2" x 28 mesh). The fine coal circuits consist of hydrocyclones and classifying cyclones for the 28 mesh x 0 inch material. During cleaning the minus 100 mesh material is not recovered. The fine reject material is pumped to the refuse thickener, combined with flocculants to enhance settling, and pumped to one of three belt filter presses. At the belt filter press, additional flocculant is added and the slurry is pressed and de-watered. The filter cake, approximately 28-30% moisture, falls on to the filter cake belt which transports the cake to the main refuse belt. At this point the filter cake is combined with the coarse refuse.

The clean coal is stored at the plant in one of three eight thousand clean coal stockpiles. From here, the coal is hauled to one of three stockpiles at the unit train loadout facility. Train loading is at a rate of 4,500 ton per hour using a computer batch-weighing, flood-loading system and a continuous railroad loop. Loading a 13,500 ton train typically takes 3 hours.

## Methods and Materials

### Refuse Cells

The refuse cell concept is new to the coal industry and is based on the idea of encapsulating the refuse material and isolating it from the environment. The refuse cells are placed in mined out areas on the tops of the mountains. Areas designated for future refuse cells are mined by mountaintop removal and the coal is removed to the pavement of the lowest seam. Construction of the cell begins by the placement of a berm around the cell area to contain water and divert it to the sediment control structures. Once the berm is constructed, an inner dike of impervious material is built to contain the refuse. A ten foot layer of large shot rock is placed on the floor of the cell, providing a french drain. This disrupts the capillary action by groundwater from the pavement to the cell property. On top of the shot rock, a three to five foot layer of shale is placed to make an impervious layer.

Once the initial work is complete, refuse is hauled into the cell and dumped by the 85 ton trucks. When a layer of refuse has been placed, the next higher level is constructed using a second inner dike structure. The process is repeated until design height is attained.

When the cell is full, the top and sides of the cell are covered with shale in a dome shape. The entire cell is covered with two to four feet of soil and reclaimed with grasses. The shale, soil, and dome shape help divert rainfall to the outside berm and reduce infiltration of water into the cell. Thus, the refuse becomes encapsulated and, after interstitial moisture is squeezed out, the long term environmental impact is minimized or eliminated. If water from the cell becomes a problem, the outside berm structure captures and diverts the water to a central pond where treatment can occur.

### Refuse Quality

The nature of the refuse varies and is a function of the particular coal seam or split being cleaned. The amount of fines in the static thickener determines the activity level of the three belt filter presses. Typically, the filter cake fraction is 20% to 35% of the refuse. However, there are times when the refuse is either all filter cake or all coarse material. The percentage of pyritic sulfur varies in the refuse. The vast majority of the coal seams at HPM are low sulfur (<1%) and thus the refuse generally contains less than 1% pyritic sulfur. However, the Stockton seam, representing about 5% of the reserve, is high in sulfur and when it is cleaned with the other coals the pyritic sulfur of the refuse climbs to over 1%.

When the project started, the refuse was tested to determine the quality and the acid producing potential of the material. Sturm Environmental Services, Inc. was contracted to do both laboratory bench testing and actual field testing. The lab testing consisted of acid-base accounting and column leaching. These tests showed the refuse to be 26 to 53 tons per 1000 tons of material $CaCO_3$ equivalent deficient. Thus, the refuse required neutralization treatment.

Field testing was developed by placing low (0.9%), medium (1.1%), and high (2.5%) sulfur refuse in three separate test areas. These areas were constructed by placing a small two foot berm around a 15 x 20 foot area lined with plastic. A discharge pipe was placed through the lower portion of the berm and water samples collected. The graph in Figure 1 delineates the change in pH over time. As shown, the water acidified very quickly.

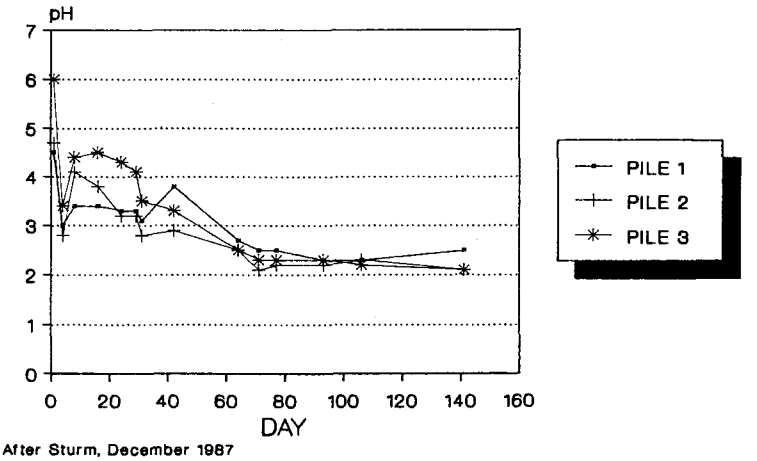
The next step was to determine the method needed for neutralization and the associated economics. This portion of the study concentrated on applying five different neutralizing agents to a bulk sample (25-30 tons) of refuse and testing the effectiveness of each.

Agents tested include:

- Ag Lime
- Spent Lime
- Rock Dust
- Rock Phosphate
- Lime Kiln Dust

A second investigation concentrating on bacterial growth was conducted to establish the relative amounts of iron oxidizing bacteria and the effects of changing from an acidic to alkaline environment. With the exception of the rock phosphate, all the agents neutralized the refuse

## WATER QUALITY - LEACHATE DATA




After Sturm, December 1987

FIGURE 1

but required various rates of application. Also, the bacterial growth was inhibited by increasing the alkalinity of the interstitial moisture.

### Results and Discussion

#### Initial Refuse Treatment

After internal discussions a decision was made to treat the refuse by use of limestone rock dust. The method employed was to treat the refuse after the trucks dumped the material into piles within the cell. The limestone rock dust was slurried and "hydroseeded" on top of the piles. This allowed water to be buffered prior to entering the underlying piles. This method worked with somewhat satisfactory results but did not allow for a good mix of the limestone rock dust in the internal portions of the piles.

Limestone rock dust was used during the construction of the first refuse cell and a portion of the second cell. As an added precaution, when the final layer of refuse was placed in the first cell, a bacterial inhibitor was applied to the top of the refuse to further reduce bacterial growth. Even at an added cost, it was felt the bacterial inhibitor would provide additional protection prior to covering the cell with the dome material. This cell, completed in September 1987, has been reclaimed and seeded. One small leak in the upper inner dike system occurred; however, it has since stopped. This leak was attributed to the weight of the overlying material squeezing the interstitial moisture out of the refuse. The water from the leak was somewhat acidic but was contained within the berm and pond system, neutralized and released.

#### Operational Problems

During the construction of the first cell and a portion of the second refuse cell, two problems existed which needed to be addressed:

1) Volume of rock between refuse layers required to support 85 ton trucks, and

2) Increase efficiency of neutralization.

First, to support the weight of fully loaded (gross weight approximately 190 tons) refuse trucks on top of the combined refuse, it was necessary to place a 5 to 7 foot rock "bridge" between the refuse layers. This rock "bridge" allowed the trucks to drive on top of the refuse but it also consumed over 50% of the volume within the cell. The active life of the cells and the ability to handle refuse was therefore substantially reduced. Besides the shortened life of the refuse cell, the required rock material added to the overall cost of cell construction.

The second problem was to evaluate and develop a better neutralization system so as to provide for greater admixing of the neutralizing agent and refuse constituents. This evaluation led to serious consideration of lime kiln dust. Also the decision was made to apply the dust directly to the refuse from a bin situated over the main refuse belt. Application was to be made prior to the refuse entering the refuse storage bin so as to take advantage of the mixing that occurs during movement of the material.

#### Lime Kiln Dust

Lime kiln dust, a product of the lime industry, is the by-product of calcining limestone in rotary kilns. The gasses and dust generated from this process are directed through a baghouse where the dust is collected and the gasses vented to the atmosphere. This dust material contains 15% to 18% calcium oxide (CaO), 70% to 75% calcium carbonate ($CaCO_3$), and the remaining portion is fly ash (high in silica and alumina). Lime kiln dust is alkaline, producing a pH of 12.4 in a saturated solution. In addition to its obvious neutralizing potential, lime kiln dust also acts as a drying agent, forming calcium hydroxide when it comes in contact with water.

The mechanism for the stabilization process is the pH increase which causes the calcium hydroxide to combine with the fly ash and clay minerals in the refuse. The soil-lime reactions are complex and not completely understood. According to Chou (1987), an oversimplified qualitative view of some typical soil-lime reactions are:

- $Ca[OH]_2 \rightarrow Ca^{++} + 2[OH]^-$
- $Ca^{++} + 2[OH]^- + SiO_2$ (clay silica) CSH
- $Ca^{++} + 2[OH]^- + Al2O_3$ (clay alumina) CAH

J.L. Eades (1962) suggests that the high pH causes silica from the clay minerals to dissolve and, in combination with Ca++ form calcium silicate. Diamond, et al. (1964) theorized that lime molecules are absorbed by clay surfaces and react with other clay surfaces to precipitate reaction products. These studies suggest the clay lattice components are "dissolved" from the clay structure and are reprecipitated as CSH and CAH. Stocker (1972) proposes the "diffuse cementation" theory in which lime reacts directly with clay crystal edges, generating accumulations of cementatious material. It appears that chemical reactions occur and new phases nucleate directly on the surfaces of the clay particles.

The reaction of the calcium hydroxide with the clay mineralogy of the refuse and fly ash not only produces the required neutralization but also the stabilizing effect. The calcium hydroxide has the added benefit of reacting with available sulfate radicals to produce the mineral gypsum (calcium sulfate).

Application of Kiln Dust

The lime industry is under environmental constraints as is the coal industry. Whereas coal refuse generally tends to be acidic, lime kiln dust produces alkaline conditions. By addressing the by-product problems (acidity and alkalinity) of both industries it appeared a symbiotic relationship existed. Besides the neutralization aspects, the use of lime kiln dust with its stabilizing properties appeared to offer HPM a solution to the volumetric and cost problems resulting from the rock bridges within the cells.

In the third quarter of 1988, HPM began the process of applying lime kiln dust to the refuse. The lime kiln dust, obtained from APG Lime Corp., located in Ripplemead, Virginia, is transported to HPM via trucks equipped with pneumatic transferring systems. After off-loading into the lime kiln dust bin, the dust is metered onto the belt by means of a variable rate screw conveyor. The application rate is approximately 2% by weight of the refuse being generated. At the computerized control panel located in the preparation plant, the rate of refuse generation is displayed on a video monitor. As the rate changes, the plant operator has the ability to adjust the screw conveyor.

Since the beginning of the application program, several key factors have been learned regarding the handling of the lime kiln dust. The first factor is that dust behaves both as a solid and as a fluid. When de-aerated, the dust has to be mechanically activated to flow. However, if the material is aerated, it flows as a fluid and, unless contained, will flow directly through the screw conveyor. When off-loading from the delivery truck it is necessary to ensure the positive pressure is minimized in the dust bin.

The second key factor is that the lime kiln dust is most effective when it is in contact with the high moisture filter cake. This cake, with its large surface area and associated chemical flocculant, does not dry easily. The lime kiln dust, if in contact with the cake, helps to dry the material and make it workable. The problem at HPM is that the coarse size fraction rests directly on top of the filter cake and not vise versa. Application of the dust is performed immediately after the refuse leaves the plant. To solve this problem, access to the filter cake was gained by installing belt plows. The first plow is situated prior to the dust application point and is used to open the coarse refuse. Once past the application point, a series of plows roll the coarse refuse back on top of the filter cake, trapping the dust between the coarse and fine material.

Refuse Disposal

As with the problems associated with the application of the kiln dust, the handling and placement of the treated refuse material in the refuse cell required rethinking of past practices. The original method was to dump pile against pile and, on a sporadic basis, push the refuse with a dozer or front-end loader. Prior to kiln dust treatment, the combined refuse material was difficult to handle and would not shed water in a reasonable timeframe (i.e., three to five days). Once the refuse became treated on a somewhat regular basis, the handling problems were addressed. The first attempt to handle treated refuse in the cell was to push the refuse into high piles by a dozer. This method not only pushed up the refuse but also pushed water back into the refuse which had accumulated on the cell floor.

To solve the problem, a rock road was built in the middle of the refuse cell approximately 10 to 15 feet above the cell floor. The treated refuse was then end dumped from the rock road down into the cell. A dozer and operator were assigned full time to push and level the treated refuse. This enabled the dozer to push the refuse down and at the same time allowed the water being shed from the refuse to drain away from the piles.

A second standard operating procedure was also instituted. It was noticed that if the piles were contiguous to each other, the excess water was trapped between the piles. The procedure was modified so that the piles were dumped with a space between adjacent piles. This allowed water to be channelled away from the piles.

Once these handling problems were addressed, the effects of the lime kiln dust became apparent. The dust not only neutralized acidic waters but also, if dozed flat and left untouched for two to three days, the refuse began to stabilize. This stabilization process, driven by the pH and pozzolan nature of the calcium hydroxide, fly ash (with associated silica and alumina), and clays began to firm the combined treated refuse material. Within a total of five days it was possible to drive a fully loaded refuse truck on top of the material with minimal (5 inches) indentation of the refuse material by the truck tires. The work of developing handling procedures and the successful results of stabilization and treatment occurred during a time of above normal precipitation (first half of 1989). Even with the high proportion of rainfall, the treated refuse stabilized to a point of being able to support the weight of the loaded trucks.

As experience was gained, it was determined that multiple end dump sites within the refuse cell

were needed. This would allow the required time for the refuse to stabilize. To this end, three distinct dumping areas were developed. From these sites a sequential dumping pattern was used, giving the refuse time to stabilize. As time has progressed, the three dump sites have become progressively larger and spread out. This has given additional area for dumping and a proportional increase in refuse stabilization time.

## Summary

Since the inception of HPM, the concern for the long term effects on the environment has been one of the factors in the planning process. The refuse cell concept is a "leading edge" technology which, to date, appears to be a very successful method for handling refuse. The addition of the lime kiln dust to the combined refuse has had the anticipated positive effects. Because calcium oxide has a higher degree of reactivity and the lime kiln dust is being mixed more intimately, the neutralization of the refuse is occurring with a higher degree of success. Secondly, and equally important to uninterrupted operations, is the stabilization resulting from the pozzolan nature of the materials. Volumetrically, a refuse cell which was designed to contain 20 million yards of material can now hold 20 million yards of refuse. Without the kiln dust, the rock bridges use over 50% of the volume. With the lime kiln dust the life of the refuse cell has been substantially increased.

Although proprietary in nature, several generalized comments can be made regarding the economic impact of using lime kiln dust in the refuse cells. First, the cost of neutralization by the combination of hydroseeded limestone rock dust and bacterial inhibitor approximates the cost of using the lime kiln dust. Secondly, because rock bridges are no longer required to support the refuse trucks, the cost of rock transport and placement is eliminated. And thirdly, with the extended life of the cells, fewer cells are required, saving permitting, construction, and monitoring costs.

## Literature Cited


Chou, L. (Ed.), 1987. Lime Stabilization, Reactions, Properties, Design, and Construction. State of the Art Report 5, Transportation Research Board, National Research Council, Washington, D.C.

Diamond, S., White, J.L., and Dolch, W.L., 1964. Transformation of Clay Minerals by Calcium Hydroxide Attack, pp. 359-379. Proc., 12th National Conference on Clays and Clay Minerals, Pergamon Press, New York, N.Y..

Eades, J.L., 1962. Reactions of $Ca(OH)_2$ with Clay Minerals in Soil Stabilization. PhD thesis, Geology Department, University of Illinois, Urbana.

Little, D.N., 1987. Fundamentals of the Stabilization of Soil with Lime. National Lime Association.

Stocker, P.T., 1972. Diffusion and Diffuse Cementation in Lime and Cement Stabilized Clayey Soils. Special Report 8, Australian Road Research Board, Victoria, Australia.

Sturm Environmental Services, Inc., June, 1985. Refuse Disposal Criterion for High Power Energy.

---- Field and Laboratory Studies: "Refuse Disposal Program, High Power Mountain Corp.", December, 1987.

This page intentionally left blank.

# Some Aspects of SSPE/PSM Modelling for Quantitative Assessment of Disturbed Hydrological Systems

Thomas Rymer, Alfred Stiller, Walter Hart, John Renton[2]

## Abstract

In previous research involving the SSPE/PSM technique it was demonstrated that this procedure could be used to develop longevity estimates for refuse sites with respect to acid effluent. It could also be used to determine the time frame for maximum acid effluent and to evaluate the potential impact upon the localized watershed. The technique evaluates two field parameters: ALPHA, the rate of acid generation, and BETA, the rate constant of acid discharge. Since this initial work was performed and presented, extensive research has been conducted that has allowed the evaluation of the factors that control ALPHA and BETA. ALPHA is a micro-diffusion phenomenon controlled by the pyrite grain size as well as pore size distribution. BETA, although a product of innumerable parameters, has been shown to be a 'steady' field parameter that is a resultant of a very large number of microsystems. Additionally, the modeling technique has been modified to handle active refuse sites. The technique was able to estimate within 1% error the amount of new material placed on an active refuse site in a 1 1/2 year period using discharge chemistry, the amount of refuse material on the site at the time that field measurements began to be taken, and the weekly average amount of new material placed on the pile. The ability of the SSPE/PSM technique to make such reliable estimates strongly supports the theoretical concepts used and applied. Presently, the technique is being refined to accurately evaluate the PHC and CHIA requirements of the permit review procedure and to deal with linear properties that BETA may possess in certain situations.

ADDITIONAL KEY WORDS: shrinking core model, mean-lifetime

[1] Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23 -26, 1990.

[2] Thomas Rymer, Alfred Stiller, and Walter Hart are Sr. Research Associate, Associate Professor, and Graduate Student, respectively, in the Department of Chemical Engineering at West Virginia University, Morgantown, WV 26506. John Renton is a Professor in the Department of Geology and Geography at West Virginia University.

## Introduction

In 1988 an initial evaluation on a mathematical model that allowed for an acid production-discharge scenario to be simulated for refuse piles and mountain top removal operations was completed . This model is called SSPE/PSM (Simultaneous Species Production-Elimination using Probability Simulation Modelling) (Rymer et al. 1988). This model generates two field variables known as ALPHA (the first order reaction rate at which pyrite oxidizes) and BETA (the first order physical rate constant at which mobile ionic species are leached from a site). The model utilizes one form of the "Bateman Equation" as its scientific and mathematical foundation. This equation is widely used in areas such as radioactivity (intermediate daughter isotope) and chemical kinetics (ammonia cycle). Probability simulation employs the use of data distributions rather than discrete data, and dispenses with deterministic formulas due to the numerical randomness of field (input) data. The complex assortment of chemical and physical processes by which acid is produced and discharged can then be reduced to the two field parameters, ALPHA and BETA.

Two types of graphs are output from SSPE/PSM. The first type of graph contains three species-time curves as shown in Figure 1. The curves represent the mass of pyrite remaining unreacted, the mass of sulfate ion that has been discharged to the environment, and the mass of sulfate remaining in the site (undischarged). To illustrate a scientific parallel a graph of the ammonia cycle is shown in Figure 2 (Sawyer 1978), showing the mass of ammonia unoxidized, the amount of nitrate that has been produced over time, and the amount of nitrite that has not, as yet, been oxidized to nitrate. The second type of graph output from the modeling is called a "longevity curve" as shown in Figure 3. This shows the concentration of sulfate ion that can be expected over time. This curve is actually the most probable mean value of the sulfate ion concentration at the most frequent discharge flow value.

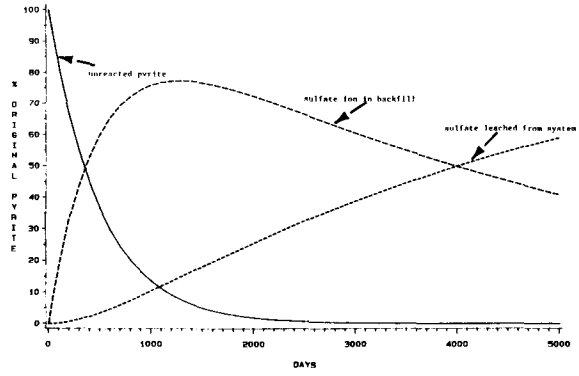



**Figure 1.**

**Classical SSPE system for pyrite oxidation. Curves represented are % of original pyrite remaining (solid line), % of original pyrite that exists in the backfill as sulfate ion (bold dashed line), and the % of original pyrite that has discharged into the environment as sulfate ion.**

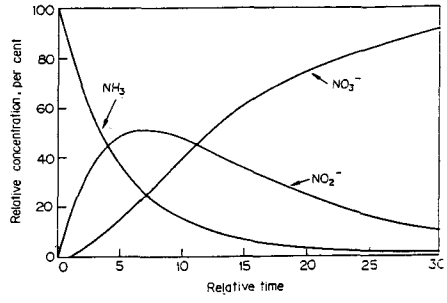



**Figure 2.**

**The Nitrogen Cycle as a classical SSPE system. (after Sawyer, 1876)**

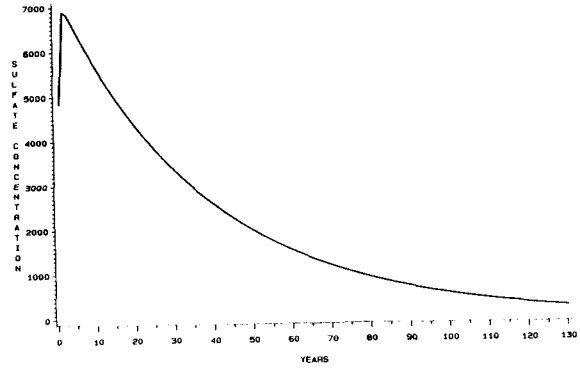



**Figure 3.**

**A longevity curve for a typical refuse site expressed as the concentration of sulfate ion. The curve represents the average of a "window" of probable concentration values.**

The preponderance of evidence seems to support the model. First, field evidence shows that a peak sulfate concentration can occur during the first 1to12 years. Longevity curves exhibit the same time window for peak sulfate concentration. Second, it is commonly accepted that the oxidation process is rapid and the discharge process (or flushing of oxidation products from a site) is slow (Rymer et al. 1988; Skowsen 1989; Ziemkewicz 1989; Hall 1989).. SSPE/PSM shows ALPHA to be greater than BETA by a magnitude of ten (Rymer et al. 1988). Third, some of the longevity curves show AMD to be discharging over a prolonged period of time, often in excess of 100 years. Seventy to eighty year old discharges are not uncommon in the field (Hall 1989). Fourth, the model relies on data distributions of total sulfur to evaluate the amount of pyrite available at the commencement of site operations. Ample evidence exists to show that AMD occurs in areas where the total sulfur content of overburden materials is low (Skowsen 1989). Drawing values from distributions rather than a reliance on discrete analytical data seems to be a logical choice in view of the strong evidence that a high probability exists for sampling error and total sulfur underestimation (Rymer et al. 1989). This choice, however, becomes a moot point in SSPE/PSM modeling as will be demonstrated later. Finally, the model was revised and utilized in an acid rain-limestone treatment project in the headwaters of Shavers Fork (Cheat River) in West Virginia . The model correctly projected the calcium ion concentration and pH over time of the tributary under study (Ivahnenko 1988).

Since the introduction of the original model, research has continued that addresses five vital questions:

(1) Are the "Bateman Series Equations" really applicable in a situation where ALPHA is a chemical phenomenon and BETA is a physical phenomenon?

(2) Does all of the pyritic material react ? If not, how much does react and what factors affect the amount reacted?

(3) How sensitive is a BETA value to the original amount of pyrite in the site material ?

(4) Since the longevity curves sometimes show a period of AMD discharge, exceeding human life expectancy, how can the model be proven viable in present time? Can enough experimental evidence be amassed to support the model without having to wait to the latter part of the 21st century ?

(5) What factors control ALPHA and BETA? Which of these factors are significant and which can aid in the quantification of BETA ?

## Results and Conclusions

The oxidation of pyrite has been determined to be an intrinsic "pseudo" first order (behaves as such for quantitative purposes) by a variety of research endeavors. Predominant is the work of Stumm and Singer (1970) in which the rate determining step for pyrite oxidation was determined on a laboratory scale. Later work of Renton et al. (1987) showed the first order behavior of pyrite oxidation to exist not only in the laboratory but also on a small field scale (350 pound fixed-bed plug-flow reactor), as well as on a larger field scale (350 ton test refuse piles).

The determination of the first order behavior of the physical leaching process, prior to this research, had never been discussed or attempted. At the onset of the development of the SSPE/PSM modeling research, three equations (including Bateman's) were studied.

$$dS/dt = aS - bS \qquad (1)$$

is the differential form of Bateman's Equation which demands that BOTH the oxidation rates and the leach rates be FIRST ORDER (Wehr 1972).

$$dS/dt = aS - b \qquad (2)$$

implies that the oxidation process is FIRST ORDER and the leach rate is a LINEAR CONSTANT.

$$dS/dt = a - b \qquad (3)$$

inplies that BOTH the oxidation process and the leach process are LINEAR CONSTANTS.

The resulting curves for each equation can be seen in Figure 4.

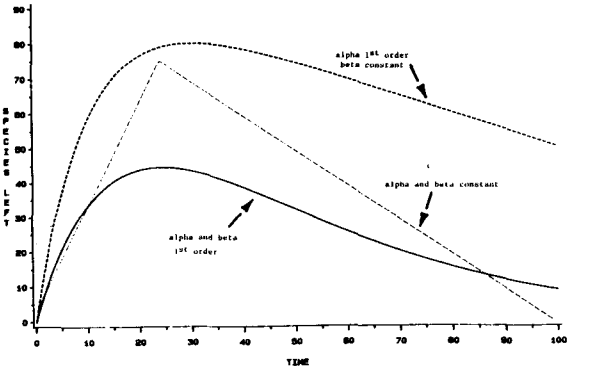



**Figure 4.**

**SSPE curves for the 3 types of systems corresponding to equations 1 through 3. Curve "A" represents the case where both ALPHA and BETA are first-order, Curve "B" represents the case where ALPHA is first-order and BETA is a linear constant, and Curve "C" represents the case where both ALPHA and BETA are linear constants.**

Equation 3 from the above selection was immediately eliminated due to the strong evidence that the oxidation process can be quantified in first order kinetic terms (Stumm and Singer 1970; Renton et al. 1987). Left with Equations 1 and 2, the only remaining factor to be determined was whether the leach rate exhibits similar 1st order kinetic behavior, despite being a purely physical process. If the leaching process exhibits first order behavior, then it will also emulate a "Poisson process" (Devore 1982). The evidence for the leaching process being a Poisson process exists:

(1) It has the properties common to Poisson processes. Other Poisson process examples include the monitoring of a computer system over time with breakdowns constituting the events of interest, recording the number of accidents in an industrial facility over time, answering calls at a telephone switchboard, and observing the number of cosmic-ray showers from a particulate observatory over time (Devore 1982). Flow discharges are functions of independent Poisson events, such as rainfall.

(2) The quantification of the leaching process involves both a chemical and physical parameter, sulfate concentration and flow rate, respectively. Thus, field data could serve as a basis to test whether this process , truly, exhibits first order rate characteristics. The BETA values obtained in the SSPE/PSM simulations have a magnitude of $5 \times 10^{-4}$ /day. It was decided to compare the distribution functions of the product of sulfate and flow present at a minesite (mine 4, Table 1) over a defined period of time with the disintegrations of a radioisotope having a known half-life of 1400 days over the same time period. Sodium-22 was the radioisotope chosen due to the fact that it has a decay rate constant nearly equal to the BETA values obtained in this research and the data was readily available from Lawrence-Livermore Labs (Berkeley, CA). Comparisons can be drawn by examination of Figure 5. The Poisson nature of the flow-sulfate concentration obtained from actual field data exhibits properties similar to the radioactive disintegrations. Therefore, in the case of a backfill discharge the assumption of first-order emulation is valid and Equation (1) can be applied. It is theorized, however, that the kinetic emulation of the physical discharge process is, in all probability, some combination of Equation (1) and Equation (2) with the latter equation coming into play at very high flows. In fact, the area of the SSPE/PSM technique in need of research is this overlapping boundary between first order rates and linear rates.

**Table 1.**

The BETA values obtained from SSPE/PSM modeling technique on selected refuse and mountain top removal sites.

| MINE SITE | BETA x $10^4$/day |
|---|---|
| 1 | 5 |
| 2 | 5 |
| 3 | 2 |
| 4 | 5 |
| 5 | 5 |
| 6 | 5 |
| 7 | 2 |
| 8 | 5 |

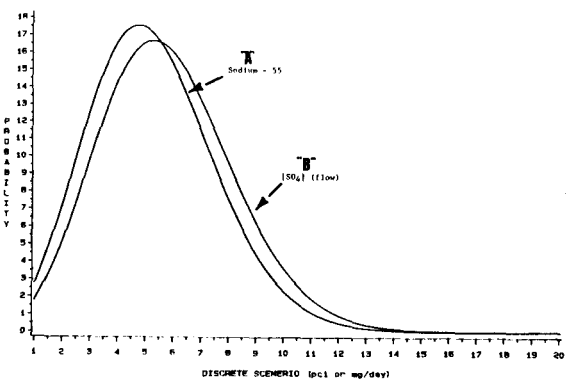



**Figure 5.**

**Comparison of two systems exhibiting Poisson Properties. Curve "A" represents the disintegrations per day of sodium-22 which has a half-life of approximately 1400 days. Curve "B" represents the product of sulfate ion concentration and flow (milligrams of sulfate per day). Sodium-22 data courtesy of Lawrence-Livermore Labs, Berkeley, California. The discharge data courtesy of WV DOE and US Dept of the Interior OSMRE.**

The amount of pyritic material that actually undergoes oxidation has been a subject of much recent speculation. To resolve whether an entire pyrite grain oxidizes, a mathematical model (Batarseh et al. 1988). was developed and tested experimentally (Batarseh 1987). The model is based on as the "shrinking core model" with a moving boundary and quantifies the reaction zone of a pyrite grain.

Classical rate kinetics predict that an average of 63% of all pyritic material will react (Appendix 1). This was borne out in the work of Baker (1983) and McConihey (1985) in which three untreated , 350 -pound waste rocks were allowed to naturally weather in a "fixed bed-plug flow" reactor. Using a measured total pyritic sulfur value of 3.14%, an average of 35% pyrite remained unreacted after 256 days. This is in excellent agreement with the amount of unreacted pyritic material predicted from reaction kinetics.

**APPENDIX 1**

**The proof that 63% of a reactive substance will remain after the "mean lifetime" in a 1st order kinetic process.**

**the 1st order rate equation states that:**

$$\%S_u = 100\, e^{-at}$$

where: $\%S_u$ = % unreacted pyritic sulfur
$a$ = rate constant (ALPHA)
$t_m$ = time

the mean lifetime $t_m = 1/a$ (Van Nostrands 1976)

then: $\%S_u = 100\, e^{-a(t_m)} = 100\, e^{-a(1/a)}$

and: $\%S_u = 100^{e-1} = 100\,(1/e) = 100(0.63) = 37\%$

then the %S (reacted) = **63%**

Batarseh et al. (1987) showed, however, that the amount of sulfate ion produced in a weathering process relates directly to the pore size distribution and pyrite grain size. He also demonstrated that the average pyrite grain will reduce in size about 20%. Therefore, the amount of unreacted pyrite should be 80%, much higher than predicted by rate kinetic theory. This non-agreement is expected since rate kinetic theory does not take into account such parameters as pyrite grain size and pore size distribution. However, to account for the discrepancy in the Baker data from 1983, which showed 35% of pyrite unreacted, another phenomenon must be taking place -- postulated to be neither a purely chemical nor physical process, but biochemically induced. It is speculated that when the role of iron oxidizing bacteria such as thiobaccillus ferrooxidans is illuminated and compared to and integrated with existing data then this discrepancy will be solved. Such an investigation has commenced. Another potential contributor to this discrepancy is analytical -- the total pyritic sulfur measured by Baker could have been underestimated. This brings the Baker data and the Batarseh data slightly closer but will not account for the gap.

The underestimation of total pyritic sulfur brings up another problem in need of resolution. Since the SSPE/PSM technique involves the calculation of some original amount of pyritic sulfur present in a minesite prior to mining, and this number may be too low, how is the final calculation of "BETA" affected? To answer this, a computer was programmed to analyze a typical flow-sulfate discharge scenario from actual field data. Instead of looking through various rock type sulfur distributions and adjusting for the area and volume of material present to obtain a value for the original amount of pyrite present prior to mining, the computer was programmed to set this original pyrite amount at certain tonnages. A "BETA" value was then calculated for any given size and corresponding sulfur tonnage. Hence, the sensitivity of "BETA" to this original amount could be studied. The results of this response of a calculated "BETA" value to the original amount of pyritic sulfur can be seen in Figure 6. It can be seen from Figure 6 that the amount of pyrite originally present ceases to exert influence on a calculated "BETA" value after a certain point ( a point well above the amounts actually present in typical minesites). This adds credence to the use of distributions in the SSPE/PSM modeling technique. This also opens the door to use this technique on deep mines as well as surface operations other than refuse disposal and mountain top removal sites (for which the model was originally developed). This also lends support to another contention -- that "BETA" is some summation of innumerable dissolution and physical microprocesses and that at some point these microprocesses begin to take on a single macroprocess emulation. Nature has conveniently sorted out these processes into a single quantifiable number -- BETA.

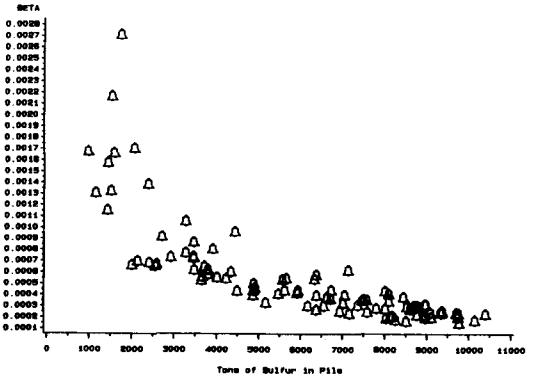



**Figure 6.**

**Shows the sensitivity of a BETA value generated via SSPE/PSM to the mass of pyrite originally present in a system. For a given discharge scenario the BETA value will become constant after a certain point.**

Since the inception of this model, an actual field proof for a coal mining scenario has been sought. The model was successfully applied to the addition of limestone into a pristine mountain stream, but AMD is not quite so pure. The unfortunate fact that verification of the model for large systems requires moving forward to the latter part of the 21st century has made direct proof impossible at the present time. However, support did emerge from research which appeared to be, at first, a situation where the model did not appear to work. The minesites studied in the development of the SSPE/PSM modeling technique were coded by the field hydrogeologist. Since refuse piles were being used, it was postulated that all of the "BETA" values would be of

comparable magnitude. Indeed, as seen in Table 1, they are. However, one minesite listed (known as Mine 4), had produced an unusually high "BETA" value and a rapid longevity curve as can be seen in Figure 7. The geometry of this site was comparable to all of the other sites and the size of this site was somewhere in the middle. If this discrepancy could not be explained then a definite fallacy exists in the critical premises used in construction of the model. It became paramount to discover what phenomenon was responsible for such a high "BETA" value. The answer was provided by the field hydrogeologist in that this site was the only one under study that was still an "active" site. The computer made calculations implicitly assuming that the sulfate coming out through the discharge was based on a finite amount of material being present, not an increasing, fluctuating amount of material. It should be noted that although the BETA value will not be significantly affected by the addition of more material beyond a certain value, the sulfate concentrations in the discharge certainly will.

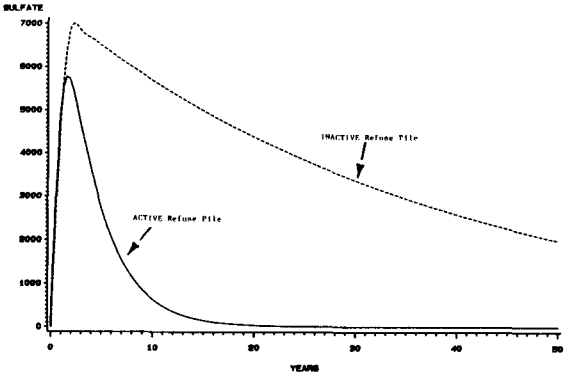



**Figure 7.**

**Shows the curve (solid line) of the "active" refuse site compared to the typical "inactive" refuse site. The BETA value generated was "inflated" due to sulfate ion in the discharge originating in new material being added.**

The SSPE/PSM technique was modified to handle new material being added. Using the "BETA" value obtained for the other minesites that were examined ($5 \times 10^{-4}$/day) and a modified simulation, the computer was reprogrammed to calculate how much new material had been added during the course of the study. The company provided the amount of new material that had actually been added and the two results were compared (17.1 million tons actually added and 17.3 million tons calculated to have been added). This is statistically considered to be excellent agreement. This is also strong verifiable field evidence that the SSPE/PSM model is valid.

An example of the SSPE/PSM curve for an active refuse pile showing the "remaining unreacted pyrite," "sulfate in system," and "sulfate discharged to the environment" curves can be seen in Figure 8. When the refuse pile becomes inactive the curve representing the "sulfate remaining in the system" will exhibit a first order decay mode with a half-life equal to

$$t_{1/2} = \ln(2) / \text{BETA} \quad (4)$$

and a mean-lifetime $t_m = 1 / \text{BETA}$ (5)

source: Van Nostrands Scientific Encyclopedia--5th Ed.

This case can be seen in Figure 9.

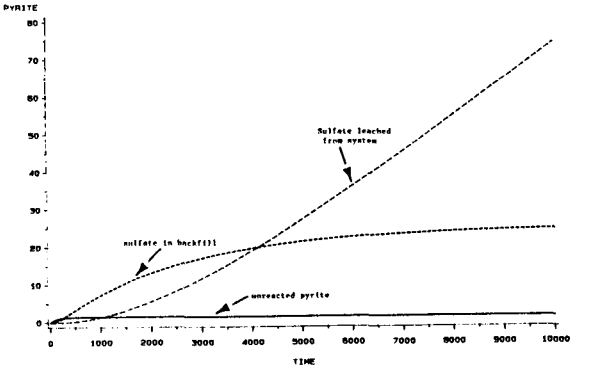



**Figure 8.**
**The SSPE/PSM system curves for an active refuse pile. The vertical axis labelled "pyrite" is expressed as the percentage of the total amount of pyrite that has been placed on the pile. The time axis is in days. The amount of new material added per day was 500 tons per day.**

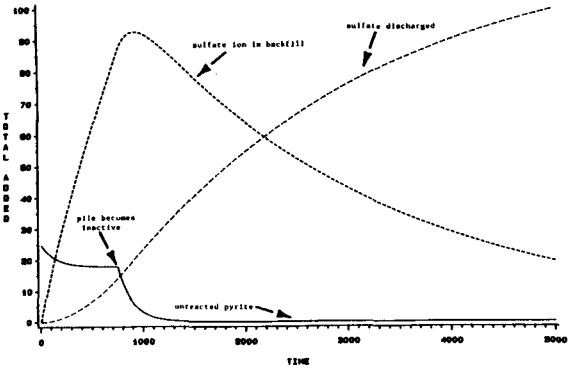



**Figure 9.**

**The SSPE/PSM system curves for an active refuse pile up to 800 days then inactive after 800 days. The vertical axis labelled "pyrite" is expressed as the percentage of the total amount of pyrite that has been placed on the pile. The time axis is in days. The amount of new material added per day was 500 tons per day.**

## Conclusions

Several conclusions have been drawn from the modeling research:

(1) The chemical and physical processes for pyrite oxidation exhibit first order kinetic behavior;

(2) All of the pyrite present in a minesite does not react and this amount is significantly less than 100%;

(3) The incalculable number of microprocesses responsible for the oxidation product dissolution and physical "flushing" from a minesite are accounted in a single, quantifiable parameter (BETA);

(4) The "BETA" values for refuse piles and mountain top removal operations consistently show a magnitude of $10^{-4}$/day;

(5) This "BETA" value may not hold for other mine types, but since "BETA" becomes less and less mass dependent at large volumes of material, unique scenarios involving these other mine types can be simulated.

## Acknowledgements

The authors would like to thank Mr. Roger Hall of the West Virginia Dept. of Energy for his support of this work and Dr. Paul Ziemkewicz of the National Mine Land Reclamation Center for his support and review of the manuscript. Mr. Paul Sutter, now Development Engineer for Apple Computer Corp., for his free and excellent systems programming on the VAXStation 3500 and to the Pittsburgh SuperComputering Center, The National Science Foundation and Westinghouse Corp. for time and use of the CRAY XP/2 Supercomputer.

## Literature Cited

Rymer, T.E., Renton, J.J., Stiller, A.H., 1988, A Computer Simulation Probability Model for Geochemical Parameters Associated with Coal Mining Operations. in Proc. Mine Drainage and Surface Mine Reclamation Conference, American Society for Surface Mining and Reclamation and the U.S. Dept. of Interior (Bureau of Mines and Office of Surface Mining and Enforcement), April 17-22, 1988, Pittsburgh, PA.

Sawyer, C.N., McCarty, P.L., 1978, Chemistry for Environmental Engineering, McGraw-Hill, New York, 532 p.

Skowsen J. Personal Communication, 1989.

Ziemkecewicz, P., Personal Communication, 1989.

Hall, R., Personel Communication, 1989

Field Personnel from West Virginia Dept. Of Energy, Personal Communication, 1989.

Rymer, T.E., Stiller, A.H., Renton, J.J., 1989, The Effects of Pyrite Grain Clustering on the Measurement of Total Sulfur, in Proc. 9th Annual WV Surface Mine Drainage Task Force Symposium, Morgantown, WV.

Ivahnenko, T., 1988, An Evaluation of Point-Source Limestone Introduction as an Ameliorative Procedure to Reduce Acidity in Two Low Buffer Capacity Streams, M.S. Thesis West Virginia University, Dept. of Geology.

Rymer. T.E., Renton, J.J., and Stiller, A.H., 1988, Modeling of Geochemical Parameters of Natural Systsms using SSPE/PSM, Northeast Fish and Wildlife Conference (Acid Precipitation Session), White Sulfur Springs, WV, March 27 - 30, 1988.

Stumm and Singer, 1970, Acid Mine Drainage--The Rate Determining Step, Science, (167).

Renton, J.J., Rymer, T.E., Stiller, A.H., 1988, A Laboratory Procedure to Evaluate the Acid Producing Potential of Coal Associated Rocks, Mining Science and Technology, Elsevier Science Publications, July 1988, pp. 227-235.

Wehr, M.R., Richards, J.A., 1967, Physics of the Atom, Addison-Wesley, Reading, MA, 481p.

Batarseh, K., Swaney, G., Stiller, A., 1989, A Mathematical Model for Heterogeneous Reactions with a Moving Boundary, Journal of the American Institute of Chemical Engineers, vol. 35, no. 4, April, 1989, p. 625.

Batarseh, K., 1987, The Effect of Physical Properties of Toxic Mine Waste on Acid Mine Drainage: A Mathematical Model, Thesis, West Virginia University.

Devore, J.L., 1982, Probability and Statistics for Engineering ant the Sciences, Brooks/Cole Publishing, 640 p.

Baker, B.,, 1983, The Evaluation of Unique Acid Mine Drainage Abatement Techniques, M.S. Thesis, Dept. of Chemical Engineering, West Virginia University.

McConaghy, B., 1985, Analysis of Novel Acid Mine Drainage Experiments, M.S. Thesis, Dept. of Chemical Engineering, West Virginia University.

Van Nostrands Scientific Encyclopedia, 5th Ed., 1976, Van Nostrand Reinholdt, New York, 1519 p.

This page intentionally left blank.

# COAL REFUSE RECLAMATION: EFFECTS OF TOPSOIL THICKNESS ON THE GROWTH AND MAINTENANCE OF VEGETATION[1]

by

Ronald L. Kolbash, PHD[2]
John R. Freeman
Thomas L. Romanoski
John W. Sturm

Abstract. The four feet topsoil requirement for coal refuse piles may be reduced if revegetation requirements can be met with less soil cover. A 1.2 acre demonstration project was initiated on a large coal refuse pile in north central West Virginia to evaluate the effects of topsoil thickness on the growth and maintenance of vegetation. Five 80 x 100 - foot plots were covered with four, three, two, one and zero feet of topsoil from an adjacent borrow area. Refuse and soil samples were collected at various depths before construction and in the third and fourth years of the project. Refuse samples were analyzed for acid-base account and soil samples were analyzed for cation exchange capacity and base saturation. All samples were analyzed for plant nutrients, moisture holding capacity and texture. The plots were limed, fertilized and seeded with a grass-legume mixture. Vegetation samples were collected in the third and fourth years. Four transect lines were established across each plot and two 2 x 2 - foot quadrats were placed on each transect. Vegetation within each quadrat was clipped at ground level, air dried and dry weights were determined. Repeated measure analysis of variance to determine significant differences among topsoil treatments showed no significant difference between the four, three and two feet treatments in Year Three and no significant difference between the four, three, two and one feet treatments in Year Four. Soil profiles were described and sampled in Year Four to evaluate root penetration. Roots penetrated to the soil-refuse interface in all soil thicknesses.

Additional Key Words: Coal, coal refuse, topsoil, revegetation, minesoil

[1]Presented at the 1990 Mining and Reclamation Conference, Charleston, West Virginia, April 23-26, 1990.

[2]Ronald L. Kolbash is Administrative Assistant-Governmental Affairs and Thomas L. Romanoski is Environmental Engineer-Planning and Engineering, American Electric Power, Fuel Supply Department, Lancaster, OH 43130. John W. Sturm is President and John R. Freeman is Vice President of Sturm Environmental Services, Inc., Bridgeport, WV 26330

This page intentionally left blank.

# MINE DRAINAGE BIOREMEDIATION - THE EVOLUTION OF THE TECHNOLOGY FROM MICROBES TO BIO-CARB

Jo Davison, President & Susan Jones, Vice President
Lambda Bioremediation Systems, Inc.
2840 Fisher Rd. Complex "E", Columbus, OH 43204

Abstract. Mine drainage, normally acidic by nature, is the usually unavoidable discharge of metallic waste products n water from mine disturbed lands. The indigenous flora and fauna of the aquatic and terrestrial systems receiving this discharge are more times than not adversely affected. The advent of the phenomena of acid precipitation(1) in the last 20 years has further complicated the problem, and even stricter N.P.D.E.S. effluent standards further restrict conventional treatment methods. Bioremediation, the use of natural, harmless, environmental microbes to both achieve and maintain enforced standards, has proven to be a viable alternative in those areas where it is presently being used. It is an old technology that nature ha used successfully for untold millions of years and one that I observed over 20 years ago in a mine site in West Virginia. Today, that technology can be successfully implemented at any mine site to clean the water and soil and/or extract more metals from tailings ponds. It is effective and appears to be a permanent solution to mine drainage and the affects of acid precipitation environmental alterations.

## Introduction

The observation of a small acid mine slip, separated by rocks and forming two small pools of similar sizes near a coal mine in Fayette County, West Virginia was the beginning of this bioremediation technology. One pool had only a few common mine bacteria and continued to impart the reddish-orange color to the rocks and debris associated with mine waste effluent. The other did for a short distance, then flowed out colorless and clarified. Tests showed the iron in both to be about 6 ppm. Where the two discharged, the colorful one was 5.7 and the clear one showed on 1.5 ppm iron. Why? The answer lies in the mixatrophic population that naturally occurred in the clarified water, but was lacking in its counterpart.

To apply this to the treatment of large mine sites requires more than just "dumping bugs" into treatment ponds. Which microbes do you use? How do you know if they'll work on a specific site? Will they drop out the metals? Will they raise the pH? Will they create ecological chaos in the receiving waters? How many gallons of water a minute will they treat? How many ponds will it take and how deep should they be? Do they build up sludge? Will it work forever? How fast do they work? Will I meet compliance by lunch tomorrow? How soon can you deliver? How much does it cost and what guarantees do I have?

Those are all valid questions - those and more that we ask ourselves every time we install the process. Each answer and each site we do leads to knowledge and understanding of this new technology that has led Lambda from chance microbes in a small pool to the Bio-Carb technology.

## The Lambda Process

The microbes we employ are those that will chelate or bond to metals, microbes that produce oxygen, decomposers and nitrogen-fixers, reducing organisms in the soil and any others that are needed to maintain an ecological balance in the system (See Fig. 1). We know from hundreds of tests what the microbes can do. To find out if they work on a specific site, we have to analyze that site to see what it has that we know we need and what it is lacking at least in adequate numbers to do the job. We can foresee a generic mix that can be modified slightly from site to site. Each site we do brings use closer to that. They can raise the pH to the historic norm of the area, but effluent pH standards 6-9 are unrealistic at best, unless you are discharging into a trout farm. Average rain water pH should be 5.5 (that has been lowered considerably by acid precipitation in the last 20 years(1), but that is another paper entirely), and the average of Appalachian water runs considerably less than that. Therefore, soda ash or mild caustic as an effluent bed prior to discharge is often in order. They are harmless organisms indigenous to the area, or we do not use them.

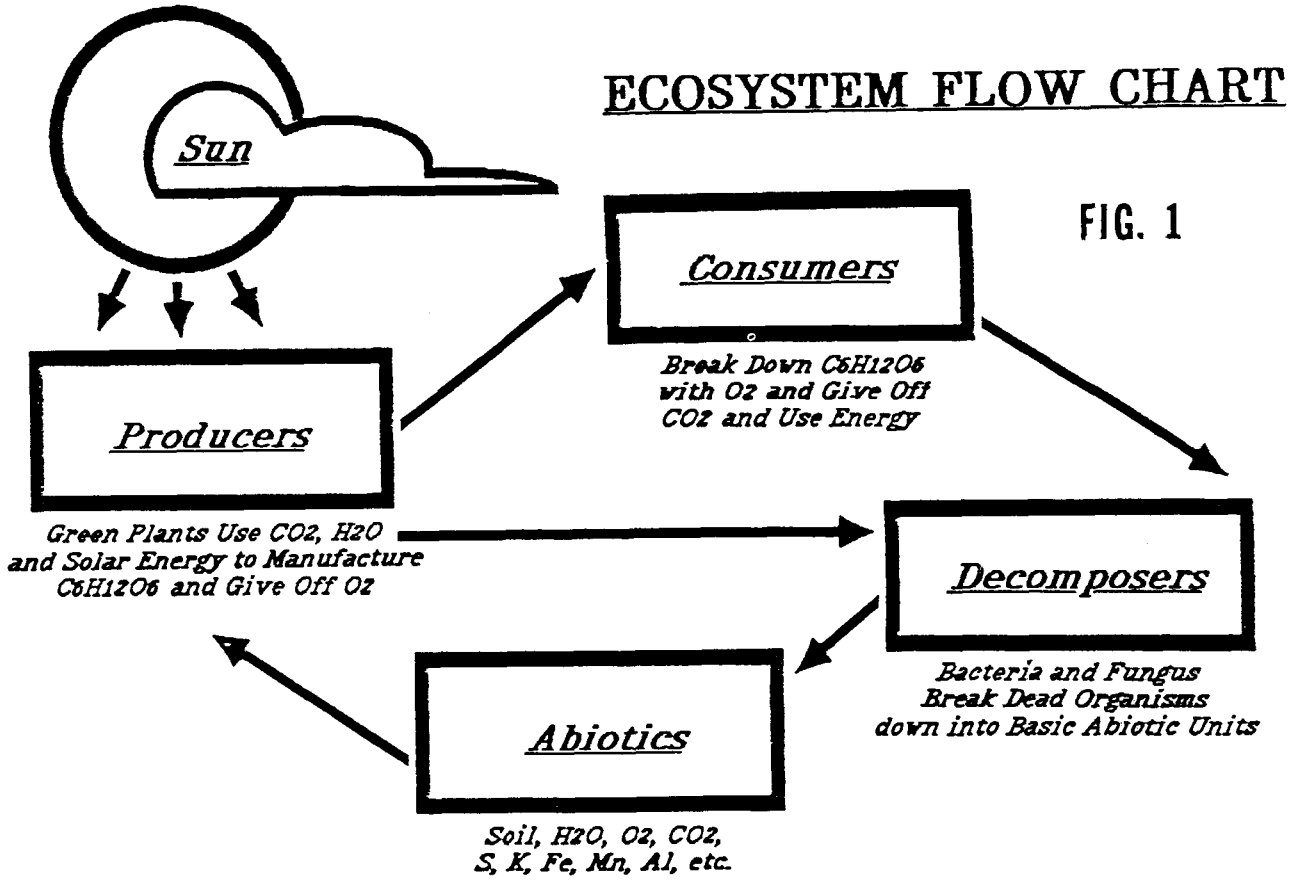



FIG. 1

## *Matter Cycles and Energy Flows through the System*

The flow rates are of concern, but can be handled by a properly engineered site and by imbedding the microbes in a substance that will stay on the bottom and not flow out with the water. We feel we need a retention time in the ponds of at least 24 hours to give the microbes time to chelate, oxidize and deposit the metals into the top soil at the bottom of the ponds. This is in a mature system and it takes an inoculated system 1-12 months to mature. Newly constructed ponds leak both water and microbes and an adequate population can't become established as long as they are leaking.

The major evolution in the technology has come in the area of how to most efficiently "imbed" the microbes. The use of the old IMPPS technology required a tanker truck for delivery and was expensive. The material we used kept the microbes alive and active and provided food and moisture and we were transporting a lot of water to keep them moist during the trip.

We are now using diatomaceous earth and activated charcoal. We can add a liquid substance to the growth mediums that suspend the microbes around the diatom skeletons or charcoal particles in a "dehydrated" state that will rehydrate with a 90-95% recovery ratio. Their effectiveness is seemingly unimpared and 200 lbs. of that material ships easier, is easier to install and less messy than liquid medium.

We have found that 3 ponds of equal size, which is determined by how much water we are treating, at least 5-6 ft. in depth are needed, with at least 15" of fill dirt or top soil are best. These should be divided by spillways with straw and sandstone rip-rap for deposition purposes, since limestone coats and becomes impervious too quickly. They should empty into a fourth round solar pond about 2 feet in depth and then discharge thru soda ash or mild caustic to achieve pH requirements. Only one inoculation is required, unless there are major changes in the system. Once in compliance, it should hold compliance. And no, they do not build up "sludge". The sludge is the result of caustics and metals in combination. Remove the caustics and you remove the sludge problem. On the soda ash at the end, you remove the metals and you remove the sludge. We have seen no sludge problems on our sites and have, in fact, used a "sludge pit" as a fourth or "polishing pond" to the benefit of the water and the sludge. The critters tend to break the sludge there into soil.

# Negative Feedback Flow Chart for Natural Population Control

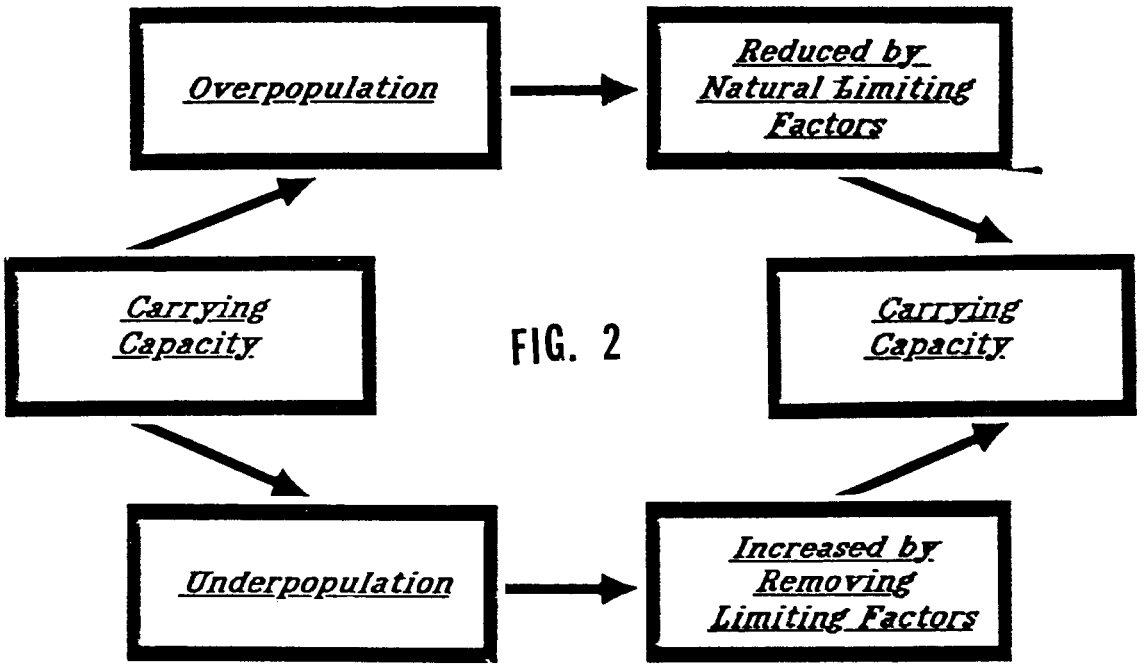



FIG. 2

Compliance comes with maturity. The bigger the system, the longer it will take. The microbes reproduce and populate very quickly, but it does take a reasonable period of time. Will they outgrow the system and develop a blob large enough to eat Charleston? No, they will not. They are controlled by a natural cycle called the "Negative Feedback System" (Fig. 2) and won't grow beyond their carrying capacity. Its nature's birth control system and highly effective. They don't die if the system dries up either. They have slime coats that make them efficient chelators, help them adapt to environmental stress and encapsulate them if the moisture is removed from their environment.

It takes 6-8 weeks to do a site study and identify what a site has and needs and another 3-4 months to grow enough for a site, imbed them all and get them ready to ship. The cost is affordable and will vary from site to site. When compared to treatment costs over 1-2-5-10 years, it will save mine owners money. As far as guarantees are concerned, we won't take a job we can't do.

## Site Studies

The technology has been successfully used at 3 mine sites over the last 3 years and 2 other non-mine sites where metal pollution was a problem. Two of these sites, The FT. HILL MINE in Somerset Co., PA, and the ROBINHOOD MINE in Twighlight, WV, have been reported at previous conferences (2)(3) and a summary of the data is presented in Graphs 1 and 2.

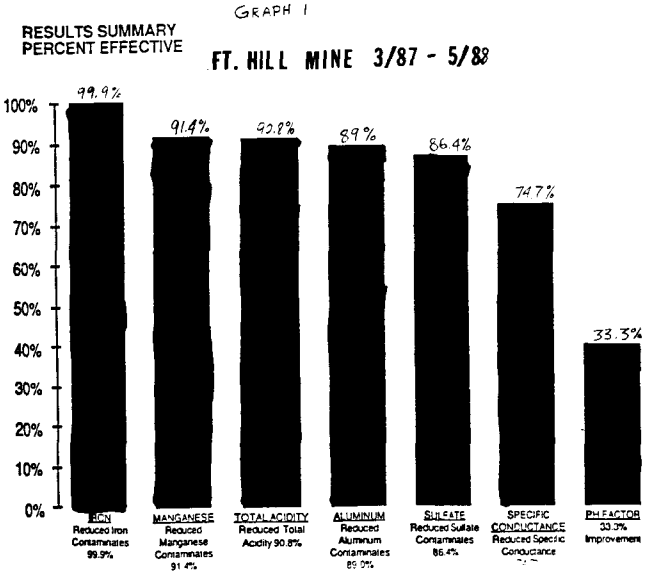
Graph 3 refers to the BOILER MAKER'S Pond and the HICKORY CREEK Pond data is show in Graph 4. These last 2 are stationary ponds. The Graph 3 pond is fed from a heavy metal containing aquifer from a pipe that flows up to 25 gal. per minute for 2-3 hours per day. Graph 4 is a pond that receives primarily surface run-off and some enhancement from a groundwater pipe with a much lower metals concentration. Both were built for decorative reasons.

The process has only recently been installed at the LECKIE SMOKELESS COAL MINE into one of their treatment areas. Inadequate data was available at the writing of this paper to include, but from personal correspondence(4), compliance is anticipated and treatment costs have already been reduced by 50%. This is an interesting site, because it had been treated with anhydrous ammonia for 2 years before the BIO-CARB process was tried. The complexing of the metals by the ammonia makes them inherently less susceptible to bioremediation than does treatment with other caustic forms. These problems will be addressed in a future paper.

## Results of Site Studies

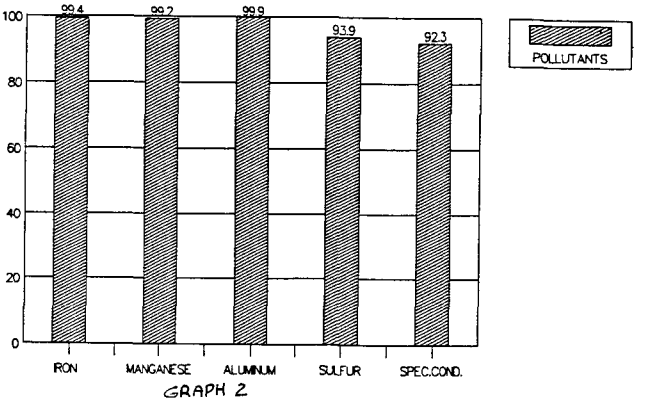
We were able to achieve a 99% reduction in iron at the Robin Hood site, and a 99% reduction in iron at the other sites.

GRAPH 1




PERCENT OF POLLUTANT REMOVAL

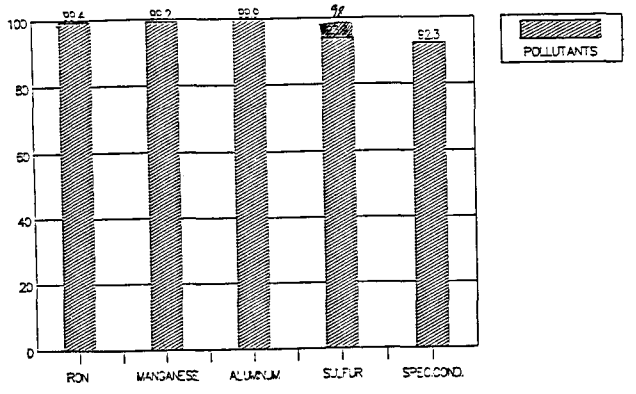
ROBIN HOOD MINE 11/88-12/89




GRAPH 2

PERCENT OF POLLUTANT REMOVAL

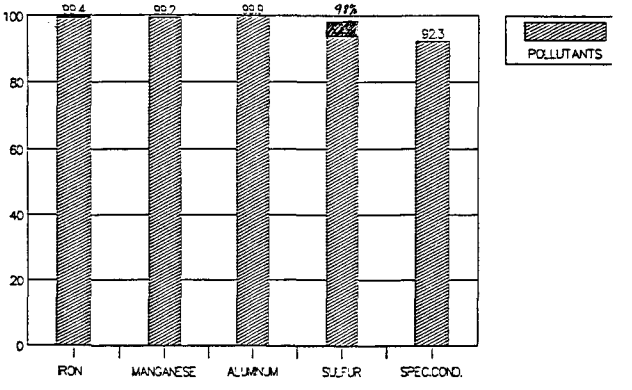
BOILERMAKER'S POND 5/88-5/89




GRAPH 3

PERCENT OF POLLUTANT REMOVAL

HICKORY CREEK POND 8/89-12/89




GRAPH 4

The manganese discharge was reduced by 99% at the Robin Hood site and 99% at the Ft. Hill site. 99% reduction was achieved at the stationary sites. Sulfate was reduced 94% at the Peabody site and 94% at the Ft. Hill site. 98% reduction was achieved at the stationary sites. The pH was improved 33% at the Ft. Hill site and 61% at the mine sites. Compliance was met and held at the mine sites. No N.P.E.S. compliance requirements were in force at the stationary ponds.

All of these results were achieved quickly for the scope of the site (one week for the stationary ponds, 36 days at Ft. Hill and 8 months at Robin Hood). Each one was inherently different and each responded well to bioremediation. Information on individual sites is available upon request from Lambda.

## Summary

The Lambda technology has evolved from a random collection of microbes in a pool to a viable new BIO-CARB application. It seeks only to successfully emulate and enhance the natural microbial cleaning processes nature has employed for countless eons to insure us of having viable aquatic, marine and terrestrial ecosystems and their surrounding ecotones. Even though we are fortunate enough to reside on a planet that is 75% water, the supply of clean, fresh water is not finite, unless we assist as best we can to clean up the messes we make in supplying our other needs. Caustics, surfactants and flocculants offer their own solutions and create problems in their solving of problems. They also require continual application, often in

greater and greater amounts if they are to be continually effective. In evaluating treatment technologies, cost and effectiveness, practicality and ease of use are all important considerations. If bioremediation can offer equal effectiveness, a single application (being dumped in from a 20 gallon drum), and permanent remediation, it will surely fulfill its bright promise.

One problem is the inability to leave a system alone once it is working properly. There is a desire in many cases to make it a "little better". Tampering with the ponds, in any way, whether it be by dumping materials other than what they were designed for, changing the pond sizes, changing the water levels, etc. can disrupt the system. It will eventually rebalance itself, given time, but it will not work as effectively as it did before, until the new balance is achieved. If what you add kills the key microbes or starves the system for balanced oxygen-carbon dioxide, it could kill it. Then you have to start over.

It is a hard lesson to learn, but the old adage still applies - "If it ain't broke, don't fix it". Left to her own devices, Mother Nature will continue to provide us with clean water, fresh air and arable lands. When we "fix" things by changing the existing systems, then she needs a helping hand. Bioremediation is that helping hand and one that won't give the earth terminal indigestion.

## Literature Cited


(1) Clark, Wm. C, "Managing Planet Earth"' Scientific Am. Special Issue; Sept. 1989.

(2) Davison, Jo, "A Natural Solution to Ocidic Pollution"; Mine Drainage Task Force Symposium; April, 1987.

(3) Davison, Jo & Crawford, Jim, "The Application of the Theta IMPPS Technology for Acid Drainage Treatment", Mine Drainage Task Force Symposium; April, 1989.

(3) Wells, Brent; Theta IMPPS Application At Robin Hood Mine, Personal Communication; December, 1989.

(4) Hilton, Tiff; Bio-Carb Application at Leckie Smokeless Mine, Personal Communication; December, 1989.

This page intentionally left blank.

# Evaluating Reclamation Success: Ecological Considerations

This page intentionally left blank.

Minesoil Morphology and Genesis. J. C. Sencindiver, West Virginia University; D. J. Dollhopf, Montana State University; and W. L. Daniels, VPI and State University.

Minesoil properties vary because of differences in geology and soils, topography, climate, vegetation, and mining and reclamation methods. A, B, and C horizons of prime farmland soils are usually removed and saved during the mining process and then replaced during reclamation. In areas such as Appalachia, soils may be too thin to separate individual horizons, so "topsoil" becomes a mixture of all premining soil horizons. On old mines and some newer mines where "topsoil variances" have been acquired, the original soil may not be replaced during reclamation. Instead, the minesoil consists of suitable crushed rock from the coal overburden. Irregardless of the mining method, properties of replaced soil horizons will always differ from properties of the premine soils. Disturbance destroys or degrades soil structure and water flow channels. Even though mining disrupts the soil forming processes, these processes begin functioning again as soon as the minesoils are reclaimed, and rapid pedogenesis may take place in fresh unweathered materials. Distinct A horizons, 5 or 6 cm thick, may develop in 3 years or less. These A horizons generally have weak to moderate granular structure. Weak cambic horizons have been described in fine-textured minesoils after 10 to 20 years of weathering. Discontinuities are common in minesoils due to grading and layering of dissimilar spoil types. Subsurface layers are frequently compacted, limiting water movement and pedogenesis. Many minesoils either have undergone or are currently undergoing acid-sulfate weathering. The effect of this process on minesoil properties depends upon the amounts of pyrite ($FeS_2$) present in minesoil parent materials. Jarosite mottles and iron oxide stains may be common in subsurface horizons of pyritic minesoils, and the pH may be extremely low (<4.0), affecting the capability of minesoils to support vegetation. In western states, saline and sodic spoil materials are often placed beneath 1 to 3 feet of coversoil. Salts from spoils slowly migrate upward into the coversoil causing concern as to the longterm minesoil productivity.

ADDITIONAL KEY WORDS: Minesoil properties, acid-sulfate weathering, soil development.

This page intentionally left blank.

Nutrient Cycling: the key to reclamation success?

John P Palmer, Director (Environmental Science), Richards Moorehead and Laing Ltd, Ruthin, Wales, United Kingdom.

The results of research into nutrient cycling on reclaimed land has enabled the control of nutrient input and species content during reclamation schemes to provide for faster nutrient cycling leading to improvements in ground cover, erosion control and appearance. However, the ecological value of abandoned mine lands where nutrient input is low and cycling very slow, is often greater than that of many reclamation schemes. Research has indicated that a 'capital' of nutrients is requred in reclamation schemes for an ecosystem to be self sustaining, that intermittent inputs of nitrogen often lead to moribund vegetation with nutrients immobilised and that small but frequent pulses of nutrients as observed in legume supported swards or others where nutrient cycling is not obstructed can support effective reclamation schemes of some ecological value. These findings are considered in the context of the manipulation of nutrient cycling in order to enhance the ecological value of reclaimed sites.

ADDITIONAL KEY WORDS : ecological value, colliery spoil, china clay waste, metalliferous spoil, nitrogen, phosphorus, legumes.

This page intentionally left blank.

Soil Microbial Processes and Dynamics: Their Importance To Effective Reclamation.John C. Zak, Dept. Biological Sciences, Texas Tech Univ; Suzanne Visser, Kananasksis Centre, Univ. Calgary; Philip R. Fresquez, Los Alamos National Laboratory.

The activity of the soil microflora is regulated by complex and synergistic interactions between carbon and nutrient availabilities, microfaunal grazing and abiotic contraints. Following mining activity,the recovery of the soil decomposer systems, in terms of species abundance, composition, and activity,is predicated upon the ability of the reclamation program to reestablish these synergistic inteactions. Unless well considered, reclamation efforts may initiate recovery sequences that will not lead to the effective development of belowground microbial processes and interactions.

ADDITIONAL KEY WORDS: Soil microflora, Decomposition, Microbial activity

This page intentionally left blank.

The use of Mycorrhizal Fungi in Reclaiming a Landscape: Importance and Predictability. Michael F. Allen, Associate Professor, and Carl F. Friese, Research Associate, Department of Biology, Systems Ecology Research Group, San Diego State University, San Diego, CA 92182-0057.

Mycorrhizal fungi form mutualistic associations with desirable plant species. They regulate nutrient, water, and $CO_2$ uptake of individual plants, alter competitive outcomes between plants, and alter soil nutrient availability. Nevertheless, there is no single "mycorrhizal response" as a high diversity of these fungi are represented in any natural area and the differing fungi have differing effects on host plants. Moreover, few of the diverse mycorrhizal fungi important for reclamation can be grown in large volumes for inoculation and these are only the ectomycorrhizal forms. However, many mycorrhizal fungi immigrate rapidly by animals and wind onto disturbed sites. By applying ecological theory, we can often predict which vectors of dispersal and what patterns of natural invasion occur in some habitats. We suggest that the application of these models should lead to planting better plant species mixtures and placing those plants in patterns that enhance natural mycorrhizal fungal invasion and establishment. Because of these factors, it is essential that several parameters be considered for reclamation success. These primarily include the spatial arrangement of plants and the animal immigration carrying the fungi. Subsequently, assessments of mycorrhizal density (plant + fungus) and mycorrhizal fungal species diversity would determine the success of management practices.

This page intentionally left blank.

**Faunal community development on disturbed lands: An indicator of reclamation success.** Robert R. Parmenter, Department of Biology, University of New Mexico, Albuquerque, and James A. MacMahon, Dean, College of Science, Utah State University, Logan, Utah.

The contemporary approach to reclamation of disturbed lands usually follows an ''agricultural'' strategy, which oftentimes results in the development of spatially homogeneous flora dominated by a small number of species. In addition, the planting of alien or ''native-but-not-resident'' species creates unique floral communities that historically have not existed on the site or in the region. As a result, recolonization of animals is accomplished by only a small subset of the potential species pool, and may result in dramatic population increases of certain opportunistic species that have been released from resource limitation or competitive and predatory pressures. In many cases, the faunal communities of reclaimed sites bear little resemblance to pre-disturbance faunas or faunas of neighboring undisturbed sites. In view of the importance of invertebrate and vertebrate wildlife to ecosystem function (e.g., pollination, herbivory, seed predation and dispersal, soil aeration, litter decomposition), we suggest that an analysis of the faunal community development (including trophic organization, species dominance hierarchies, guild apportionments, population demographics, and species turnover rates) can be used as an evaluation method for successful reclamation. We further suggest that, during revegetation planning, the deliberate inclusion of heterogeneous plant architectures (both horizontal and vertical) at various spatial scales (micro-, meso- and macro-) will result in a landscape mosaic more conducive to the development of a highly diverse, self-perpetuating faunal community.

This page intentionally left blank.

Seed and Seedbed Ecology, James A. Young, Agricultural Research Service, USDA, Reno, Nevada.

Most procedures for revegetation of lands disturbed by surface mining involve the planting of seeds in some form of seedbed. Seeds are essentially embryonic plants held in a state of rest or dormancy by desiccation. Growth of the embryonic plant will resume as the seeds rehydrate under suitable incubation temperatures. Seedbeds serve as 1) a matrix to supply the seeds with moisture, 2) insulation controlling diurnal temperature fluctuations, and 3) a medium for subsequent seedling growth. The inherent physiological system of the seeds interacts with stimuli received from the seedbed environment to control the quantity and periodicity of seed germination in revegetation attempts.

ADDITIONAL KEY WORDS: Germination, disturbed land seeding, seedbed construction.

This page intentionally left blank.

Evaluating Community Level Processes to Determine Reclamation Success. Edith B. Allen, Research Associate Professor, Department of Biology and Systems Ecology Research Group, San Diego State University, San Diego, CA 92182-0057.

The 1977 SMCRA refers to the reclamation of community level structure and processes in Sec. 515 (19) when it requires the operator to: "establish...a diverse, effective, and permanent vegetative cover of the same seasonal variety..." Naturally high levels of diversity can seldom be economically replicated on mined land, but a functional diversity that includes the dominant life forms and plants of varying phenology can be achieved. Within- and between habitat diversity also need to be considered, as natural landscapes contain heterogeneous patches of vegetation. An effort to simulate natural patchiness may result in more successful reclamation if plant mixtures are chosen for adaptations to microenvironmental conditions. To maximize species diversity and productivity, plants should be chosen to minimize competitive interactions in mixtures. This can be done by choosing species that are different in their temporal and spatial use of resources. The goals of high diversity and high productivity are sometimes at odds, as there are numerous examples of high yielding species that respond to fertilization, and that exclude more slowly growing species. The goal of reclamation is to create communities that are as resilient and stable as are undisturbed communities, or more so. There are as yet few examples of stable, artificially reclaimed communities that have also met the criteria of diversity. With time, researchers will be able to test hypotheses concerning long-term stability.

This page intentionally left blank.

Successional Processes: Importance for Obtaining and Evaluating Reclamation Success. James A. MacMahon, Dean, College of Science, Utah State University, Logan, Utah, Jeanne C. Chambers, Plant Ecologist, Intermountain Research Station, Logan, Utah, and Gary L. Wade, Botanist, Northeastern Forest Experiment Station, Berea, Kentucky.

Numerous biotic and abiotic factors affect succession. In the reclamation of disturbed lands some of these factors are under the control of humans - many are not. The key is in knowing for any given ecosystem what effects inputs of the reclamationist have on these factors and, consequently, on the rate, pattern, and trajectory of succession. This information gives reclamationists the ability to select the most appropriate land use and to "direct" successional processes towards that land use. It also gives an objective basis for evaluating if reclamation success has been achieved and, in some cases, indicates the best measures of success. This paper focuses on how knowledge of successional processes in different biomes can be used to obtain desired outcomes and select appropriate measures of reclamation success. Successional processes common to all biomes are defined and differences among those biomes are examined. The manner in which these differences influence inputs of reclamationists and consequent successional outcomes is then explored. Finally, we examine whether currently used measures of reclamation success are appropriate for the different biomes based on the desired successional stage and end land use.

This page intentionally left blank.

Landscape Ecology and Reclamation Success. Paul T. Tueller, Professor of Range Ecology, Department of Range, Wildlife and Forestry, University of Nevada, Reno.

To all landscapes, including reclaimed ones, there is a spatial diversity with both horizontal and vertical boundaries. The juxtaposition of these boundaries and the influence of these entities (patches or polygons) one on the other must be considered as reclamation is carried out. It is conventional wisdom that reclaimed landscapes seldom correspond to predisturbance landscapes. The arrangement of plant communities on reclaimed landscape will follow patterns explained by ecosystem ecology and therefore the resultant patterns are predictable leading to knowledge useful to managers. The result will be improved long-term management of reclaimed landscapes. Remote sensing, because of it's unique capability of looking at spatial aspects of landscapes and relating them to ecological ancillary information, can provide data useful for the understanding of these relationships. This, however, will require a detailed understanding of the spectral characteristics of scene components followed by pixel modelling to map and interpret the landscape polygons that are depicted as spectral information classes. Geographic Information Systems (GIS) technology for spatial data analysis coupled with multivariate statistical evaluations of the data will provide rapid interpretation of potential scenarios as outcomes to the reclamation process. Multivariate statistical approaches include both classifications to place like segments of a landscape together and ordinations to assist in explaining the environmental gradients that control organisms on a landscape.

ADDITIONAL KEY WORDS: Landscape ecology; reclamation; remote sensing; Geographic Information Systems.

This page intentionally left blank.

Cumulative Effects of Mining on Hydrology, Water Quality, and Vegetation. Howard G. Halverson, Project Leader, Northeastern Forest Experiment Station, Berea, Kentucky and Roy C. Sidle, Project Leader, Intermountain Research Station, Logan, UT.

Evaluating the cumulative effects of mining and other land disturbances at the watershed-scale poses a challenge to researchers and practitioners. Off-site cumulative effects may arise as a result of multiple mining disturbances occurring simultaneously within a watershed or from the interaction of mining activities with adjacent land uses (e.g., grazing, residential development, recreation, timber harvest). Progressive surface and underground mining may generate on-site cumulative effects by modifying soil and vegetation composition, slope stability and surface erosion potential. In addition, this temporal sequence, typical of many coal mining operations in the East and gold mining operations in the West, can impact off-site resources such as water chemistry, channel sedimentation, riparian vegetation, aquatic habitat, and peak flows. To address these important cumulative effects issues we need to apply our knowledge of physical, chemical, and ecological processes to the temporal and spatial scales at which these activities occur.

ADDITIONAL KEY WORDS: multiple land-use; peak flows; weathering processes; erosion; aquatic habitat.

This page intentionally left blank.

Ecological Importance of Post-reclamation Land Uses on Evaluating Reclamation Success. R. E. Ries and R. S. Hedin. Range Scientist, USDA-Agricultural Research Service, Northern Great Plains Research Laboratory, Mandan, ND and Research Biologist, USBM-Biotechnology Group, Pittsburgh Research Center, Pittsburgh, PA, respectively.

Determination of post-reclamation land use or uses is an important first step in evaluating reclamation success. An effort should be made to work with and not against ecological processes that occur during the liability period which can enhance final reclamation success.

ADDITIONAL KEY WORDS: Succession; Environmental factors; Management

This page intentionally left blank.

# Minesoil Productivity and Postmining Land Uses

This page intentionally left blank.

# GUIDE FOR SELECTING AN APPROPRIATE METHOD TO ANALYZE THE STABILITY OF SLOPES ON RECLAIMED SURFACE MINES[1]

John J. Bowders, Jr. and Sun Chai Lee[2]

Abstract: Geotechnical engineers have long recognized the seriousness of slope failures, and numerous methods of slope stability analysis have been developed and are available. In addition, the number of computer programs which utilize one or more of the methods of analysis has been expanding rapidly. Unfortunately, there still exists considerable uncertainty about the accuracy of the various methods of analysis. Many of the programs allow rapid application of sometimes old and approximate methods. Often, program users do not have a sufficient background in the principles involved in the stability analysis. Selection of the appropriate method of analysis and input variables is paramount to obtaining the true margin of safety against slope failure. Use of an inappropriate method or nonrepresentative variables can lead to prediction of what appears to be a safe slope; however, post-reclamation failure indicates otherwise. In order to assist both the expert and novice analyzer of slope stability, a detailed investigation of the accuracy and applicability of six varied methods of analysis was undertaken. The assumptions and mechanics underlying each method were studied. The methods were used to evaluate the stability of several hypothetical slopes. Safety factors produced by the various methods were compared and method applicability was assessed. The six methods were also used to analyze the stability of three actual slope failures. Results of the investigation have been compiled into a table listing the methods and their respective applicability to various slope conditions. This table may be used as a quick reference to distinguish between the numerous methods of analysis. It provides those charged with ascertaining the stability of slopes a process to select the most appropriate method of analysis and to know the relative accuracy of the method. Use of the table developed through this study should promote, increased accuracy in the analysis of proposed returned-to-approximate-original contour slopes. Thus, post-reclamation failures can be reduced through accurately analyzing proposed reclaimed slopes and altering potentially unstable designs.

Additional Key words: Slope stability, analysis, factor of safety, ground movements, slips, limit equilibrium methods, computer methods.

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]John J. Bowders is an Assistant Professor, Department of Civil Engineering, West Virginia University, Morgantown, WV 26506. Sun Chai Lee is a Geotechnical Engineer, Gannett Flemming, Camp Hill, PA 17105.

## Introduction

Since 1977, surface mine operators have been required to restore their mined sites to the approximate pre-mining contours. Operators must place the spoil material against the face of the rock (highwall) that becomes exposed during mining. This reclamation practice has created some difficulties for mines located in steeply contoured regions such as Appalachia. Zipper and his co-workers (1985) investigated this practice and found numerous failures of reclaimed slopes. Factors contributing to failure of these slopes included: excessively steep regraded slopes, high pore water pressures in the fills, toes of slopes being located beyond the edge of the mining bench, and the increased speed of weathering of the newly exposed material. Failure of these slopes, although constructed as per federal regulations, indicate that it may not be possible to restore every site to the original contours.

Although not specifically cited as a cause of failure, inadequate analysis of the stability of reclamation slopes may be contributing to the list of failed reclaimed slopes. Regardless, if insufficient analysis is not the cause, improvements in the existing analyses may lead to lowering the number of failures. The stability of slopes against sliding is a serious problem and as such, geotechnical engineers have led an effort to develop methods for the accurate analysis of the stability of slopes. As a result of these efforts, numerous methods of analysis have become available to the practicing engineer. Unfortunately, considerable uncertainty remains with regard to knowledge about the accuracy of each method. In addition, a large variety of different computer programs exists for the analyses of slope stability. Many of these permit the rapid application of sometimes old and approximate methods.

Occasionally, users of the computer programs have insufficient backgrounds in the geotechnical principals involved in slope stability analysis. They may not have a good knowledge of the type of information they will obtain from the computer solution to the problem. This can lead to selection of an inappropriate computer method for a specific slop analysis or the acceptance of inaccurate solutions. Either of these actions can result in the failure of the analyzed slope

In order to increase the ability of slope stability analysts to select the appropriate method of analysis for a given condition, a set of guidelines and recommendations has been developed. The recommendations allow the user to select the appropriate method of analysis and also provide him with a range in the relative degree of conservancy in the analysis.

## Selection of the Methods of Slope Stability Analysis

Many methods of slope stability analysis have been developed in attempts to permit the quantitative assessment of the safety or stability of slopes. However, they can be classified into two approaches: deformation approach and limit equilibrium approach. The deformation approach is based on stress-strain characteristics of the soil and requires an appropriate analytical technique to determine the deformation of the slopes. Practical application of these methods have been limited. As such, engineers still rely on the limit equilibrium approach.

All limit equilibrium methods have similar features with regard to their computational procedures. A slip mechanism is postulated along a known or assumed failure surface. Then, a condition of incipient failure along the potential failure surface is assumed, i.e., a limit equilibrium state is reached. The shear resistance required for equilibrium is computed and compared with the available shear strength. The resulting ratio (available shear strength to required shear strength) is termed the factor of safety against slope failure. The factors of safety for a number of trial slip surfaces are computed to find the minimum factor of safety, i.e., the most critical slip surface. The differences among the various methods of limit equilibrium result from the assumptions made in order to satisfy the equations of static equilibrium. These differences in assumptions lead to three different categories of limit equilibrium methods:

1) methods which satisfy overall moment equilibrium but not individual slice moment equilibrium,
2) methods which satisfy overall and individual force equilibrium, and
3) methods which satisfy overall moment and individual slice (complete) equilibrium.

The variety of solutions and computer programs available for each of these catagories is extensive. Two of the most popular methods from each of the catagories were chosen to evaluate the applicability and accuracy of each method. The method are listed in Table 1.

## Methodology

After selecting the methods of analysis shown in Table 1, three hypothetical slope cases were developed in order to test each method and compare the results. Parametric studies on these slopes were designed to yield as much information as possible about the applicability of the methods and to provide information for use in developing guidelines for selecting appropriate methods of analysis of slope stability.

The slopes were divided in homogeneous and nonhomogeneous cases. The homogeneous slope is shown in Figure 1. Shown in Figures 2 and 3 are the nonhomogeneous slopes used in the parametric study, a fill embankment on a clay foundation and a sloping core dam on a rock or firm foundation, respectively.

Three classes of stability problems as related to the drainage conditions were considered for each class of slope. The three classes included undrained, drained and partially drained problems. The ranges of the variables used in the sensitivity analyses of the homogeneous slope are shown in Table 2. The corresponding values used in the sensitivity analyses of the nonhomogeneous slopes are shown in Table 3.

In addition to the hypothetical slopes, the six methods of analysis were each used to analyze the stability of three actual slope failures. The data for these failures were obtained from the literature. They were chosen for their completeness with regard to soil property description, geometry, and behavior. Case 1 was a test embarkment on sensitive clay (Ladd 1972). Case 2 was a flood control dam on a clay shale foundation (U.S. Army 1963). Case was a test embankment on soft sensitive cemented clay (La Rochelle et.al. 1974). These three cases provided circumstances which are reasonably representative of a majority of such failures.

## Results of The Stability Analyses

The research program consisted of sensitivity and comparative analyses of the six methods of limit equilibrium slope stability analysis. Both drained and undrained conditions were considered for homogeneous and nonhomogeneous slopes.

In general, the most important parameter in the stability analysis was the shear strength, i.e., the soil's cohesion and friction angle. Regardless of the slope height or inclination from the horizontal, the strength parameters played the dominant role in determining the safety factor of the slope against failure. Lee (1989) provides extensive tables and figures illustrating the influence of the parameters listed in Tables 2 and 3. The results presented by Lee make it possible to determine the percentage of change in the safety factor for a given change in any of the variables used in the analysis.

The six analysis methods were used to predict the factor of safety of the slopes in each of the three case histories. The methods were also used to predict the failure surface. The results of the prediction are shown in Table 4. In each case, the actual factor of safety is known to be unity at the time of failure. Therefore, those predictions which come nearest to unity are the best methods at predicting failure, provided the assumed or predicted failure surface is representative of the actual surface. Examination of Table 4 shows that the Spencer method (SP) (Spencer 1967) provides the most nearly correct values of the factor of safety in each case. The failure surfaces predicted using the SP method, shown in Figures 4, 5, and 6, are representative of the actual surfaces.

In the comparative analyses, it was found that the Ordinary Method of Slices (OMS) (Peck 1967) can lead to unreasonably low (conservative) factors of safety. For total stress (undrained) analyses, the degree of conservatism ranged from 1 to 7 percent. For the effective stress analyses, the extent of the underestimate increased with increasing magnitude of pore water pressure. The Simplified Bishop (SB) method (Bishop 1955) (satisfies overall moment equilibrium) resulted in slightly lower factors of safety than the methods which satisfied all static equilibrium conditions. Factors of safety were in error by no more than +/- 7 percent. This was true in both the drained and undrained analyses. However, the SB method is limited to circular failure surfaces. The Simplified Janbu (SJ) method (Janbu 1954) resulted in factors of safety within +/-10 to +/-15 percent of the correct value. The results were equally as accurate for the drained and undrained analyses and for both circular and non-circular failure surfaces. The Spencer method satisfied all conditions of equilibrium and gave accurate results for all practical conditions. The Lowe & Karafiath method (1960) and the Corps of Engineers (COE) method (1968) generally gave values of factor of safety which were too high (unconservative) for homogeneous slopes. The Corps of Engineers method generally gave values which were higher (less conservative) than the Lowe & Karafiath (LK) method.

## Selection Guide to the Method of Slope Stability Analysis

Shown in Table 5 are the recommendations of the various methods of slope stability analysis for different slope problems and conditions. Four ratings (1 through 4) were established based on the degree of conservatism with 1 being the best method and 4 being the least suitable or not applicable method. To use Table 5, the type of slope problem must first be identified (Cases A through F). Next, examine the left side of the table to determine the rating of analysis methods and select the method which yields the best rating, e.g., 1. If the performance level of a certain analysis method is desired, then look across the table for that particular method. The slope problem or condition which has a rating of 1 indicates that the given method of analysis is best suited for that particular type of slope problem.

For example, consider a slope problem of Case A. The OMS, SB, and SP method are recommended for use because they have the best rating. In addition, to check the slope problems for which the OMS method is most appropriate, examine across the table for the rating of the OMS method in each of the slope problems. OMS method is rated as 1 for cases A, B, D, and E. The comments on the right side of Table 5 also indicate that the OMS method can yield results of up to minus 50 percent in error (conservative) of the actual value of the factor of safety.

## Conclusions

A detailed investigation of the accuracy and applicability of six varied methods of slope stability analysis was undertaken. The assumptions and mechanics underlying each method were studied. The methods were used to evaluate the stability of several hypothetical slopes. Factors of safety produced by the various methods were compared and applicability of the methods was assessed. The six methods were also used to analyze the stability of three actual slope failures.

Results of the investigation have been compiled into a table listing the methods and their respective applicability to various slope conditions. This table may be used as a quick and convenient reference to distinguish between the numerous methods of analysis. It provides those charged with ascertaining the stability of slopes a process to select the most appropriate method of analysis and to know the relative accuracy of the method. Use of the table developed through this investigation should promote increased accuracy in the analysis of proposed returned-to-approximate-original contour slopes. Thus, post-reclamation failures can be reduced through accurately analyzing proposed reclaimed slopes and altering potentially unstable designs.

## Acknowledgements

The second author completed parts of this research while under financial support by the Department of Civil Engineering and the Energy and Water Research Center at West Virginia University. The support and interest of Dr. Lyle K. Moulton and Dr. Carl L. Irwin of those two organizations, respectively, are gratefully appreciated.

## Literature Cited

Zipper, C.J. Bell, and W.L. Daniels, 1985 A Detailed Study of the Stability, Land Use Potential, and Erosivity of Surface Mined Lands Returned to Their Approximate Original Contours Proc., of the Powell River Project Symposium, Clinch Valley College, Wise VA, pp. 38-44.

Ladd, C.C., 1972 Test Embankment of Sensitive Clay, Proc., ASCE Specialty Conf. on the Performance of Earth and Earth Supported Structures, Purdue Univ., Lafayette, IN, June 11-14, pp. 101-128

La Rochelle, et.al., 1974 Failure of a Test Embankment on a Sensitive Champlain Clay Deposit, Canadian Geotechnical Journal, 11:142-164.

Lee, S.C., 1989 The Applicability of Limit Equilibrium Methods of Analysis To Different Types of Slope Stability Problems, Ph.D. Dissertation, West Virginia University, p. 231.

U.S. Army Engineering District, 1963 Waco Dam: Report on Embankment Slide and Reanalysis of Design Vols. 1, 2, and 5, Fort Worth, TX.

Janbu, N., 1954 Application of Composite Slip Surface for Stability Analysis. Proceedings Europe Conf. on Stability of Earth Slopes, Vol. 3, Stockholm, Sweden, pp. 43-49.

Spencer, E., 1967 A Method of Analysis of the Stability of Embankments Assuming Parallel Interslice Forces, Geotechnique, V. 17, March, pp. 11-26.

Bishop, A.W., 1955 Use of the Slip Circle in Stability Analysis of Slopes, Geotechnique, Vol. 5, pp. 7-17.

Lowe, J., and L. Karafiath, 1960 Stability of Earth Dams Upon Drawdown. Proceedings 1st Pan American Conf. on Soil Mechanics and Foundation Engineering, Mexico City, Vol. 2, pp. 537-552.

U.S. Army Corps of Engineers, 1968 Engineering Design, Stability of Earth and Rockfill Dam, Revision Draft, February.

Peck, R.B., 1967 Stability of Natural Slopes, Journal of the Soil Mechanics and Foundation Div., ASCE Vol. 93, SM4, July, pp. 403-417.

Table 1. Slope Stability Methods Evaluated for Applicability Accuracy

| Equilibrium Conditions | Stability Analysis Method | Computer Program | Ref. |
|---|---|---|---|
| Moment and Force | Spencer (SP) | 1 | Spencer, 1967 |
| | Simplified Janbu (SJ) | 1 | Janbu, 1954 |
| Force | Lowe & Karafiath (LK) | 2 | Lowe and Karafiath, 1960 |
| | Army Corps of Engrgs (COE) | 2 | U.S. Army, 1968 |
| Overall Moment | Ordinary Method of Slices (OMS) | 3 | Peck, 1967 |
| | Simplified Bishop (SB) | 1,3 | Bishop, 1955 |

1. STABL5, Department of Civil Engineering, Purdue University.
2. K-SLOPE, Department of Civil Engineering, West Virginia University.
3. STABR, School of Civil Engineering, Univ. of California, Berkeley.

Table 2. Ranges of Variables for Sensitivity Analysis of Homogeneous Slopes.

| Parameter | Symbol | Range | Units |
|---|---|---|---|
| soil unit weight | $\gamma$ | 90 - 135 | $lb/ft^3$(pcf) |
| soil frictional angle | $\phi$ | 0 - 30 | degrees ($^\circ$) |
| soil cohesion | C | 100 - 2100 | $lb/ft^2$(psf) |
| slope height | H | 10 - 100 | feet (ft) |
| slope inclination | B | 15 - 90 | degrees ($^\circ$) |
| porewater pressure ratio | Ru | 0 - 0.5 | dimensionless |

Table 3. Range of Variables for Sensitivity Analysis of Nonhomogeneous Slopes.

Undrained Analysis

| Embankment / Dam Shell | | Foundation Soil / Dam Core | |
|---|---|---|---|
| C = 600 psf | $\phi = 0^\circ$ | C = 400 psf | $\phi = 0^\circ$ |
| C = 800 psf | $\phi = 0^\circ$ | C = 2100 psf | $\phi = 0^\circ$ |
| C = 1000 psf | $\phi = 0^\circ$ | | |

Drained Analysis

| Embankment / Dam Shell | | Foundation Soil / Dam Core | |
|---|---|---|---|
| $\bar{C}$ = 0 psf | $\bar{\phi} = 30^\circ$ | $\bar{C}$ = 200 psf | $\bar{\phi} = 11^\circ$ |
| $\bar{C}$ = 200 psf | $\bar{\phi} = 30^\circ$ | $\bar{C}$ = 600 psf | $\bar{\phi} = 18^\circ$ |

Porewater Pressure Ratio: Ru = 0, 0.25, 0.50

Table 4. Results of Slope Stability Analysis for the Case Histories

| Case History (Ref.) | OMS | SB | SJ | SP | L & K | COE |
|---|---|---|---|---|---|---|
| (1) | 0.916 | 0.994 | 1.075 | 1.020 | 1.148 | 1.196 |
| (2) | 0.986 | 1.070 | 1.043 | 1.030 | 1.110 | 1.114 |
| (3) | 0.971 | 1.021 | 1.023 | 1.007 | 1.134 | 1.158 |


(1.) Ladd, C.C. 1972 Test Embankment on Sensitive Clay. Proceedings Specialty Conference on Performance of Earth and Earth Supported Structures, Purdue University, Lafayette IN, June 11-14, pp. 101-128.

(2.) U.S. Army Engineer District,Fort Worth, TX, 1963, Waco Dam: Report on Embankment slide and Reanalysis of Design. Vols. 1, 2, and 5.

(3.) La Rochelle et.al., 1974 Failure of a Test Embankment on a Sensitive Champlain Clay Deposit. Canadian Geotechnical Journal, Vol. 11., pp. 142-164.


Table 5. Selection Guide to Methods of Slope Stability Analysis

| Methods of Analysis | Types of Slope Problem/Situation A | B | C | D | E | F | Comments |
|---|---|---|---|---|---|---|---|
| OMS | 1 | 1 | 4 | 1 | 1 | 4 | Up to -50% error when deep failure surface exists in Case C and F. (Circular Surface) |
| SB | 1 | 1 | 1 | 1 | 1 | 1 | Up to -7% error for deep failure surface and high pore pressure. (Circular Surface) |
| SJ | 2 | 2 | 2 | 2 | 2 | 2 | Vary from 0 to +15% error in Case A. Up to -12% error for deep failure surface with high pore pressures in Case C. (Any Shape Surface) |
| SP | 1 | 1 | 1 | 1 | 1 | 1 | Best Method with longer time and greater effort. (Any Shape Surface) |
| L & k | 4 | 3 | 3 | 2 | 2 | 2 | Up to +50% unconservative in Case A. Vary from +7.5% to +17% in Case E & F. (Non-Circular Surface) |
| COE | 4 | 3 | 3 | 3 | 2 | 2 | Up to +60% unconservative in Case A. Vary from +8% to +18% in Cases E & F. (Non-Circular Surface) |

Rating System:
1. Best (within -5% to -10% conservative)
2. Okay (within +5% to +10% unconservative)
3. More Unconservative (within +15% to +20% unconservative)
4. No Good (over +20% unconservative)

Slope Problems
A) Homogeneous slope under undrained condition (Figure 1)
B) Homogeneous slope under drained condition, no pore pressure (Figure 1)
C) Homogeneous slope under drained condition with high pore pressures (Figure 1)
D) Nonhomogeneous slope under undrained condition (Figures 2 and 3)
E) Nonhomogeneous slope under drained condition with high pore pressures (Figures 2 and 3)
F) Nonhomogeneous slope under drained condition with high pore Pressures (Figures 2 and 3)

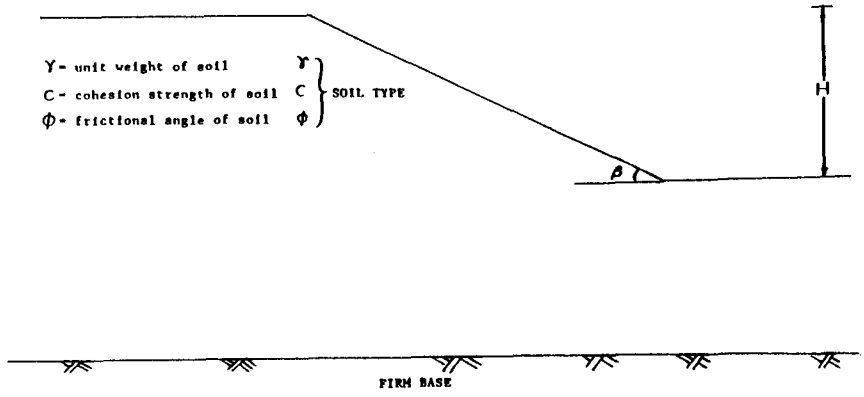



Figure 1. Typical Slope Geometry Used for the Homogeneous Soil Conditions.

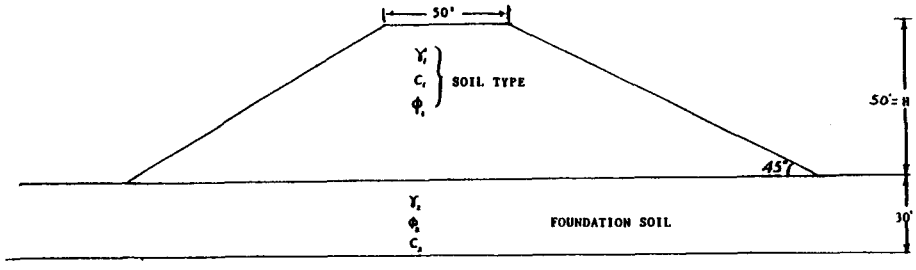



Figure 2. Typical Slope Geometry Used for the Nonhomogeneous Soil Conditions.

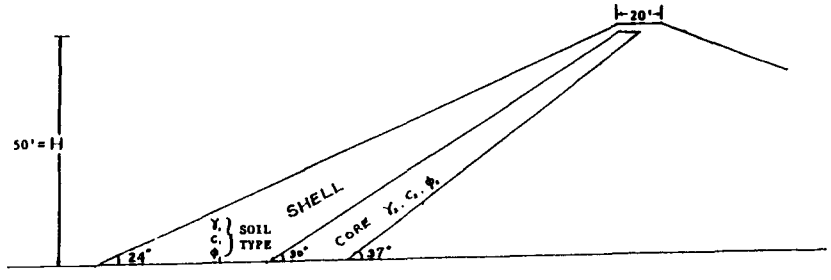



Figure 3. Typical Slope Geometry Used for the Nonhomogeneous Soil Conditions.

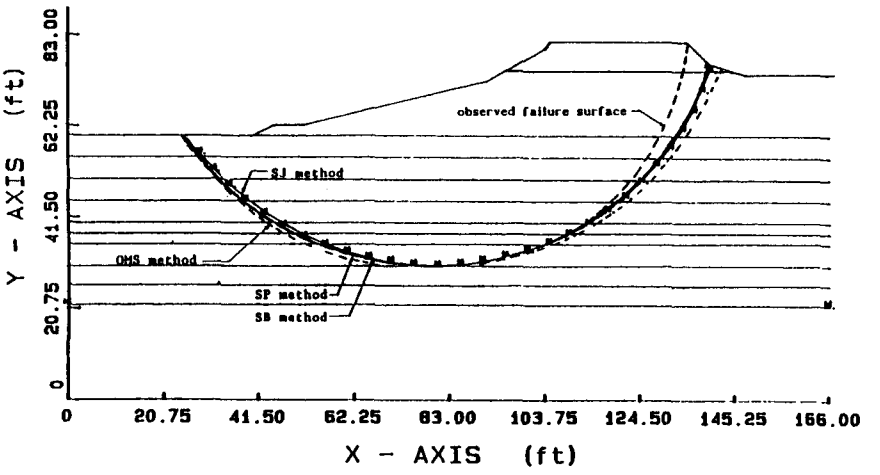



Figure 4. The Most Critical Slip Surfaces for the Test Embankment of Case History No.1.

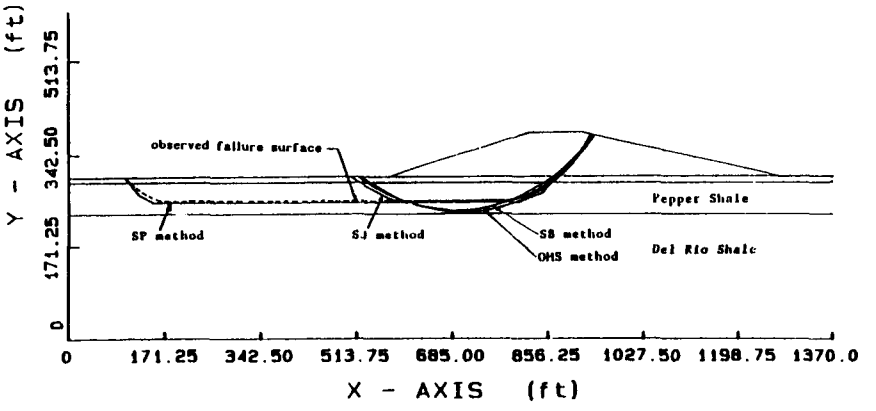



Figure 5. The Most Critical Failure Surfaces for the Case History No.2.

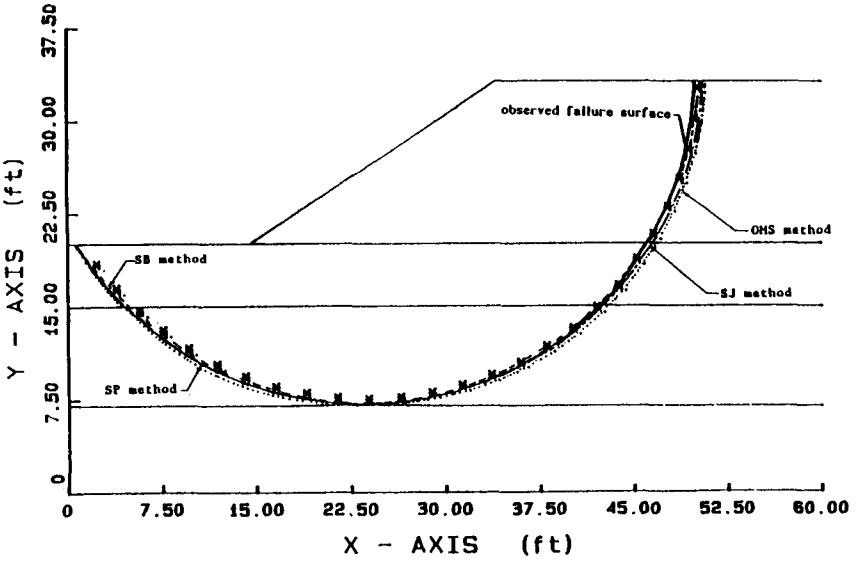



Figure 6. The Most Critical Slip Surfaces for the Test Embankment of Case History No. 3.

# APPLICATION OF THE PRINCIPLES OF MICROBIAL ECOLOGY TO THE ASSESSMENT OF SURFACE MINE RECLAMATION[1]

by

**J.A.Harris and P.Birch[2]**

---

**Abstract** Soil micro-organisms fulfill a number of vital functions in the below ground ecosystem, including mediating many plant nutrient transformations and contributing to soil structure formation and maintenance. It has been demonstrated that the disruption to the soil during surface mining operations results in changes in the size, composition and activity of the soil microbial community. Any decrease in the size and diversity of microbial populations may have serious consequences for the development of self-sustaining ecosystems on disturbed areas. However, these changes also offer an opportunity to assess and monitor the state of restored ecosystems by using the soil microbial community as an indicator of the success of differing restoration strategies. This study reports on an investigation into areas restored after opencast coal mining in Britain. Differences in soil microbial activity with depth have been used as such an indicator and significant differences in profile development have been demonstrated, with both time after restoration and management practices. Rates of dehydrogenase activity in areas restored up to 10 years prior to sampling were determined. The rates of activity in recently restored areas were only 2.5% as large as those of adjacent undisturbed semi-natural areas, and in some samples microbial activity was undetectable. Organic carbon data was less able to reveal differences between sites and treatments. Comparison with undisturbed arable sites was also made.

Additional key words: Dehydrogenase activity, soil activity profiles, opencast mine restoration.

---

[1] Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2] Dr. James Harris is Post-Doctoral Research Fellow, Environment and Industry Research Unit, Dr. Paul Birch is Head of External Affairs, School of Science, Polytechnic of East London, Romford Road, London E15 4LZ.

## Introduction

Conventionally, the assessment of the success of restoration after mineral extraction has been based on the use of chemical techniques and plant growth performance (Bradshaw and Chadwick 1980). However, chemical analyses are unable to provide a summary of the dynamics of the changes occurring within any restoration scheme, unless sampled repeatedly, and plant growth measured *in situ* may take a number of years to manifest themselves. By that stage, vital time and opportunity may have been lost. One aspect of the ecosystem which has been largely neglected and may prove extremely useful in this regard is the soil microbial community. The micro-organisms forming this community are intimately associated with the cycling of nutrients and the formation and maintenance of a stable and open soil structure, both essential features of a successful restoration. Molope et al. (1987) have shown that the microbial community is important in maintaining aggregate stability in soils, with fungi enmeshing soil particles in hyphae and bacteria secreting polysaccharides. Clough and Sutton (1978) indicated that a mycorrhizal fungus stabilized aggregates in sand dunes, and Rothwell (1984) has shown that vesicular-arbuscular mycorrhizae (VAM) had a role in producing aggregates in a surface minesoil.

Cundell (1977) was among the first authors to stress the importance of micro-organisms in the reestablishment of a functioning ecosystem in strip mined land. He reviewed the limited work done in this area and drew heavily on material from undisturbed ecosystems. Cundell reported on a study of spent shale wastes in western Colorado and lignite overburden in North Dakota. He outlined the important functions of the soil microbial community, and indicated how treatments such as fertilization, mulching, seeding, and inoculation with nitrogen fixing heterotrophs affected the establishment of a self-sustaining system. The major conclusion was, however, that a more detailed knowledge of the microbial processes involved in soil formation and plant growth on new substrates was required. This review appears to have stimulated other work from the number of subsequent citations.

Other workers have reported on the microbial communities found in restored or abandoned mineral extraction operations. Miller and Cameron (1978) and Miller and May (1981) reported on the state of the microflora of two abandoned mines in the midwestern United States. They found that the substrate, bare spoil in many cases, was unfavourable to microbial colonisation and growth and required some amendment to support plant life. There were fewer micro-organisms in the minesoil than some adjacent undisturbed areas, and there was a lower species diversity in the minesoils. This tended to produce a 'zymogenous' microflora capable of short term rapid growth in response to addition of fertilizers, and was deficient in the 'autochthonous' species capable of long-term degradation of organic matter typical of a

self sustaining system. They recommended the addition of organic amendments to spoil, a conclusion supported by subsequent workers in the field (Fresquez and Lindemann 1982; Visser 1985). Stroo and Jencks (1982) indicated that using inorganic fertilizers allowed the zymogenous population of micro-organisms to attack any organic matter which did exist, leading to regression of the system within ten years of revegetation treatments. Several workers have shown that certain aspects of civil engineering operations have deleterious effects on the soil microbial community (Abdul-Kareem and McRae 1984; Harris et al, 1989).

Recent work in Britain has indicated the importance of micro-organisms in re-establishing ecosystem function, and has suggested a link between microbial activity and soil structure (Harris and Birch 1989). There is increasing interest in restoration to conservation systems which, coupled with the traditional pasture and wild flower systems in the British Isles, has made restoration to meadow particularly attractive. There is little doubt that systems can be moved, and that suitable existing stable natural systems can be managed toward meadow. However, re-creation of such habitats after the disturbance of mining is little understood, with measures of restoration success relying on traditional methods.

## Aims

The objective was to investigate the biological activity in restorations up to 10 years old and to compare them with adjacent similar areas (used as controls) and, in particular, to investigate the potential for using soil microbial activity profiles to discriminate between treatments.

## Materials and Methods

### Site Details

Soil cores were collected from the following Open Cast Coal Sites (OCCS) and areas within them:

1. Shilo: An undisturbed site with a variety of species-rich meadows; two were sampled, one dry and one wet area. National Grid Ref SK 460460.
2. South Shilo: Restored for 5 years; two fields were sampled, one grazed and one ungrazed (cut for silage). National Grid Ref SK 475435
3. Old Shilo: Restored for 10 years; one grazed and one ungrazed restoration were sampled. National Grid Ref SK 470450

These areas were adjacent to one another, and comprised of soils of the same soil series and textural class.

### Sampling

Five 30 cm deep cores were taken by means of sterilised 4 cm gouge augers from each area and placed in sterile plastic bags. They were transported back to the laboratory in insulated bags containing ice packs. In the laboratory the cores were cut into 5 cm segments in order of increasing depth, representing depths of 0-5, 5-10, 10-15, 15-20, 20-25, and 25-30 cm. The core segments were chopped, using aseptic technique, and kept separate

for subsequent analysis. Fresh soil was used for moisture content and dehydrogenase activity assay, a separate subsample was air-dried at 30°C and ground to pass through a 0.5 mm sieve for carbon analysis.

Analysis

Dehydrogenase activity was determined by the method of Tabatabai (1982). A subsample of moist soil was incubated with a triphenyl tetrazolium chloride solution for 24 h at 37°C, and the formazan produced extracted with methanol, and the absorbency measured at 485 nm. Carbon was determined using a wet-oxidation technique (Nelson and Sommers 1982).

Results and Discussion

Steady decreases in microbial activity with increasing depth were found in the case of both profiles from the undisturbed areas, with high activities in the top 0-5 cm segments (Fig 1, bars represent standard deviation). This finding is typical of profiles to be expected in a soil ecosystem that has developed over many years and has remained essentially undisturbed. Over time, organic matter incorporation into the surface layers and oxygen depletion in the deeper layers has led to microbial activity in the surface as compared to the deeper layers.

In the five year old grazed restoration at South Shilo very poor microbial profile development has occurred, the total activity is only 2.5% of the value obtained for the undisturbed dry meadow area. There was no detectable microbial activity in the two deepest core segments at 20-25 and 25-30 cm. The ungrazed restoration has a little higher activity, and is consistent throughout the profile, albeit with some low values. The overall activity was 8.3% of that of the dry meadow control (Table 1).

The 10 year old restoration (Old Shilo) had significantly higher activity (at $p<0.001$) down the profile as compared to the five year old site, with the grazed and ungrazed profiles having activities of 19.6% and 32.4% of the control, respectively. There is marked development of a profile, especially in the ungrazed site. Fairley (1985) demonstrated that grazing led to a marked reduction in the quantity of roots produced on areas re-instated after opencast coal mining in the UK. This is an effect of the constant defoliation leading to a reduction in the above-ground photosynthetic area. This in turn would lead to less root exudate and turnover in the grazed areas as compared to the ungrazed areas.

This data suggests that there is a strong influence of both age of restoration and type of management regime on the development of activity profile. The carbon data shows a poor discrimination between the different areas (Fig. 2). Although the differences between the unworked and worked areas are clear, the differences between age of restoration and management practice are far less clear. The overall levels of carbon in the worked areas (6.13 to 8.04% profile average) are very similar to those in the dry meadow at Shilo (7.81%),

Fig. 1. Microbial Activity

(Formazan formation $\mu g\ g^{-1}$ soil $24h^{-1}$)

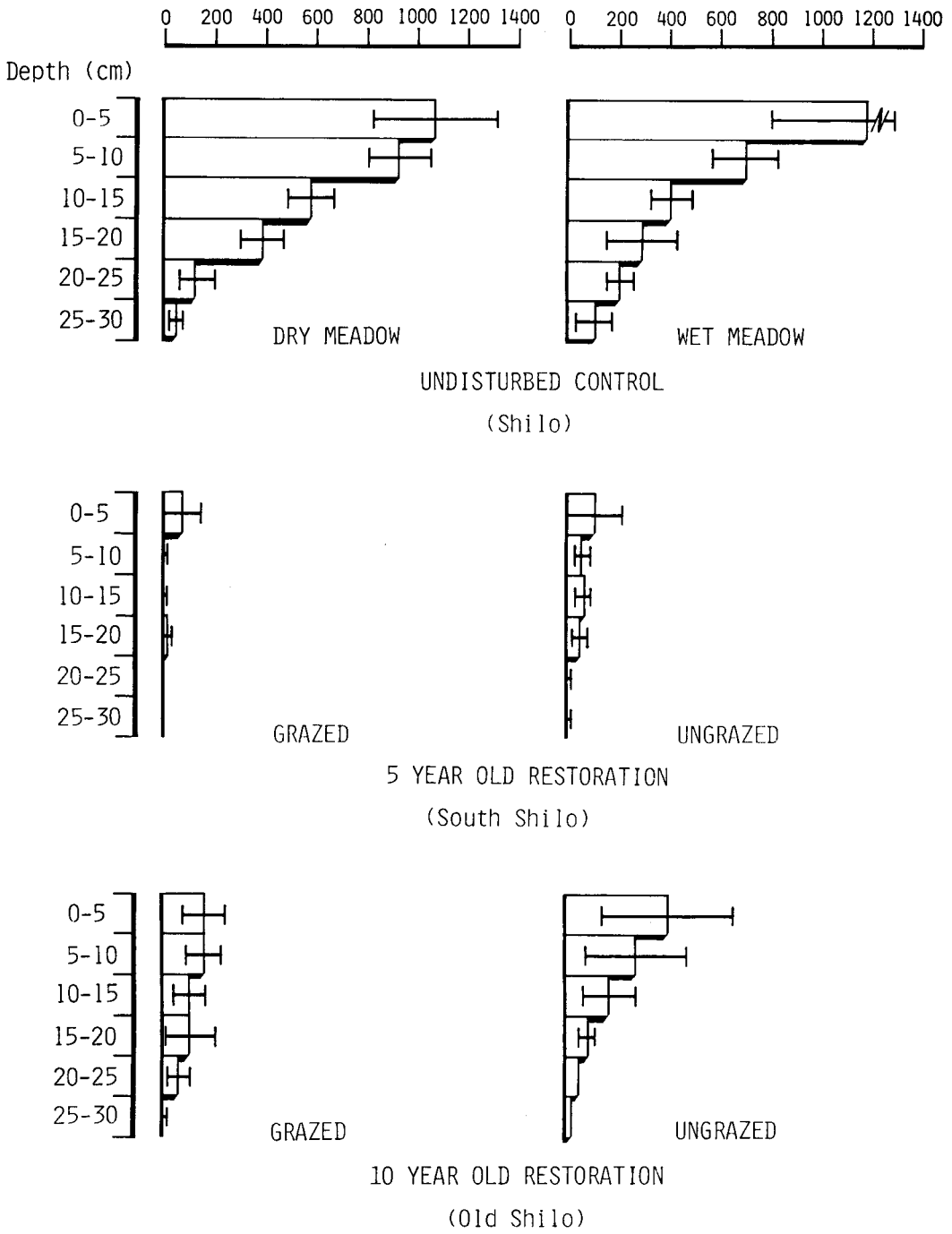


Fig. 2. Total Carbon (%)

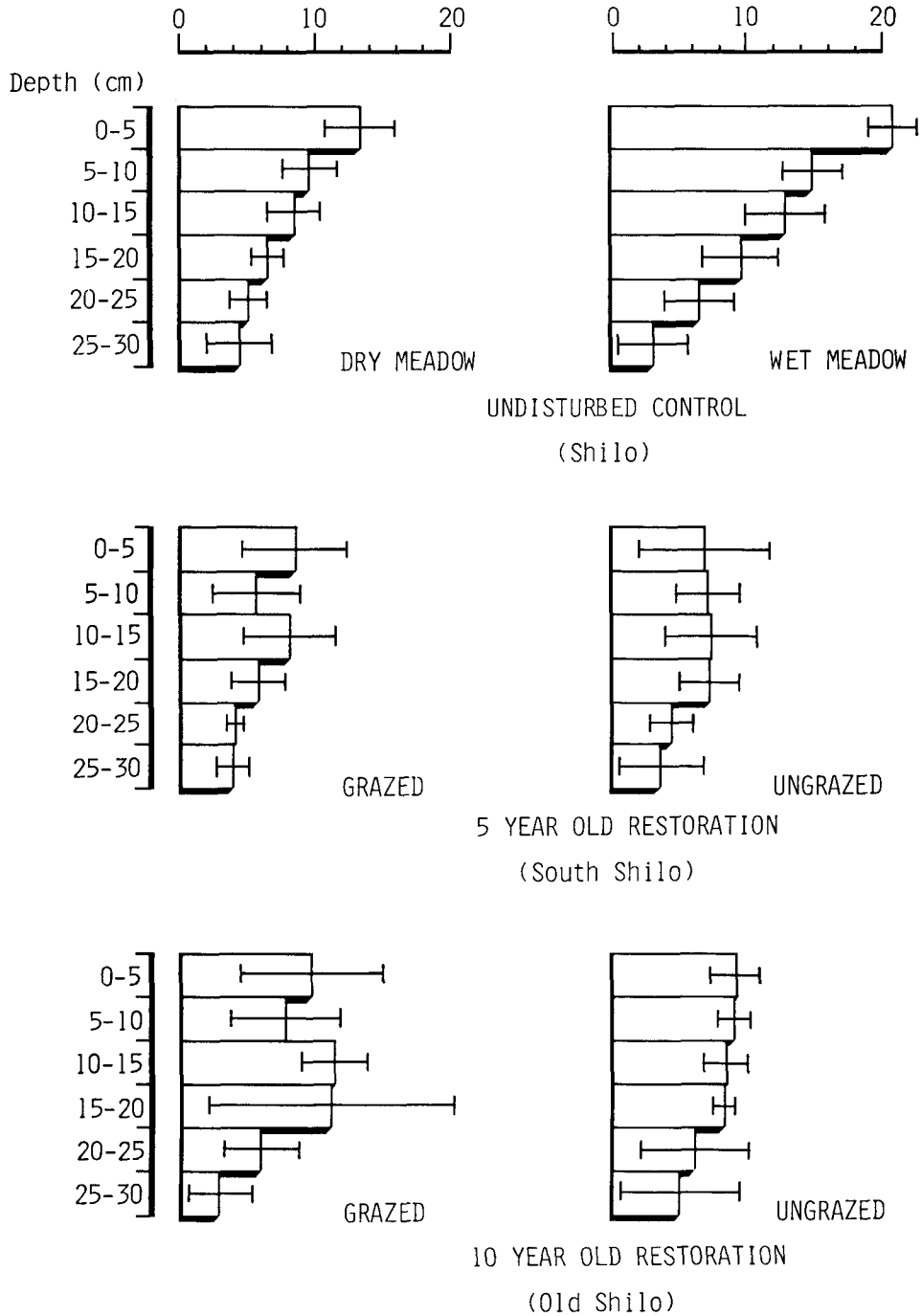


Table 1. Comparison of whole profile (0-30 cm) activity averages

| Area | Activity ($\mu g\ g^{-1}\ 24h^{-1}$). | % of dry meadow control |
|---|---|---|
| Shilo dry meadow | 515 | 100 a |
| Shilo wet meadow | 478 | 93 a |
| S.Shilo grazed | 13 | 3 d |
| S.Shilo ungrazed | 43 | 8 cd |
| O.Shilo grazed | 101 | 20 bcd |
| O.Shilo ungrazed | 167 | 32 b |
| Kirk cereal field | 140 | 27 bc |
| Kirk pasture | 210 | 41 b |
| Buckhead pasture | 580 | 113 a |

Letters following percentages indicate homogenous groups, as determined by multiple range test following ANOVA, at 95% confidence and $p<0.001$

indicating that the total amount of organic matter has returned to its former levels in the 10 year old restorations, but the profile is rather uneven with no clear depth trends as seen in the microbial activity values (Fig.1).

Table 1 allows a comparison of the data from the various Shilo sites with the 0-30 cm profile averages from three undisturbed areas sampled previously, which consisted of of a pastures and a cereal field at Kirk and Buckhead, (Harris and Birch 1989). Very similar activity was observed in the undisturbed Shilo areas as compared to the undisturbed pasture at Buckhead, which are all considerably more active than the mined areas at Shilo. However, the Old Shilo restorations have similar levels of (not significantly different at $p<0.001$) activity to the undisturbed areas at Kirk (Table 1). If this is the desired end point then the ungrazed area at Old Shilo would appear to be very close to achieving it. From this small data set, it would appear that some 25-30 years may be required for the 'full' meadow profile to reappear, at these rates of increase. However, more detailed and intensive sampling at both areas would be required to confirm this. A database of basic ecosystem types containing information on the size, composition and activity of the soil microbial community, ensuring accountability of seasonality and spatial variation. A survey of early, mid and late successional types could be carried out to provide a computer based ordination, perhaps using principal components and factor analysis in conjunction with classification procedures such as cluster analysis. This could then be used to assess the state of restored ecosystems in terms of undisturbed systems. Further, it could give early indications as to which direction the system needed to be 'pushed' in

order to achieve the desired restoration end point.

## Conclusions

This study would appear to suggest that microbial activity profiles are potentially useful tools for assessing restoration success and discriminating between different management practices, and that further work is required in this area of research.


## Acknowledgements

The authors thank Miss D. Edgerton for assistance with soil sampling and analysis. We also thank Mr I. Carolan of the Opencast Executive, British Coal, for arranging access and useful discussion on site. We acknowledge the financial support from the Opencast Executive, British Coal.


## Literature cited


Abdul-Kareem, A.W. and S.G. McRae. 1984. The effects on topsoil of long-term storage in stockpiles. **Plant and Soil** 76:357-363.

Bradshaw, A.D. and M.J. Chadwick. 1980. **The Restoration of Land: The ecology and reclamation of derelict and degraded land** Blackwell, Oxford.

Clough, K.S. and J.C. Sutton. 1978. Direct observation of fungal aggregates in sand dune soil. **Canadian Journal of Microbiology** 24: 333-335.

Cundell, A.M. 1977. The role of micro-organisms in the revegetation of strip-mined land in the western United States. **Journal of Range Management** 30:299-305.

Fairley, R.I. 1985. Grass root production in restored soil following opencast mining. In A.H. Fitter, D. Atkinson, D.J. Read, and M.B. Usher (Eds) **Ecological Interactions in Soil: Plants Microbes and Animals** Special Publication Number 4 of the British Ecological Society, Blackwell Scientific Publications, Oxford.

Fresquez, P.R. and W.C. Lindemann. 1982. Soil and rhizosphere micro-organisms in amended coal mine spoils. **Soil Science Society of America, Journal** 46:751-755.

Harris, J.A. and P. Birch. 1989. Soil microbial activity in opencast coal mine restorations. **Soil Use and Management** 5:155-160.

Harris, J.A., P. Birch, and K.C. Short. 1989. Changes in the microbial community and physico-chemical characteristics of topsoils stockpiled during opencast mining. **Soil Use and Management** 5:161-168.

Miller, R.M. and R.E. Cameron. 1978. Microbial ecology studies of two coal mine refuse sites in Illinois. **Argonne National Laboratory Report ANL/LRP-3.**

Miller, R.M. and S.W. May. 1981. Staunton 1 Reclamation Demonstration Project, Soil microbial structure and

function: Final Report. **Argonne National Laboratory Report ANL/LRP-11.**

Molope, M.B., I.C. Grieve and E.R. Page. 1987. Contributions by fungi and bacteria to aggregate stability of cultivated soils. **Journal of Soil Science** 38:71-77.

Nelson, D.W. and L.E. Sommers. 1982. Total carbon, organic carbon and organic matter. In A.L. Page (Ed) **Methods of Soil Analysis, Part 2, Agronomy Monograph No.9.** (2nd Edition) American Society of Agronomy, Madison, Wisconsin.

Rothwell, F.M. 1984. Aggregation of surface minesoil by interaction between VAM fungi and lignin degradation products of lespedeza. **Plant and Soil** 88:99-104.

Stroo, H.F. and E.M. Jencks, 1982. Enzyme activity and respiration in mine spoils. **Soil Science Society of America Journal** 46: 548-553.

Tabatabai, M.A. 1982. Soil Enzymes In A.L. Page (Ed) **methods of Soil Analysis, Part 2. Agronomy Monograph No. 9.** 2nd Edition. American Society of Agronomy, Madison, Wisconsin.

Visser, S. 1985. Management of microbial processes in surface-mined land reclamation in Western Canada In R.L. Tate III and D.A. Klein (Eds) **Soil Rehabilitation Processes, microbial analyses and Application** Dekker, New York.

This page intentionally left blank.

# FORAGE PRODUCTION ON A MINESOIL IN SOUTHERN WEST VIRGINIA[1]

B. L. Wolf, L. D. Emerson and J. C. Sencindiver[2]

**Abstract**. Forage production and legume persistence were evaluated on a Kaymine minesoil (loamy-skeletal, mixed, nonacid, mesic Typic Udorthents) in Boone County, West Virginia for the 1987 through 1989 growing seasons. The area was hydroseeded in the spring of 1985 with a mixture of grasses and legumes. Starting in 1987, three harvests per year were taken at three areas with four subplots per area. The 1987 yields averaged 2.48 ton/acre for area 1, 2.83 ton/acre for area 2, 1.94 ton/acre for area 3. The 1988 yields averaged 1.16 ton/acre for area 1, 1.28 ton/acre for area 2, and 1.15 ton/acre for area 3. The 1989 yields averaged 1.44 ton/acre for area 1, 1.78 ton/acre for area 2, and 1.81 ton/acre for area 3. The total production for each year varied greatly from plot to plot due to a number of environmental factors that included a severe drought in August 1987 and all of 1988. However, reasonable forage production was obtained over the three-year period. These results demonstrated the potential for high quality forage production on a nonacid minesoil in West Virginia.

Additional key words: Land use, minesoil classification, minesoil mapping, crop yields.

## Introduction

Studies on classification and characterization of minesoils in West Virginia began approximately 20 years ago. These studies generated data on minesoil properties and variability which resulted in the development of minesoil series. Through cooperative efforts of the USDA Soil Conservation Service and the West Virginia Agricultural and Forestry Experiment Station seven minesoil series have been identified and established in West Virginia. Five of these series were originally classified, mapped and correlated in southern West Virginia (Wolf, 1988).

Although minesoil series have been identified, land-use data for reclaimed mined lands are scarce. These data are needed to develop interpretations for land-use management decisions. Therefore, the purpose of this study was to evaluate forage production on the Kaymine series, a southern West Virginia minesoil with high potential for growing quality grasses and legumes.

## Study Area

The study area is located in Boone County, where approximately 10 percent of the county's 322,000 acres had been disturbed by surface mining by 1987 (Wolf, B. L., Soil Survey of Boone County, West Virginia. Unpublished manuscript). Boone County is in the southern coal fields of West Virginia, which have been identified as Surface Mining Province 1 (West Virginia University, 1971). This area is within the deeply dissected unglaciated Appalachian plateau. Steep and very

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exibition, Charleston, West Virginia, April 23-26, 1990.

[2]B. L. Wolf is Soil Scientist, SCS, Beckley, WV; L. D. Emerson is Agronomist, Hobet Mining, Inc., Madison, WV; J. C. Sencindiver is Professor of Soil Science, West Virginia University, Morgantown, WV.

steep slopes range in relief from 600 feet to as much as 1400 feet. The coal beds represent the earlier of two basins of coal deposition during Pennsylvanian time. In general the sulfur content of coal and overburden is low and neutralization potential ranges from low to high.

The humid, continental type climate of the area is characterized by sharp seasonal temperature changes. Average daily temperatures are approximately 73°F in summer and 34°F in winter. Average annual precipitation is 40 to 45 inches. Native vegetation of the area is mixed hardwood forest (Core, 1966).

The study site is located at the Hobet Mining, Inc. surface mine near Madison in the northwestern part of Boone county. Fiveblock and Stockton-Lewiston coal seams (Allegheny and Pottsville formations) are mined by mountaintop removal methods. Fiveblock overburden is drilled, blasted, and moved with a large electric shovel and large trucks. Stockton-Lewiston is 80 to 100 feet beneath the Fiveblock seam. The overburden is removed by blasting, and it is relocated by an electric dragline with a 72-cu yd bucket. A mixture of acid and neutral to alkaline mudstone, shale and sandstone occurs between the coal seams. Hobet Mining, Inc. uses the neutral to alkaline mudstone, shale and sandstone as a topsoil substitute (Emerson, 1988).

## Materials and Methods

Three representative areas of the surface mine were subdivided into four plots, each 34 inches x 12 feet, for a total of 12 plots. In 1985 the plot areas were fertilized with 800 lb/acre of 15-30-15. Wood cellulose mulch was applied at 900 lb/acre. No lime was applied. Prior to seeding, the seedbed was prepared by a Rome disk-harrow with 32-inch disk blades pulled by a bulldozer. The areas were hydroseeded in the spring of 1985 with 27 lb/acre of mammoth red clover (Trifolium pratense L.), 27 lb/acre of birdsfoot trefoil (Lotus corniculatus L.), 27 lb/acre of vernal and buffalo alfalfa (Medicago sativa L.), 54 lb/acre of orchardgrass (Dactylis glomerata L.), and 18 lb/acre of timothy (Phleum pratense L.).

At the time of the first harvest the species composition on area 1 was birdsfoot trefoil, red clover, alfalfa, orchardgrass and KY-31 tall fescue (Festuca arundinacea Schreb.). Very little timothy was present. Area 2 consisted primarily of alfalfa, orchardgrass and KY-31 tall fescue. Vegetation on area 3 was similar to area 1. Although KY-31 tall fescue was not seeded on the plots, it was seeded on adjacent steeper areas of the minesoil and had spread to the research plots.

All plots were undisturbed until the first harvest. In 1987, 1988, and 1989 three cuttings per year were taken: late May or early June, mid to late July, and early to mid September. All plots were harvested with a 34-inch wide sickle bar mower. Oven-dried weights of all vegetation were recorded.

Maintenance fertilizer was applied to all plots at the rate of 280 lb/acre of 0-46-60 in September 1987 and 1988, and 500 lb/acre of 10-20-20 in August 1989.

The minesoils on the study site were mapped according to National Cooperative Soil Survey standards. One soil pit was dug, and the minesoil was described and classified (Soil Survey Staff, 1975). This pit represented the minesoils at all three plot areas. Each horizon was sampled and analyzed for standard physical and chemical properties. Total sulfur was determined by an automated sulfur analyzer for samples crushed to pass a 60-mesh screen. All other properties were determined by methods outlined by the Soil Survey Staff (1984).

## Results and Discussion

### Minesoil Properties

The minesoil on the study site was mapped as Kaymine very channery loam, 3 to 8 percent slopes, stony. The soil surface was covered with 0.01 to 1.0 percent stones 3 to 24 inches in diameter and a few boulders larger than 24 inches. The Kaymine series is classified as loamy-skeletal, mixed, nonacid, mesic Typic Udorthents. The minesoil profile described on the site (Table 1) is representative of the series. This profile was described on an area contiguous to the plot areas, but on a steeper slope. The major difference between this profile description and the official series description (OSD) is the presence of an AC horizon which was not described in the OSD. Although there may be minor differences between the minesoil profile description and the minesoils on the plots, both fit within the range of characteristics of the Kaymine OSD.

The parent material of the official Kaymine series consists of a mixture of mudstone, shale, sandstone, and frequently small amounts of coal wasted during the mining process. Rock fragments for the particle-size control section (depth of 10 to 40 inches) for the series range from 35 to 80 percent by volume. In the control section of the profile at the study site the rock fragments ranged from 60 to 75 percent by volume (Table 1). The surface horizon had 60 percent rock fragments. Dominant textures of the official Kaymine series are loam and silt loam,

Table 1. Kaymine soil profile description.

Classification: loamy-skeletal, mixed, nonacid, mesic Typic Udorthents
Pedon No.: S-86-WV-005-12-(1-4)
Location: Boone County, Hobet Mining surface mine 1.7 miles west of main office complex in Pit #4.
Vegetation: Birdsfoot trefoil, orchard grass, KY-31 tall fescue.
Parent Material: Surface mine spoil of mudstone, sandstone and coal from mining between No. 5 Block and Upper Stockton coal seams.
Physiography: Side slope of ridgetop removal.
Slope: 52 percent
Drainage: Well Drained
Elevation: Approximately 1240 feet.
Geology: Upper Kanawha formation and lower Allegheny formation
Aspect: Northwest
Described by: Wolf, Sencindiver, Kingsbury
Sampled by: Wolf, Sencindiver, Kingsbury
Date sampled: 8-6-86
USGS Quad Coord.: 81° 52' 55'' W; 38° 06' 30'' N. Mud Quad

A --0 to 7 inches. Dark gray (10YR 4/2) very channery silt loam; weak fine granular structure; very friable; many very fine and fine roots; many fine vesicular pores; 60 percent stones, channers and boulders (50 percent sandstone, 40 percent mudstone and 10 percent coal); neutral; clear smooth boundary.

AC--7 to 14 inches. Dark gray (10YR 4/1) extremely channery loam; weak fine subangular blocky structure; friable; common very fine and fine roots; many fine vesicular pores; 65 percent stones, channers and boulders (45 percent sandstone, 45 percent mudstone and 10 percent coal); mildly alkaline; clear wavy boundary.

C1--14 to 23 inches. Dark gray (10YR 4/1) very channery loam; massive; firm; few very fine and fine roots; common fine vesicular pores; 60 percent stones, channers and boulders (45 percent sandstone, 40 percent mudstone and 10 percent coal); neutral; gradual wavy boundary.

C2--23 to 65 inches. Dark gray (10YR 4/1) extremely channery loam; massive; firm; 75 percent stones, channers and boulders (50 percent sandstone, 40 percent mudstone and 10 percent coal); neutral.

which were the textures of the minesoil at the study site.

Clay percentages of minesoil horizons on the study site ranged from approximately 22 percent to 24 percent (Table 2). For the official Kaymine series, clay ranges from 18 to 27 percent. Rock fragment (> 2 mm) percentages by weight ranged from 72 to 85 in all minesoil horizons at the study site (Table 2).

Organic carbon values for all horizons at the study site were low (Table 3). Electrical conductivity and total sulfur values were relatively low. Salt levels in the minesoil were not high enough to adversely affect plant growth. All pH values were above 6.5, which is about the middle of the range of 5.6 to 7.8 for the official Kaymine series. In some horizons, base saturation is greater than 100 percent indicating some free Ca from calcium carbonates in the minesoil.

## Forage Yields

Forage yields varied over the three years with maximum yields occuring in 1987 (Figure 1). Yields were reduced in 1988 by an extreme drought, and the weakened plant condition probably contributed to lower than expected yields in 1989. Yields for the first cutting were consistently higher than yields of the other two cuttings, which is common for cool-season species. Several factors may have affected the yields. First, fertilization near the end of each growing season may have boosted spring growth. Second, more precipitation may have fallen in the spring than in the summer. This hypothesis was probably true in 1988, the very dry year, but in most years the average monthly precipitation in the summer is similar to the average monthly precipitation in the spring (Wolf, 1988). However, summer rains often come as thunderstorms, and much of the water runs off the soil surface. Fourth, deer and insect damage was noted on some plots, and this damage seemed to be worse at the second and third cuttings.

Species composition changed over

Table 2. Physical properties of Kaymine minesoil.

| Horizon | Particle Size (mm) >20 | 6-20 | 2-6 | <2 | Texture (<2mm) clay | silt | sand | Moisture Retention (15 Bars) |
|---|---|---|---|---|---|---|---|---|
| | ----------% by weight---------- | | | | | | | |
| A | 35.5 | 21.6 | 15.0 | 27.9 | 22.8 | 51.1 | 26.1 | 6.4 |
| AC | 38.3 | 23.0 | 14.0 | 24.7 | 24.4 | 48.7 | 26.9 | 6.0 |
| C1 | 29.0 | 26.7 | 16.2 | 28.1 | 21.9 | 45.6 | 32.5 | 5.7 |
| C2 | 53.6 | 16.2 | 15.3 | 14.9 | 23.1 | 45.6 | 31.3 | 5.8 |

the period of the study in areas 1 and 3 primarily because of a loss of red clover. Alfalfa, orchardgrass, and tall fescue persisted in all of the plots.

## Conclusions

Kaymine minesoil can sustain a mixed grass-legume crop under low levels of management. In this study, yields were reduced by drought, extreme heat, insects and deer, but the high pH values and potential rooting depths of greater than 60 inches in the Kaymine series are conducive to sustaining a deep-rooted legume such as alfalfa. Better production could result by soil testing for possible macro- and micro-nutrient deficiencies and by implementing a better fertility program. The major obstacle to large scale harvesting would be the surface stones which could cause equipment breakage during the first couple of cuttings until they were found and removed. During future reclamation of similar sites, mine operators should make an effort to place non-stony materials or materials with small stones at the minesoil surface.

## References


Core, E.L. 1966. Vegetation of West Virginia. McClain Printing Co., Parsons, WV 217p.

Emerson, L.D. 1988. A substitute plan for reclamation of a mountaintop removal mine in southern West Virginia. p. 274-276. In Mine Drainage and Surface Mine Reclamation. Vol. II. U.S. Bureau of Mines Information Circular 9184.

Soil Survey Staff. 1975. Soil Taxonomy: a basic system of soil classification for making and interpreting soil surveys. USDA Soil Conservation Service. Agric. Handbook No. 436.

Soil Survey Staff. 1984. Procedures for collecting soil samples and methods of analysis for soil survey. Soil Survey Investigations Report No. 1. USDA Soil Conservation Service. 5.


Table 3. Chemical properties of Kaymine minesoil.

| Horizon | pH | Total S | EC[1/] | OC[2/] | Exch. Bases Ca | Mg | K | CEC[3/] | Base Sat. |
|---|---|---|---|---|---|---|---|---|---|
| | 1:1 | % | mmho/cm | % | ----------me/100g---------- | | | | % |
| A | 6.6 | 0.37 | 2.38 | 0.97 | 8.55 | 3.14 | 0.61 | 8.77 | 140 |
| AC | 7.4 | 0.40 | 1.75 | 0.80 | 8.70 | 2.84 | 0.36 | 15.81 | 75 |
| C1 | 7.3 | 0.40 | 2.82 | 0.74 | 13.20 | 2.81 | 0.37 | 8.16 | 201 |
| C2 | 7.2 | 0.45 | 3.02 | 0.80 | 16.26 | 3.63 | 0.40 | 10.51 | 193 |

[1/] EC = Electrical conductivity of saturated extract.

[2/] OC = Organic carbon.

[3/] Cation exchange capacity by ammonium acetate saturation (Soil Survey Staff, 1984).

West Virginia University. 1971. Mine spoil potentials for water quality and controlled erosion. U.S. Environmental Protection Agency. Water Pollution Control Research Series 14010 EJE 12/71.

Wolf, B.L. 1988. Soil Survey of Wyoming County, West Virginia. USDA Soil Conservation Service in cooperation with West Virginia Agricultural and Forestry Experiment Station.

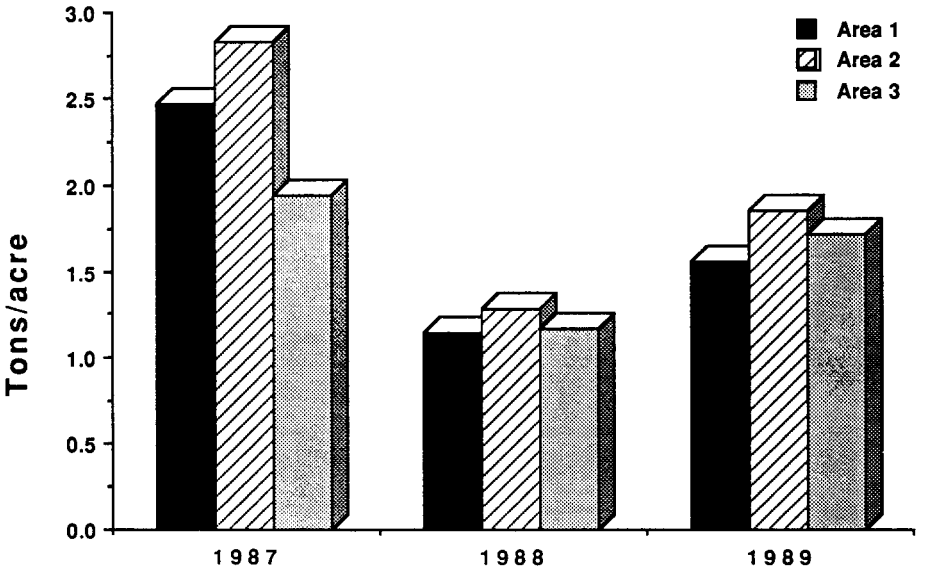



**Figure 1. Three-year summary of forage production on the Kaymine minesoil. (Each bar is the sum of 3 cuttings averaged over 4 replications.)**

This page intentionally left blank.

# ORGANIC MATTER DYNAMICS ON RECLAIMED COAL MINES IN THE CANADIAN ROCKIES [1]

P.F. Ziemkiewicz and S.K. Takyi [2]

Abstract. Development of self-sustaining plant communities on harsh reclamation sites in the Rocky Mountains has often proven difficult. This paper describes the organic matter cycles of three grass species and the influences of fertilization and topsoiling. The study took place on an abandoned coal mine on the eastern slope of the Rocky Mountains near Hinton, Alberta, Canada. The role of species selection and topsoil addition in assisting the development of a stable nutrient cycle is quantified and the effectiveness of various revegetation strategies was evaluated.

Species selection and, to a lesser extent, topsoil addition controlled reclamation success. Successful native and agronomic species were identified. Successful grasses, both native and agronomic had characteristic patterns of organic matter accumulation, retention and loss.

Additional key words. Organic matter dynamics, species selection, rocky mountains, reclamation.

## Introduction

In the Canadian Rockies revegetation of reclaimed mined land generally aims to initiate an herbaceous plant community which will be capable of supporting itself either indefinitely or until successfully colonized by the indigenous plant communities. Either way, the reclamation plant community is expected to control erosion, ameliorate site microclimatic and soil conditions, to propagate itself for some time and finally to do so without artificial assistance. Failure results in retrograde community development, bare ground, soil erosion and the liability of a second round of site reclamation. These reclaimed mine sites present a novel opportunity to characterize organic matter dynamics during primary succession.

Reclaimed mines in this region often appear successful for the first several years then deteriorate as surface cover diminishes and erosion occurs. Identification of the factors which control a plant community's persistence will shed light not only on the mechanisms for plant community and soil development but help to identify reclamation techniques which favor long-term success.

When revegetating raw spoil, fertilization is usually applied to initiate a mobile pool of plant nutrients. Also, legumes may contribute significant amounts of N to the site. Under optimal conditions the added nutrients remain mobile and maintenance fertilization can be terminated leaving a healthy plant community which slowly accumulates nutrients and ultimately assimilates with the surrounding native vegetation types. Otherwise, nutrient cycling is curtailed and the site

[1] Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2] P.F. Ziemkiewicz is a Research Associate Professor of Plant and Soil Science, West Virginia Univ. Morgantown, WV, USA 26505. S.K. Takyi is a Soil Scientist with the Forest Research Branch, Alberta Forest Service, Edmonton, AB, Canada T5K 2M4.

deteriorates.

Organic and Mineral Cycling. In cold environments decomposition of dead plant material is the major restriction on nutrient mobility. Decomposition is restricted by low temperatures, drying of surface materials, high C:N:P ratios and low decomposer populations. In-plant transfer supplies a portion of the plant's needs and may be particularly critical in the spring, releasing the plant from even greater dependence on detritus-supplied nutrients. This strategy appears to be prominent in most tundra and grassland vegetation. Plants in these areas consistently exhibit high proportions of root mass to shoot mass (Aleksandrova, 1970; Bliss, 1963; Dennis and Johnson, 1970; Rehder 1976; Thilenius, 1975).

Many reclamation studies in the Arctic, Alpine and Subalpine have reported good initial growth but poor persistence (Brown et al. 1976; Hull, 1974; Younkin, 1976). The decline in vigor is often matched by an accumulation of dead plant material on the soil surface (Younkin, 1976). Frequently, this decline in vigor coincides with the withdrawal of fertilization. While other factors undoubtedly contribute to this deterioration, mineral nutrition is critical due to the depression of decomposition and nutrient cycling in cold regions. Consequently, repeated fertilization is often required (Brown, et al., 1978).

Techniques developed in nutrient cycling studies permit isolation of the structural components within a plant community (e.g. Batzli 1974). Organic matter dynamics among the principal compartments (shoot, detritus, root) illustrate functional relationships (e.g., net production, carbohydrate partitioning, decomposition). These values comprised the dependent variables in this study. The independent variables were two of the principal treatments available to the reclamation planner: soil reconstruction and species selection.

## Study Site

In 1978 the Alberta Forest Service established a series of revegetation test plots at the Cadomin surface mine near Hinton, Alberta. The mine was abandoned in 1952. The Cadomin mine lies about 40 km south of Hinton, Alberta. The elevation of the site is 1675 m ASL, at the upper margin of subalpine plant communities. The surrounding mountains rise to the west toward the Jasper National Park boundary. The upper limit of tree growth in this area is about 1980 m above sea level. The abandoned mine remains nearly unvegetated except for isolated pockets of vegetation under the lee of several steep spoil banks (Takyi, 1985).

The site was chosen for its exceptional harshness. The spoil surface consisted of sandstone, coal, ironstone and shale fragments. The mine is subject to chinook winds throughout the winter. These winds can reach velocities in excess of 160 kph and are responsible for removing much of the winter snow cover (Root, 1976).

The experimental site was located on a series of species adaptability trials which had been established in June, 1978. These test plots were located on both east and west facing 15 degree slopes. The experimental plots had been seeded with a variety of native and agronomic grass and legume species.

More than 70% of the top 15 cm of spoil material was larger than 2 mm. The porous nature of the spoil coupled with the high prevailing wind makes it likely that drought conditions pertained at least sporadically through the growing season. On some plots the surface material contained significant coal fragments. This material is termed spoil. Non-coaly materials are termed overburden (mindful that, in fact, both materials are spoils). Topsoil, consisting of the A and B horizons of a nearby forested Entisol was applied at a thickness of 20 cm to one half of the plots. Table 1 summarizes the chemical properties of the spoil, overburden and topsoil.

Table 1. Some properties of the spoil, overburden and mineral soil used for topdressing at the Cadomin Test Site.

AGRONOMIC SPECIES PLOTS

| | SPOIL | SOIL |
|---|---|---|
| pH | 8.8 | 6.9 |
| Ec (mS/cm) | 0.3 | 0.1 |
| AVAIL. NUTRIENTS (ppm) | | |
| NH4-N | 5.6 | 4.5 |
| N03-N | 0 | 1 |
| P | 0 | 10 |
| K | 186 | 214 |
| TOTAL N (%) | 0.13 | 0.07 |
| T.E.C. (me/100 g) | 6.6 | 13.2 |

NATIVE SPECIES PLOTS

| | OVERBURDEN | SOIL |
|---|---|---|
| pH | 8.9 | 6.5 |
| Ec (mS/cm) | 0.3 | 0.1 |
| AVAIL. NUTRIENTS (ppm) | | |
| NH4-N | 3.7 | 4.1 |
| N03-N | 0 | 0 |
| P | 0 | 12 |
| K | 235 | 219 |
| TOTAL N (%) | 0.03 | 0.07 |
| T.E.C. (me/100 g) | 6.4 | 12.7 |

By 1982 the Alberta Forest Service had maintained and monitored the agronomic and maintenance fertilization plots for four years while the native species plots had been maintained for three years. Results of these studies are reported in Takyi and

Islam, 1985 and Takyi, 1985. It was agreed that, in addition to their intended purpose, the plots could supply valuable information about the influence of topsoil and species selection on development of stable, self sufficient plant communities.

## Methods

Two of the experiments were selected for this study:

1. Agronomic Species Adaptability,

2. Native Species Adaptability

Duplicate experiments had been established on both topsoil and spoil. The Agronomic Species Adaptability experiment was established on coaly spoil while the Native Species Adaptability experiment had been established on non coal spoil.

The individual Agronomic Species Adaptability plots measured 2.5 m by 6 m. A number of commercially available grasses and legumes were seeded at a rate of 60 kg/ha in June 1978. Immediately after seeding the plots were fertilized with 60 kg/ha N, 30 kg/ha P and 62 kg/ha K. The plots received the same rate of maintenance fertilization until the 1981 growing season. Two species were chosen for this project: red fescue (Festuca rubra L.) and crested wheatgrass (Agropyron cristatum (L.) (Gaertn.).

Red fescue was chosen because it is consistently the most successful agronomic grass for revegetation in the mountains of Western Canada and crested wheatgrass was chosen because it is consistently unsuccessful in the mountains. Success is defined as the ability of a species to become established, to persist and to colonize new sites. While crested wheatgrass generally establishes well, individual plants begin dying off within a few years. It almost never reseeds a second generation of individuals in the mountains. The comparison of red fescue's and crested wheatgrass's carbon budgets should shed some light on successful versus unsuccessful allocation strategies.

The Native Species Adaptability Trial was duplicated on spoil and on spoil capped with 20 cm of the same mineral soil used in the Agronomic Species Adaptability plots. Each species was seeded to provide 5000 viable seeds per square meter. All plots were fertilized by hand to provide 71-50-65 kg/ha of N, P and K, respectively. The plots received maintenance rates of fertilization up to and including the 1981 growing season. Individual plots measured 2 m X 2 m.

The native Slender Wheatgrass (Agropyron caninum L.) was also selected for the study. It was a variety which had been collected near the study site and propagated at the University of Alberta by R. Sadisivaiah and D.G. Walker. It is a common component of native subalpine grasslands in the region.

Sampling. The test plots were sampled to permit partitioning of the organic component into shoot, root and detritus compartments. This allowed examination of the major pathways of intra and extra-plant organic matter accumulation, storage and loss. Both shoot and detritus samples were obtained by clipping. The shoot compartment contained all above-ground living tissues. The detritus sample included dead standing and fallen shoot materials as well as other surface organic matter longer than 5 mm. Root samples were obtained to a maximum depth of 15 cm from a soil core sampler. Roots were separated from soil by immersion of the entire sample in a beaker of water, stirring and repeatedly removing the floating roots. Soil materials on the study plots, even the "topsoiled" plots were nearly devoid of organic matter prior to the study. So, separation of root material from soil organic matter was not a problem and virtually complete root recovery was possible. Core depth was noted for each sample. All plant and detritus samples were dried for 48 hours at 50C immediately after collection. Root samples were additionally separated then dried and massed.

This study required destructive sampling over a 14-month period on small plots. It imposed several constraints on sampling. Primarily, both clipping samples and root cores had to be small to avoid damaging the plots during the study yet sufficiently intense to yield reasonably precise estimates of means.

The plots were sampled on six dates: 2 June 1988, 7 July 1982, 9 Aug. 1982, 20 Sept. 1982, 9 June 1983 and 17 Aug. 1983. Five shoot and detritus samples were clipped each within a 30 X 15 cm frame and five 8 cm diameter root cores were taken and processed. From the smaller Native Species plots only three samples were taken. This level of sampling yielded estimates whose standard deviations of the mean to mean ratio ranged from 20 to nearly 100%. The process of collecting a complete set of samples took two days.

Calculation of Organic Matter Transfers. Within each plant community changes in organic matter within compartments were estimated by adding gains and losses between each of the sample dates. In characterizing this organic matter flux three parameters were of interest: Percent Retention, Percent Total Transfer Per Compartment and Total Transfer. The following equations define the derivation of each of the parameters of interest:

$$\text{Total Transfer} = Tt = \sum_{i=1}^{c} \sum_{j=1}^{ni} |\Delta i|j$$

$$\text{Compartmental Transfer} = Tc = \sum_{j=1}^{n} |\Delta|j$$

$$\text{Net Change} = Nc = \sum_{j=1}^{n} \Delta j$$

$$\text{\% Total Transfer per compartment} = \frac{Tc}{Tt} \times 100$$

$$\text{\% Retention} = \frac{Nc}{Tc} \times 100$$

where:

T = Organic matter transfer (Flux) in g/m2/year
N = Net change in organic matter level
t = total ( e.g. all compartments)
c = compartment (e.g. shoot, detritus, root)
n = sampling intervals

## Results and Discussion

The following discussion focusses on the changes during the period 9 August 1982 and 17 August 1983 in each organic matter compartment and on estimates of the total organic matter flux per compartment and per species.

### Performance of Various Grasses.

Slender Wheatgrass. At the end of the first growing season after cessation of maintenance fertilization (August 1982) the standing crop was evenly distributed between the detritus and root compartments (between 44 and 50%). About 7% of the standing crop was in the shoots (Table 2). Within a year, however, the ratio had shifted strongly toward the roots (about 80%) while detritus contained between 20 and 14% and the shoots contained 5% or less of the standing crop.

This shift of organic matter toward the root system was reflected in organic matter flux during the August 1982 to August 1983 period. Table 2 illustrates that the bulk of organic matter transfers occurred in the roots and to a much lesser extent in the detritus and shoots.

Topsoil apparently enhanced the proportion of standing crop and organic matter flux through the root compartment. Slender Wheatgrass communities on both topsoil and spoil increased their standing crops after withdrawal of maintenance fertilization. Almost all of the gain was in root mass (Table 3). Root gains were more pronounced on the topsoiled plots.

Red Fescue. At the end of both growing seasons standing crop was strongly concentrated in root mass (about 80%) while detritus and shoots recorded about 15 and 5% respectively (Table 2). Total standing crop was nearly double that of the Slender Wheatgrass. Most of the difference was in the root masses. Again, organic matter flux through various compartments reflected the distribution of standing crop. Ratios of root, detritus and shoot flux were around 75, 19, and 7%, respectively.

Topsoil had little effect on the proportion of standing crop and organic matter flux in the three compartments.

Red Fescue communities on the topsoiled plots lost some standing crop particularly the shoots and roots (Table 3). The Red Fescue plots on spoil showed a gain in the detritus and particularly the root compartments though the total mass was still less than on the topsoiled plots.

Crested Wheatgrass. At the beginning of the study the ratios of standing crop in the root, detritus and shoot compartments were similar to those of Slender Wheatgrass and Red Fescue at the end of the study: 78-13-9 on the topsoil plot and 63-23-15 on the non-topsoiled plot. However, by the end of the study the ratios had reverted to a nearly equal distribution between the roots and detritus (about 45% each) with the remainder in the shoot compartment (Table 2). Total community standing crop by August 1983 had fallen sharply to levels well below those of Slender Wheatgrass and Red Fescue with most of the loss in the root system.

Organic matter flux reflected the initial standing crop levels. However, most of the changes were negative (Table 3). Only the detritus compartment showed an increase.

The rate of organic matter loss was greatest on the topsoiled plots, though by August 1983 it still contained more organic matter than the spoil plot.

Organic Matter Retention. Organic matter retention is expressed as a ratio of net annual change to the annual flux through a compartment. It represents a significance index for gains and losses in each compartment. For example, red fescue on topsoil and crested wheatgrass without topsoil lost 178 and 180 g/m2 of root mass respectively (Table 3). For the red fescue, this represented only 9% of the organic matter which moved through the root compartment during the year. However, for crested wheatgrass it indicates that 26% of the organic matter which moved through the roots was lost. It is unlikely that losses of this magnitude can be maintained for more than a few years.

TABLE 2. Distribution of organic matter within shoot, detritus and root compartments in August 1982 and August 1983, and organic matter flux between these dates.

| SLENDER WHEATGRASS TOPSOIL | STANDING CROP ($G/M^2$) | | | | FLUX ($G/M^2/YR$) | |
|---|---|---|---|---|---|---|
| | AUG 82 | % TOTAL | AUG 83 | % TOTAL | AUG-AUG | % TOTAL |
| SHOOT | 39 | 7 | 41 | 4 | 68 | 6 |
| DETRITUS | 270 | 49 | 142 | 14 | 128 | 11 |
| ROOT | 247 | 44 | 824 | 82 | 939 | 83 |
| TOTAL | 556 | 100 | 1007 | 100 | 1135 | 100 |
| SLENDER WHEATGRASS NO TOPSOIL | | | | | | |
| | AUG 82 | % TOTAL | AUG 83 | % TOTAL | AUG-AUG | % TOTAL |
| SHOOT | 27 | 6 | 2 | 0 | 45 | 6 |
| DETRITUS | 216 | 45 | 202 | 21 | 266 | 33 |
| ROOT | 242 | 50 | 740 | 78 | 498 | 62 |
| TOTAL | 485 | 100 | 944 | 100 | 809 | 100 |
| RED FESCUE TOPSOIL | | | | | | |
| | AUG 82 | % TOTAL | AUG 83 | % TOTAL | AUG-AUG | % TOTAL |
| SHOOT | 172 | 8 | 109 | 5 | 259 | 9 |
| DETRITUS | 252 | 11 | 297 | 15 | 609 | 21 |
| ROOT | 1806 | 81 | 1628 | 80 | 2092 | 71 |
| TOTAL | 2230 | 100 | 2034 | 100 | 2960 | 100 |
| RED FESCUE NO TOPSOIL | | | | | | |
| | AUG 82 | % TOTAL | AUG 83 | % TOTAL | AUG-AUG | % TOTAL |
| SHOOT | 80 | 7 | 4 | 0 | 120 | 6 |
| DETRITUS | 177 | 16 | 286 | 17 | 351 | 17 |
| ROOT | 835 | 76 | 1418 | 83 | 1631 | 78 |
| TOTAL | 1092 | 100 | 1708 | 100 | 2102 | 100 |
| CRESTED WHEATGRASS TOPSOIL | | | | | | |
| | AUG 82 | % TOTAL | AUG 83 | % TOTAL | AUG-AUG | % TOTAL |
| SHOOT | 182 | 9 | 42 | 5 | 214 | 10 |
| DETRITUS | 264 | 13 | 376 | 48 | 130 | 6 |
| ROOT | 1603 | 78 | 365 | 47 | 1712 | 83 |
| TOTAL | 2049 | 100 | 783 | 100 | 2056 | 100 |
| CRESTED WHEATGRASS NO TOPSOIL | | | | | | |
| | AUG 82 | % TOTAL | AUG 83 | % TOTAL | AUG-AUG | % TOTAL |
| SHOOT | 94 | 15 | 64 | 12 | 150 | 14 |
| DETRITUS | 146 | 23 | 228 | 44 | 222 | 21 |
| ROOT | 407 | 63 | 227 | 44 | 698 | 65 |
| TOTAL | 647 | 100 | 519 | 100 | 1070 | 100 |

TABLE 3. Net change in organic matter standing crop and per cent retention for the shoot, detritus and root compartments.

| SLENDER WHEATGRASS TOPSOIL | NET CHANGE ($G/M^2/YR$) | % RETENTION |
|---|---|---|
| SHOOT | 2 | 3 |
| DETRITUS | -128 | -100 |
| ROOT | 577 | 61 |
| TOTAL | 451 | 40 |
| SLENDER WHEATGRASS NO TOPSOIL | | |
| SHOOT | -25 | -56 |
| DETRITUS | -14 | -5 |
| ROOT | 498 | 100 |
| TOTAL | 459 | 57 |
| RED FESCUE TOPSOIL | | |
| SHOOT | -63 | -24 |
| DETRITUS | 45 | 7 |
| ROOT | -178 | -9 |
| TOTAL | -196 | -7 |
| RED FESCUE NO TOPSOIL | | |
| SHOOT | -76 | -63 |
| DETRITUS | 109 | 31 |
| ROOT | 583 | 36 |
| TOTAL | 616 | 29 |
| CRESTED WHEATGRASS TOPSOIL | | |
| SHOOT | -140 | -65 |
| DETRITUS | 112 | 86 |
| ROOT | -1238 | -72 |
| TOTAL | -1266 | -62 |
| CRESTED WHEATGRASS NO TOPSOIL | | |
| SHOOT | -30 | -20 |
| DETRITUS | 82 | 37 |
| ROOT | -180 | -26 |
| TOTAL | -128 | -12 |

Even more severe losses were noted on the topsoiled crested wheatgrass plot, where root retention was -72%. Root retention on the non-topsoiled red fescue plot and both slender wheatgrass plots were strongly positive ranging from 31 to 100%.

Detritus retention was positive for slender wheatgrass and negative for both red fescue and crested wheatgrass. It tended to vary inversely with shoot retention of the latter two grasses. Detritus should be retained at a high rate in the first year after terminating maintenance fertilization as the excess shoot mass begins to decompose. A healthy decomposer population is evident on the topsoiled red fescue plot where despite high inputs from the shoots (-24% shoot retention) only 7% of the detrital flux was retained.

Finally, total organic matter retention indicates the proportional change in the plot's organic matter pool. There was a slight negative retention on the topsoiled red fescue plot and strongly positive retention values for the other red fescue and both slender wheatgrass plots. The topsoiled crested wheatgrass plot lost 62% of the organic matter which cycled through it while the non-topsoiled plot lost 12%.

In summary, positive organic matter retention values indicate that the plant community is developing and successfully colonizing the site. Under the stress of termination of maintenance fertilization such development is regarded as an indication that, at least for the short term, the plants will continue to develop.

Structure and Adaptability to the Subalpine.

The results suggest plant structural adaptations which explain the observed fitness of some grasses to harsh subalpine reclamation environments. The successful grasses in this study developed an organic matter cycle in which most of the activity was below ground. All of these species seem to have a limited ability to retain structure and, by inference, carbohydrates and nutrients. Their root systems expand and contract rapidly, releasing their nutrients to the surrounding soil. In this sort of open cycle there appears to be an advantage in the capability of the root system to recapture these externally stored nutrients. Slender wheatgrass and red fescue evidently have this ability.

Species such as crested wheatgrass which expend a disproportionate amount of resources generating shoot material appear to allocate photosynthate to a compartment which cannot be supported. Thus, the tendency to accumulate detritus until shoot production fails.

Observations.

Mid August is the end of the growing season in this region. Light snow is common by this time and the native plant communities have already set seed and are approaching toward winter dormancy. By the end of the study (17 August 1983) the Slender Wheatgrass and Red Fescue appeared slightly chlorotic and the reddish cast to the leaves suggested P deficiency. However, during the growing season the plants appeared vigorous. Both species had flowered and set seed.

Crested Wheatgrass appeared unhealthy from the beginning of the 1983 growing season. This was the first growing season without maintenance fertilization. The plants failed to flower, aerial cover appeared to diminish and the plants seemed to generate an unusual amount of detritus. In short, they looked unhealthy.

## Summary

A methodology was developed in this study whereby organic matter dynamics in the shoot, detritus and root compartments of grass stands were monitored and evaluated over a one year period. This approach provides insight into patterns which can predict and explain the long-term performance of grass species in harsh subalpine reclamation environments. These single species test plots were all subjected to the same experimental perturbation: withdrawal of maintenance fertilization. The following summarizes how each species responded to this perturbation.

While each grass species was affected differently by withdrawal, each supported less shoot mass by mid-August. The only exception was the slight shoot increase in slender wheatgrass grown on topsoil. Otherwise, it appears that upon withdrawal of maintenance fertilization the shoots enter fall dormancy sooner.

Slender wheatgrass and red fescue roots developed strongly during the study. The only exception was a slight decrease in the topsoiled red fescue plot. The total organic matter pools of both slender wheatgrass and red fescue also increased. Root mass and total organic matter of both crested wheatgrass plots declined sharply.

The behavior of the topsoiled red fescue plot was unique. Its organic matter pool was stable throughout the study, losing only 7% of its total flux. Except for the large drop in shoot mass the other compartments changed little relative to their initial mass. This suggests that this plot had reached a level of "maturity" and was little affected by the withdrawal of maintenance fertilization. The other red fescue plot and both slender wheatgrass plots developed rapidly after withdrawal suggesting that fertilization may have been an impediment to root development, possibly through overstimulation of shoot production.

Crested wheatgrass, however, was clearly dependent on maintenance fertilization. After its withdrawal both topsoiled and non-topsoiled plots declined in all parameters except detritus.

## Literature Cited


Aleksandrova, V.D. 1970. Above-ground and below-ground plant biomass of different Arctic subzone assns. In . E.E. Ershov (ed.) Biological Basis of the Utilization of Northern Nature. pp. 13-19 SSR Academy of Science.

Batzli, G.O. 1974. Production, assimilation and accumulation of organic matter in ecosystems. J. Theor. Bio. 43-1-13.

Bliss, L.C. 1963. Alpine plant communities of the Presidential Range, New Hampshire, Ecology 44: 678-697.

Brown, R.W., Johnston, R.S. and VanCleve K. 1978. Rehabilitation problems in Arctic and Alpine Regions. In. Reclamation of Drastically Disturbed Lands. American Society of Agronomy, Madison, Wisc. pp. 23-44.

Brown, R.W., Johnston, R.S., Richardson, B.Z. and Farmer, E.E. 1976. Rehabilitation of Alpine disturbances: Beartooth Plateau, Montana. In. Proc., Workshop in Revegetation of High Altitude Disturbed Lands. Info. Series 21. Environ. Resources Center, Colo. State Univ., Ft. Collins. pp. 58-73.

Dennis, J.G. and Johnson, P.L. 1970. Shoot and rhizome-root standing crops at Barrow, Alaska. Arctic and Alpine Research 2:253-266.

Hull, A.C. 1973. Duration of seeded stands on terraced mountain lands, Davis County, Utah. J. Range Manage 26:133-136.

Hull, A.C. 1974. Species for Seeding Mountain Rangelands in Southeastern Idaho, Northeastern Utah and Western Wyoming. Journal of Range Management 27(2):150-1-53.

Laycock, W.A. 1982. Seeding and fertilizing to improve high-elevation rangelands. USDA For. Serv. Gen. Tech. Rep. INT-120. 19 P.

Plummer, A.P., Christensen, D.R. and Monsen, S.B. 1968. Restoring Big-Game Range in Utah. Publication No. 68-3 Utah Division of Fish and Game. 183 pp.

Rehder, H. 1976. Nutrient turnover studies in Alpine ecosystems, I. Phytomass and nutrient relations in four mat communities of the northern Calcareous Alps. Oecologia (Berl.) 22:411-423.

Root, J.D. 1976. Physical environment of an abandoned strip mine near Cadomin, Alberta. University of Alberta, Alberta Research Council, Edmonton, Alberta. Bulletin No. 34. 33 pp.

Takyi, S.K. 1985. Plant cover establishment methods on a severe Rocky Mountain Foothills Site. In. Ziemkiewicz, P.F. 1985 (ed.) Revegetation Methods for Alberta's Mountains and Foothills. Alberta Land Conservation and Reclamation Council Report #RRTAC 85-1 pp 112-144.

Takyi, S.K. and Islam, R. 1985. Maintenance Fertilization of revegetated subalpine coal mine spoil. In. Ziemkiewicz, P.F. 1985 (ed.) Revegetation Methods for Alberta's Mountains and Foothills. Alberta Land Conservation and Reclamation Council Report #RRTAC 85-1 pp 78-89.

Thilenius, J.F. 1975. Plant production of three high-elevation ecosystems. In. Final Report: Medicine Bow Ecology Project. USDI Bur. Recl. pp. 60-75.

Younkin, W.E. 1976. (Ed.) Revegetation Studies in the Northern MacKenzie Valley Region. Arctic Gas Biophysical Report Series 38:4-20.

Ziemkiewicz, P.F. 1985. Revegetation of Oil and Gas Disturbances: Seed Mixes, Fertilizers and Application Rates. In. Ziemkiewicz, P.F. 1985 (ed.) Revegetation Methods for Alberta's Mountains and Foothills. Alberta Land Conservation and Reclamation Council Report #RRTAC 85-1 pp 78-89.

This page intentionally left blank.

# EVALUATION OF MINESOIL QUALITY AND CAPABILITY IN ALBERTA, CANADA[1]

by

T.M. Macyk[2]

Abstract. The objective of surface mine reclamation in the Plains Region of Alberta is to produce a landscape that has a capability for agricultural production that is equivalent to or greater than that which existed prior to mining. The process of adopting and implementing a system for measurement of reclamation success was and continues to be an evolutionary one. One of the first stages in the process was the development of soil quality criteria which were prepared to provide a basis for evaluating soils prior to disturbance and subsequent to reclamation. Criteria were outlined for soil survey and sampling programs and analytical requirements associated with baseline and post-disturbance characterization. Criteria for evaluating the suitability or quality of undisturbed and reconstructed soils were developed based on a broad range of physical and chemical parameters. Concurrent with the establishment of these criteria, a capability rating system for reconstructed soils was developed by adapting the existing Canada Land Inventory soil capability for agriculture rating system. Subsequent to this a land capability classification for arable agriculture in Alberta was developed which retains a close similarity to the previously developed systems but tends to be more quantitative. The system is based on seven classes and twenty-one subclasses or types of limitation.

## Introduction

The objective of reclamation of surface mines in the Plains Region of Alberta is to produce a landscape that has a capability for agricultural production that is equivalent to or greater than that which existed prior to mining. Put in economic terms, it is clearly the intent of reclamation legislation in Alberta that the cost of assuring agricultural productivity in the post-mining landscape is to be borne as a capital investment in the land rather than as an operating cost by the farmer. The process of adopting and implementing a system for measuring reclamation success was and continues to be evolutionary and is based on several strategic activities and research efforts. This paper describes these strategic activities and summarizes some of the documents which resulted from the work conducted.

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]T.M. Macyk is Senior Research Officer, Terrain Sciences Department, Alberta Research Council, Edmonton, Alberta, Canada.

## Methods

### Development of Soil Quality Criteria

Surface mining or any land disturbances arising from resource extraction and transport are intended to be only temporary disruptions to the normal use of land. Although no two sites are absolutely identical, disturbances cause similar types of problems and concerns but in varying degrees of intensity. Assurance that such disturbances are temporary is possible only if information concerning a specific site is well documented and an appropriate reclamation program is planned prior to disturbance. Evaluation of the nature of the materials at hand prior to disturbance and subsequent to reclamation, however, requires criteria by which to assess the quality of those materials. To this end a Soil Quality Criteria Subcommittee of the Alberta Soils Advisory Committee was formed in 1978 to develop criteria relative to:

a) Soil mapping and sampling for baseline and post disturbance activity;

b) Overburden sampling;

c) Analytical requirements;

d) Physical, chemical and biological criteria for evaluating the suitability of soil materials for revegetation.

The first step in the process was to divide the province into three distinct regions to allow for the establishment of criteria that would apply to each region in general. There are differences within each of these regions but it was beyond the scope of the document (Alberta Soils Advisory Committee 1987a) produced to suggest criteria for subdivisions of each of the three major regions because individual operations within each of the major zones could have unique conditions or characteristics resulting in specific problems and requirements. The three major regions are the:

a) Plains Region which includes the Central Plains and Peace River Plains and has a predominantly agricultural land use;

b) Eastern Slopes Region which includes the Lower and Upper Foothills and the Rocky Mountains to the British Columbia border; and

c) The Northern Forested Region which includes the remainder of the province.

This paper will deal with the development of criteria for the Plains Region or agricultural areas of the province.

The next step in the process of soil quality criteria development was to provide guidelines relative to soil survey activities. A soil survey with relevant interpretations helps in understanding the soil relationships in an area prior to preparing a development plan to ensure adequate evaluation of the potential for reclamation. The survey also is beneficial in planning activities such as materials handling and soil reconstruction.

The document (Alberta Soils Advisory Committee 1987a) provides guidelines relative to recommended inspection density, soil profile characteristics, landscape features to be recorded, and map presentation. Use of the photo mosaic base is recommended in part because it is particularly helpful in working with post disturbance landscapes. Guidelines pertinent to sampling for baseline or evaluation purposes including sampling intensity and sampling methodology are described. Similarly, guidelines pertinent to post-disturbance or reconstructed soil sampling are provided. Again, this includes guidelines regarding sampling intensity and sampling methodology. For example, the document states that "sampling of reconstructed soils should be done on the basis of layers or materials such as topsoil, subsoil and spoil and on depth intervals within each of these discrete layers".

Analytical requirements are defined for both baseline characterization and post-disturbance or reconstructed areas. The minimum analytical requirements listed aid in properly characterizing soils for classification and mapping purposes and making interpretations relative to the quality of the soils as they occur in the undisturbed and reconstructed states. Preferred methods of analysis are described.

Soil suitability evaluations are made by considering the interaction of various soil properties and characteristics to give an overall rating of the degree of suitability. Three categories of suitability and one category to indicate unsuitability are used. The four categories are as follow:

a) Good (G) - None to slight soil limitations that affect use as a plant growth medium;

b) Fair (F) - Moderate soil limitations that affect use but which can be overcome by proper planning and good management;

c) Poor (P) - Severe soil limitations that make use questionable. This does not mean the soil cannot be used, but rather careful planning and very good management are required;

d) Unsuitable (U) - Chemical and/or physical properties of the soil are so severe reclamation would not be economically feasible or in some cases impossible.

In agricultural areas the selective salvage of topsoil and subsoil and subsequent sequential replacement of these materials is currently practiced. It is also useful to characterize the material below the subsoil in the predisturbance setting because this usually becomes the "spoil" upon which the reconstructed soils are built. In some instances these parent materials can and do become part of the reconstructed subsoil. To facilitate the identification of suitable sources of soil materials for replacement, it is recommended that the upper five metres be characterized prior to disturbance. Topsoil is defined as the surface "A" (organo-mineral) horizons of the soil profile. Subsoil is defined as the "B" horizon(s) and the upper portion of the parent material.

The criteria for evaluating the suitability of the soils for their use as topsoil and subsoil are provided in Tables 1 and 2. The ratings (good, fair, poor, unsuitable) are determined by assessing the site factors and analytical data in terms of the limits presented in the criteria tables. Each horizon or layer is rated relative to the individual parameters and an overall rating can be developed for each horizon or layer. The most limiting property (rating) determines the ultimate rating for each horizon or layer.

A number of the parameters assessed and used in developing ratings are interrelated. For example, sodicity, saturation percentage and texture are closely related. Therefore, in the event that a given soil horizon or layer has a fair rating assessed for each of these parameters and a fair or better rating for the remainder of the parameters considered, the overall rating for that horizon or layer should be fair.

It is important to note some parameters are more important than others in terms of assessing quality and there are those where management practices can overcome or compensate for some limitations. The intent of this system is not to suggest the extent to which management practice could impact ratings that are developed. For

Table 1. Criteria for evaluating suitability of topsoil in the Plains Region.

| Rating/Property | Good (G) | Fair (F) | Poor (P) | Unsuitable (U) |
|---|---|---|---|---|
| Reaction (pH) | 6.5 to 7.5 | 5.5 to 6.4 & 7.6 to 8.4 | 4.5 to 5.4 & 8.5 to 9.0 | <4.5 and >9.0 |
| Salinity (EC) (dS/m) | <2 | 2 to 4 | 4 to 8 | >8 |
| Sodicity (SAR) | <4 | 4 to 8 | 8 to 12 | >12[1] |
| Saturation (%) | 30 to 60 | 20 to 30, 60 to 80 | 15 to 20, 80 to 120 | <15 and >120 |
| Stoniness Class | S0, S1 | S2 | S3, S4 | S5 |
| Texture | FSL, VFSL, L, SL, SiL | CL, SCL, SiCL | LS, SiC, C[2], S, HC[3] | |
| Moist Consistency | very friable, friable | loose | firm, very firm | extremely firm |
| Organic Carbon (%) | >2 | 1 to 2 | <1 | |
| $CaCO_3$ Equivalent (%) | <2 | 2 to 20 | 20 to 70 | >70 |

[1]Materials characterized by an SAR of 12 to 20 may be rated as poor if texture is sandy loam or coarser and saturation % is less than 100.

[2]C - May be upgraded to fair or good in some arid areas.

[3]HC (Heavy Clay) - May be upgraded to fair or good in some arid areas.

Table 2. Criteria for evaluating suitability of subsoil material in the Plains Region.

| Rating/Property | Good (G) | Fair (F) | Poor (P) | Unsuitable (U) |
|---|---|---|---|---|
| Reaction (pH) | 6.5 to 7.5 | 5.5 to 6.4 & 7.6 to 8.5 | 4.5 to 5.4 & 8.6 to 9.0 | <4.5 and >9.0 |
| Salinity (EC) (dS/m) | <3 | 3 to 5 | 5 to 10 | >10 |
| Sodicity (SAR) | <4 | 4 to 8 | 8 to 12 | >12[1] |
| Saturation (%) | 30 to 60 | 20 to 30, 60 to 80 | 15 to 20, 80 to 120 | <15 and >120 |
| Stone Content (% Vol) | <3 | 3 to 25 | 25 to 50 | >50 |
| Texture | FSL, VFSL, L, SiL, SL | CL, SCL, SiCL | S, LS, SiC, C, HCL[2] | Bedrock |
| Moisture Consistency | very friable, friable | loose, firm | very firm | extremely firm |
| Gypsum<br>$CaCO_3$ Equivalent (%) | The suitability criteria for sodicity (SAR) may be altered by the presence of high levels of either lime ($CaCO_3$) or gypsum ($CaSO_4$) in excess of other soluble salts. | | | |

[1]Materials characterized by an SAR of 12 to 20 may be rated as poor if texture is sandy loam or coarser and saturation % is less than 100.

[2]HCL - Heavy Clay Loam.

example, an unmined or reconstructed soil could be rated fair, poor, or unsuitable on the basis of degree of stoniness while the remaining parameters considered are not limiting. In this instance the rating should be qualified with a statement to the effect that management practice (stone picking) could be used to result in a better soil material.

## Discussion

### Soil Capability vs Soil Productivity

The coal development policy for Alberta (1976) stated that "the primary objective in land reclamation is to ensure that the mined or disturbed land will be returned to a state which will support plant and animal life or be otherwise productive or useful to man at least to the degree it was before it was disturbed. In many instances the land can be reclaimed to make it more productive, useful, or desirable than it was in its original state; every effort will be made towards this end".

In 1979 the Alberta Research Council initiated the Plains Hydrology and Reclamation Project (PHRP) which involved a holistic approach to reclamation by integrating studies of geology, hydrogeology and soils in a mining area and in the adjoining unmined areas. One of the subobjectives of this study was to evaluate the productivity potential of post-mining landscapes and the significance of changes in productivity as a result of mining (Moran et al. 1981). During the proposal stage of the project it was felt that the concept of productivity was one that could be utilized in comparing pre-mining and post-mining situations. Although initially it appeared feasible to measure productivity, the approach was modified to include the characterization of reconstructed soils and to determine their suitabilities and limitations for agricultural production.

Capability for agriculture was chosen as the basis for evaluating the product of reclamation rather than productivity primarily because capability considers intrinsic properties of the landscape. Productivity, on the other hand, addresses a parameter that is very much subject to alteration by management practices. In simple terms, a given level of productivity can be achieved from either good land with minimal management input or poorer land with greater management input. The significance of this is that in the latter case, removal of management input results in deterioration of productivity. Therefore, using productivity as a measure of reclamation performance does not allow for separation of the relative contributions of the land itself and management inputs.

Doll and Wollenhaupt (1985) indicated that sole dependence upon vegetation establishment and production without measuring the degree of re-establishment of the root zone factors that govern root growth is neither a reliable nor a cost-effective approach to measurement of reclamation success. Smith (1983), in quoting excerpts from the USDA Soil Survey Manual described the difficulty and uncertainty of establishing precise soil productivity values. Precise statements about soil productivity must be related to a specific type of soil, a certain type of crop or combination of crops and a specific set of management practices.

### Development of a Capability Rating System for Reconstructed Soils

The development of a capability rating system for reconstructed soils was based upon the evaluation of various parameters which were compared to similar parameters of undisturbed soils (Macyk 1986). The system to be developed required a "common thread" with an existing system used to rate the capability of natural or undisturbed soils so that relevant comparisons could be made.

At the time, the system used to rate soil capability in Alberta was the Canada Land Inventory (CLI) soil capability for agriculture rating system (Canada Land Inventory 1965). The CLI soil capability system for agriculture is an interpretive grouping that can be made from soil survey information wherein mineral soils are grouped into seven classes according to their potential and limitations for agricultural use. It is comparable to the USDA-Land Capability Classification System which is based on three levels of classification including eight classes (Klingbiel and Montgomery 1961).

To ensure continuity between the existing CLI system which assessed the suitability of a soil for a given use prior to disturbance, the system or component classes associated with reconstructed soils had to reflect a respectively similar capability (Macyk 1987).

At the time that the reconstructed soil capability rating system was being developed under the auspices of the Plains Hydrology and Reclamation Project, a system was being developed to provide a single, consistent assessment of agricultural capability in the province. The systems commonly used included the Farmland Assessment Schedule of Municipal Affairs (Department of Municipal Affairs 1979), the Public Lands System (Storie 1933) and the CLI - Soil Capability for Agriculture (Brocke 1977, Canada Land Inventory 1965). This most recently developed system is based on land and environmental factors as they affect dryland agriculture including the common crops and management systems of the day (Alberta Soils Advisory Committee 1987b). It can be used to assess the agricultural capability of the post- as well as the pre-disturbed condition. The component factors are all measurable climate, soil or landscape features that affect plant growth and which are not dependent on undisturbed sites or traditional taxonomic classifications.

The latest system retains a close similarity to the previously used CLI - Soil Capability for Agriculture system but tends to be more quantitative. The classes which are the broadest category in the system provide an assessment of the degree of intensity of the limitation. A range of index points is assigned to each class. The land capability classes are defined as:

Class 1 - These lands have no significant limitations for crop production (80 to 100 index points).

Class 2 - These lands have slight limitations that restrict the range of crops or require modified management practices (60 to 79 index points).

Class 3 - These lands have moderate limitations that restrict the range of crops or require special management practices (45 to 59 index points).

Class 4 - These lands have severe limitations that restrict the range of crops that can be grown or require special management practices or both (30 to 44 index points).

Class 5 - These lands have very severe limitations for sustained arable agriculture. Annual cultivation using common cropping practices is not recommended (20 to 29 index points).

Class 6 - These lands have such severe limitations for arable agriculture that cropping is not feasible even on an occasional basis (10 to 19 index points).

Class 7 - These lands have no capability for arable agriculture (0 to 9 index points).

It is important to note that lands within a capability class are similar only with respect to the degree of intensity of limitation. Any one class may include many different kinds of soil and land characteristics which require a variety of management practices.

The second major category of the system is the subclass which describes the type of limitation responsible for the class designation. The subclass is a grouping of soils and lands with the same kind of limitation. Twenty-one different types of limitations are recognized as a result of adverse climate, soil or landscape characteristics. The subclasses are defined as follows:

Climate (C)
- moisture limiting factor (A)
- temperature limiting factor (H)

Soils (S)
- texture in mineral soils (M)
- structure and consistence in mineral

Soils (D)
- organic matter content (F)
- depth of Ah or Ap in mineral soils (E)
- acidity/alkalinity (V)
- salinity (N)
- sodicity/saturation percentage (Y)
- calcareousness (K)
- peaty surface (O)
- depth to nonconforming layer (R) (D) (M)
- drainage (W)

Organic soils (O)
- organic soil temperature (Z)
- degree of decomposition (B)
- wood content (I)
- nutrient status (G)
- salinity (N)
- depth of deposit (X)
- subsurface acidity (V)
- drainage (W)

Landscape (L)
- slope (T)
- surface stoniness (P)
- pattern (J)

Data are required for each of the factors identified within the climate, soils and landscape components when using the rating system. These data can be obtained from maps and reports and specific site inspections. The level of detail of information required is determined by the purpose of the capability rating. For example, more detailed information would be required for development of capability ratings for reconstructed soils in a portion of a surface mined area than for ratings for a municipal district or county level assessment.

Data pertinent to climatic parameters are based on long term (1951-1980) records. Soil and landscape data are more site specific than the climate requirements and should be collected in the field wherever possible. Standards established by the soils community in Canada are followed in the definition and description of individual parameters.

Climate Factors. Climate plays a major role in the definition of agricultural capability in Alberta. The indices developed for this evaluation relate to dryland agriculture, the common crops of the area and the summer growing season. The climate rating is based on two principal variables including an energy factor and a moisture factor. Four modifiers including spring moisture, fall moisture, fall frost and hail are also recognized as having an impact on the climatic assessment of agricultural capability.

The moisture component is a simplified moisture balance calculation based on monthly precipitation (P) and potential evapotranspiration (PE) (Baier and Robertson 1965). The energy component is based on an effective growing degree day (EGDD) calculation which includes length of season, degree days, day length and diurnal temperature range parameters. The start of the growing season is taken as the first occurrence of five consecutive days with mean temperatures above 5°C after March 15. The end of the growing season is represented by the average date of first occurrence of 0°C after July 15. The number of degree days over 5°C are accumulated over the season defined above.

Soil Factors. Mineral soils are defined as those which have less than 40 cm of surface peat. The rating for mineral soils includes consideration of surface features (0 to 20 cm), subsurface features (20 to 100 cm) and drainage. Emphasis is placed on the surface (0 to 20 cm) layer because it represents the portion that is managed or the portion where seed is placed, seedlings develop and the major portion of nutrients are found.

The rating of the surface represents the basic soil rating with the subsurface and drainage factors used as modifiers. Eight parameters including texture, structure and consistence, organic matter, depth of topsoil, acidity,

salinity, calcareousness and peaty surface are recognized for characterization of the surface 0 to 20 cm. Sodicity is an additional factor that is suggested for use in developing ratings for reconstructed soils.

Point deduction criteria were developed for each of the above mentioned parameters, however, space limitations allow for inclusion of examples of point deduction criteria for only a few parameters. For example, point deduction criteria for surface soil acidity and saturation percentage as they would apply to reconstructed soils are presented in Table 3.

Landscape Factors. The features associated with the landscape evaluation include topography, stoniness and landscape pattern or obstacles to farming. Topography represents the basic rating with stoniness and pattern as modifiers.

Length and steepness of slope are the controlling landscape parameters. As slope increases above 10% there are increasing machinery limitations, however the main concern is the sustainability of the productive capacity of the land.

Final Capability Rating. Determination of a final capability rating involves determination of the most limiting component and other significant components. For example, the individual component ratings and the final capability rating derived therefrom for a given parcel of land might be:

| | |
|---|---|
| Climate Rating | = 2A |
| Soil Rating | = 3M |
| Landscape Rating | = 1 |
| Final Capability Rating | = 3M |

When a rating for a parcel of reconstructed land is applied it is done so based on the properties of the reconstructed soils determined at a specific time. It is important that all users understand and accept the concept that change is likely to, and certainly will occur in these reconstructed soils and the associated appropriate capability rating may also change. The capability rating system and, therefore, any particular rating assigned as applied to reconstructed soils is based on existing conditions and not on what the conditions are perceived or predicted to be some time hence.

## Relating Productivity to Capability

The capability ratings that can be applied to reconstructed soil areas allow for an ordered ranking of relative capability; however, these ratings do not provide for quantification or allow for the detailed assessment of production capacities and for the effects of different management techniques. This can only be accomplished through the measurement of yield.

Table 3. Point deductions for surface soil sodicity and saturation percentage.

| Sodicity (SAR) | Saturation Percentage (Sat %) | Points Deducted |
|---|---|---|
| 4 | 60 | 0 |
| 8 | 80 | 10 |
| 12 | 120 | 30 |
| 16 | 160 | 50 |
| 20 | >160 | 80 |

Peters and Pettapiece (1981) indicated that there is a predictable relationship between yields of agricultural soils and CLI capability classes and subclasses within any one climatic zone. Evaluating the relationship between capability ratings and yield or productivity levels pertinent to reconstructed soils has not been completed in Alberta. Productivity or yield measurements have been made in controlled plot experiments (Graveland et al. 1988, Can-Ag Enterprises Ltd. 1987) and in uncontrolled field size measurements associated with operating mines (Logan 1988).

Controlled experiments will be required to provide conclusive results regarding the relationship between productivity and capability in reconstructed soil settings.

## Disclaimer

The recommendations and conclusions in this report are those of the author and not necessarily those of the Alberta Government or its representatives.

## Literature Cited


Alberta Soils Advisory Committee. 1987a. Soil quality criteria relative to disturbance and reclamation (revised). 56 p. Prepared by Soil Quality Criteria working Group, Soil Reclamation Subcommittee.

Alberta Soils Advisory Committee. 1987b. Land capability classification for arable agriculture in Alberta. 103 p. Prepared by Agricultural Rating System Working Group and Agrometeorology Working Group. Edited by W.W. Pettapiece.

Baier, W. and G.D. Robertson. 1965. Estimation of latent evaporation from simple weather observations. Can. J. Plant Sci. 45:276-284.

Brocke, L.K. 1977. Soil capability for agriculture in Alberta. 24 p. Prepared for Alberta Environment.

Canada Land Inventory. 1965. Soil capability classification for agriculture. The Canada Land Inventory Report No. 2. Environment Canada, Ottawa.

Can-Ag Enterprises Ltd. 1987. Battle River soil reconstruction project 1986/87 annual report. Unpublished report prepared for Plains Coal Reclamation Research Program, Land Conservation and Reclamation Council, Alberta Environment.

Department of Municipal Affairs. 1979. 51 p. Assessment Manual: Section 3, Farm Land Assessment. Government of the Province of Alberta, Edmonton.

Doll, E.C. and N.C. Wollenhaupt. 1985. Use of soil parameters in the evaluation of reclamation success in North Dakota. In Proceedings of the National Meeting of the American Society for Surface Mining and Reclamation. (Denver, Colorado, October 8-10, 1985.)

Government of Alberta. 1976. A coal development policy for Alberta. Department of Energy and Natural Resources.

Graveland, D.N., T.A. Oddie, A.E. Osborne and L.A. Panek. 1988. 102 p. Highvale soil reconstruction project: five-year summary. Alberta Conservation and Reclamation Council Report #RRTAC 88-11.

Klingbiel, A.A. and P.H. Montgomery. 1961. Land capability classification. Agriculture Handbook 210. U.S. Government Printing Office, Washington, D.C.

Logan, R.J. 1988. Reclamation of sodic coal mine spoils. In Partial Proceedings of the 3rd Annual CSSS/AIC Meeting. Energy Extraction: Concerns and Issues Related to Soil Reclamation. (Calgary, Alberta, August 21-24, 1988.)

Macyk, T.M. 1986. Characterization and variability of soils reconstructed after surface mining in Central Alberta. 146 p. Alberta Land Conservation and Reclamation Council Report RRTAC#86-2.

Macyk, T.M. 1987. An agricultural capability rating system for reconstructed soils. 46 p. Alberta Land Conservation and Reclamation Council Report #RRTAC 87-13.

Moran, S.R., R. Li, T.M. Macyk, D. Scafe, M. Trudell and E.I. Wallick. 1981. Assessment of reclamation potential and hydrologic impact of large-scale surface mining of coal in Plains areas of Alberta: Summary Report of Activities 1980-81. 177 p. Volume II.

Peters, T.W. and W.W. Pettapiece. 1981. Crop yields in Alberta. Preliminary relationships to soil capability for agriculture and soil type. Alberta Institute of Pedology Contribution No. M-81-1.

Smith, D.F. 1983. Options for evaluation of prime farmland reclamation success (Soil survey versus crop production as a measure of soil productivity). In Proceedings of the 1983 Symposium on Surface Mining, Hydrology, Sedimentology and Reclamation. (University of Kentucky, Lexington, Kentucky November 27 - December 2, 1983.)

Storie, R.G. 1933. An index for rating the agricultural value of soils. California Agriculture Experimental Bull. 556. 38 p. University of California, Berkley.

This page intentionally left blank.

# MANAGING RESTORED PRIME FARMLAND FOR CORN PRODUCTION[1]

by

R.I. Barnhisel[2] and R.B. Gray[2]

**Abstract**. In Kentucky, as well as many other states, coal companies must demonstrate the return of productivity of prime farmland by growing corn at least one of the three required years to obtain Phase III Bond Release. Achieving sufficiently high corn yields is likely the most difficult task in meeting the productivity standards. In this paper, we have presented data from large corn fields where Peabody Coal Co. has applied knowledge gained from several research projects. Key factors will be presented in association with growing corn on prime farmland as a part of the final bond release process. We have tried to use the best facets of many basic research projects in this effort, and some aspects have not previously been tested side by side under "research" conditions. The result of this endeavor was successful by meeting the corn yield levels for Phase III Bond Release on approximately 80 hectares (200 acres) of prime land in 1989.

Additional Key Words: Phase III Bond release, applied research.

## Introduction

Numerous research projects have been conducted on prime farmland, with the bulk being done in Illinois and Kentucky. In Kentucky, several research projects have been initiated with the central focus of compaction removal (Barnhisel et al., 1979; Barnhisel, 1983; Barnhisel and Powell, 1985; Barnhisel et al., 1987; Barnhisel et al., 1988; Huntington et al., 1980; Powell, et al., 1985; and Powell, et al., 1987). In one project, mining methods have been studied in which end-dump trucks versus scraper pans were used during soil replacement (Barnhisel et al., 1987). In this same study, the effect of stockpiling or double handling soil was also investigated. The effect to soil horizon replacement on row crop yields has been reported by Jansen et al. (1984).

Ripping and/or subsoiling has also been used in order to reduce the effects of compaction from heavy equipment used in soil transport (Barnhisel and Powell, 1985; Barnhisel et al., 1988; Huntington et al., 1980; Powell, et al., 1985; and Powell, et al., 1987). Comparisons are not easily made, since the operating depth and/or design of such subsoiling devices varied from experiment to experiment. We also investigated the effect of the soil moisture content on the ripping action of such equipment (Powell, et al., 1985). This latter experiment was conducted in an attempt to answer why ripping helped to increase yield in one field or set of circumstances, whereas in other cases, yields were not improved and/or even lowered.

Barnhisel et al. (1989) observed from a crop management experiment that crop

---

1 Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, WV, April 23-26, 1990.

Published with the approval of the Director of Ky. Agr. Exp. Stn., as Journal Article No. (90-4-16).

2 R.I. Barnhisel is a Professor of Agronomy and Geology, University of Kentucky, Lexington, KY 40546, and R.B. Gray is Reclamation Supervisor, Peabody Coal Co., Route #3, Box 259, Central City, KY 42330.

yields may be affected one or more years following a practice such as growing a deep-rooted crop. Combinations of deep-rooted crops and ripping were also investigated in this experiment and affect yields for at least five years after employed. Corn root development has been studied in constructed surfaced-mined land in Illinois by Fehernbacher et al. (1982).

Corn varieties were also evaluated on restored prime farmland in order to determine if a yield advantage could be achieved by simply making the correct choice of corn hybrid (Powell et al., 1988).

Although some of the above research projects are ongoing, many have been completed. In 1989, the utilization of ideas gained over the past 10 years was applied by Peabody Coal Co. to approximately 200 acres of prime farmland in an attempt to meet corn yields needed for Phase III bond release. The objective of this paper is to illustrate that productivity of disturbed prime farmland can be restored on a large scale.

## Methods and Materials

In general, soil was replaced on each of the three mines being studied by scraper pans or end-dump trucks to the depth specified in the permit. This restored depth varied somewhat on each reclamation project due to differences in thickness of the original prime land soil. In one case on the River Queen mine, the data reported was from the second attempt to meet the corn production standards, as in 1988 the corn yield was too low. The sequence of the restoration process used on the Alston mine will be described in detail, and major differences from this procedure will be pointed out for the other two mines. The starting point for corn production following soil replacement varied no more than two years.

### Alston Surface Mine.

This permit consisted of about 200 hectares (490 acres) with 30 percent being prime farmland. Prior to soil replacement, the spoil was graded to the topographic configuration approximating the original contour. In early summer of 1986, 75 cm (30 in.) of subsoil was replaced with end-dump trucks on all of the areas designated to be returned to prime farmland. After leveling the subsoil, 15-30 Mg/ha (7-14 T/Ac) of agricultural limestone was applied. The lime rate was adjusted to reflect differences in lime requirement for the different soils being replaced and was based on soil test results. The subsoil was then ripped with a Rome® "4T" ripper operated at an average depth of 75 cm. The next step was to prepare a seedbed with a heavy disk harrow and plant a grass/legume cover crop in the subsoil. The purpose of the cover crop was to increase the organic matter of the subsoil and to biologically help break up any compaction from the dozers and graders used in subsoil leveling.

The seed mixture consisted of: 45 kg sudangrass (Sorghum bicolor L. Moench.); 34 kg tall fescue (Festuca arundinacea Schrad.); 11 kg each of annual ryegrass (Lolium multiflorum Lam); alfalfa (Medicago sativa L.); and yellow blossom sweetclover (Melilotus officinalis L. Lam). These rates were applied on a hectare basis. Fertilizer was applied using 112 kg/ha of 18-46-0 and 336 kg/ha of 0-0-60.

This cover crop was disked into the subsoil surface in late May 1987, and later that summer 12 cm of topsoil was replaced with end-dump trucks. The soil was ripped with the Rome® ripper and limed at 15 Mg/ha of agricultural limestone according to soil tests. In the fall of 1987, a grass/legume (annual ryegrass/hairy vetch, Vicia villosa Roth) cover crop was seeded along with a companion crop of wheat (Triticum aestivum L.) using the variety, Pioneer® 2551.

The original plan was to plant corn (Zea mays L.) into this grass/legume/wheat cover crop in May 1988. However, due to the unusually dry fall of 1987, very little grass or legume was established, but an excellent stand of wheat was obtained. The revised plan was to attempt to meet the target yield goal for wheat, hence the wheat was topdressed with 225 kg/ha of urea (46-0-0) and harvested in June of 1988. The yield obtained for wheat was 3.4 Mg/ha (57 bu/a) which was large enough to exceed the target yield value for the soils being restored.

It was observed that several small depressions occurred in two tracts of the replaced prime land. Two treatments were used in an attempt to reduce, if not remove, the effects that these low spots might have on succeeding corn crops. In the summer of 1988 after wheat harvesting, these areas were first disked and then "land leveled" with a bucket type land leveler which is commonly used on river bottom soils in western Kentucky. This technique appeared to remove most of the small depressions, but still remaining were some larger low spots. It is presumed that the large depressions were caused by differential settling of the spoils occurring under the replaced top- and subsoil. These areas were "ditched" with sub-surface field drain (perforated plastic) pipe. This drain pipe was installed at a depth ranging from 45-60 cm.

A cover crop of ryegrass and hairy vetch, both seeded at 28 kg/ha, was established in the fall of 1988. Fertilizers used in the fall were 112 kg/ha of 18-46-0 and 56 kg/ha of 0-0-60.

In the spring of 1989, corn was planted at 54,600 kernels per hectare using a no-til corn drill into the cover crop "killed" with herbicide. Since not all of the area treated with herbicide resulted in a killed sod, part of the area (6 hectares

occurring in two fields) had to be replanted to corn using the conventional approach, including plowing and seedbed preparation. The corn hybrid used on most of the area was McCurdy® 7676, which had been the top producer in the 1987 variety test conducted by Powell et al., 1988, and very near the top in the 1988. The two areas that were replanted were seeded to Pioneer® 3165, the top yielding corn variety in 1988.

Fertilizer used for the corn in 1989 was as follows: 450 kg/ha of urea (46-0-0); 112 kg/ha of 18-46-0; and 170 kg/ha of 0-0-60. The urea and potash were broadcast just prior to planting, and the 18-46-0 was applied in the row with the corn drill. In addition, 112 kg/ha of 34-0-0 was side-dressed on the corn when it had reached approximately 30 cm in height. However, anhydrous ammonia side-dressing was used for the replanted areas.

The herbicides and application sequences used to kill the cover crop as well as to control weeds were as follows. About two weeks prior to planting corn, glyphosate [N(phosphonomethyl) glycine] was sprayed at a rate of 4.6 l/ha (2qt./acre) in 325 l (35 gal.) of water with a tractor-mounted boom sprayer. The corn drill was also equipped with a sprayer and the following herbicides were used at planting: alachlor [2-chloro-N-(2,6-diethylphenyl)-N-(methoxymethyl) acetamide], 5.8 l/ha; triazine [6-chloro-N-ethyl-N'-(1-methyethyl)-1,3,5-triazine-2,4-diamine] at 4.7 l/ha. In addition, an insecticide [o,o-diethyl o-(3,5,6-trichloro-2-pyridinyl)-phosphorothioate] was also used at recommended rates. One field was resprayed with triazine following corn emergence to control the perennial ryegrass not killed initially.

Corn harvest began September 19 using two combines, and was completed October 10. One combine was a two-row "plot" combine which was used to measure yields on every 19th and 20th row (i.e., 10% of total acreage). This combine was driven at a constant speed across these reclaimed areas, and yields were determined about every 20 meters. Data (weight and % moisture) were recorded on paper (strip-charts). All grain harvested by both the plot combine and the conventional combine was placed in trucks and sold at a local grain elevator in order to obtain total yield from each field. At Alston, the permit was divided into 12 fields and corn yield from each was kept separate. This was useful for other reasons, as some of these fields were treated differently with respect to herbicide, corn hybrid, and planting technique used. At the other mines, only one yield was determined for each property.

**River Queen Mine.**

Two areas existed on this mine and both were treated in the same way. The two sites are referred to as "Cherry Hill" and "Cedar Grove," respectively. The soil at both sites had been replaced with scraper pans and not end-dump trucks as was used at Alston. In general, all of the other reclamation techniques, herbicide rates, and harvesting methods were the same as described for the Alston Mine with one exception. These areas had been in a sod cover crop for two years before they were planted to corn in 1988. In the fall of 1987, both areas were bush hogged as low as possible and 112 kg/ha of 34-0-0 was applied.

At one location, Cherry Hill, the target yield level for corn was obtained in 1988. Except for small experimental plots, this was the first time yield goals were met by Peabody Coal Co. in Kentucky on a field scale. Target yields were not obtained at the Cedar Grove site. Hence in the fall, this site was seeded to a grass/legume cover crop as described for the Alston Mine site, but this cover crop was largely grass.

**Moorman Mine.**

The soil for this area had been replaced using scraper pans, and was treated in a similar manner as described for the Alston Mine. Corn was first planted on this reclaimed soil in 1989.

## Results and Discussion

Many soil and crop management factors can affect corn yield of restored prime farmland soils. Using the right combination (or combinations) should produce yields sufficiently high to meet the target yield for Phase III bond release. Minimizing the effect of excessive soil compaction is often the key factor in achieving a high corn yield on mined lands. Other factors such as providing adequate plant nutrients, selecting the best crop and crop varieties, proper land shaping, and the correct soil depth are also likely to limit corn yield when incorrect choices are made. It is as important to prepare a good mining and reclamation plan as it is to properly carry out these plans in the field. This includes everything from the initial soil removal phase to the harvesting of the crops.

The corn yield data are summarized in Table 1. All values were collected in 1989 except for the Cherry Hill site, for which the data was obtained in 1988. Field average yields ranged from 6620 kg/ha (100.8 bu/acre) to 8480 kg/ha (135.2 bu/acre). The weighted average for all sites was 7390 kg/ha (117.9 bu/acre). Differences between sites and/or fields are attributed to soil management practices, i.e., conventional versus no-til; sloping versus flat fields; and differences in weed control or efficiency of herbicides in controlling the cover crop.

Table 1. Corn yields from restored prime farmland.

| Site | Field Name | Size | | Yield |
|---|---|---|---|---|
| | | hectares | | kg/ha |
| Alston | Bell | 18.9 | | 7310 |
| Alston | Tichner | 32.6 | | 7680 |
| Alston | Tichner (plowed) | 2.0 | | 6480 |
| Alston | Hoover (plowed) | 4.3 | | 8480 |
| River Queen | Cedar Grove | 3.7 | | 7340 |
| River Queen | Cherry Hill (1988) | 8.1 | | 6620 |
| Moorman | | 3.8 | | 6320 |
| Summary | Total | 73.4 | Average* | 7390 |

* Weighted Average -- All yields listed for individual fields are based on the elevator weigh tickets and area for the whole field.

## Cover Crop and Soil Management

In Kentucky, the use of no-til corn planting equipment will more likely be the best approach to achieve maximum corn yield on restored mine soils. This allows a mulch to remain on the surface in the early stages of corn growth. Such a mulch reduces erosion and water evaporation losses, and since replaced soils are usually droughty, the reduced water loss should help achieve target yield level. However, problems may occur that could jeopardize yields. An example of this occurred for three fields at the Alston site. The herbicides used to control the cover crop did not work properly. A decision had to be made to answer the following question. Will the remaining grasses use water that the corn will need for maximum production? Two approaches were tried. One area was resprayed (Tichner) following corn emergance, and two fields were plowed and replanted using conventional methods. The reason for the poor kill was attributed to cold rainy weather and which rendered the contact glyphosate herbicide ineffective. Improper application or collection or the incorrect herbicide could have also resulted in a similar problem.

When the cover crop is not killed early enough prior to planting, corn yields may actually be reduced as a result of lower available soil moisture. Although it is not believed that reduced available soil moisture had occurred this early in the growing season, another problem was associated with the poor killing of the cover crop, this being the grasses and legumes caused the ground to become dry and hard at the surface, thus preventing the planter from running at the proper planting depth. A second factor could also be involved, the vigorous cover crop was so thick that it may have caused the depth gauges of the planter to hold the disk blades out of the ground. However, it is believed the problem was largely the result of the soil being dry and hard because in an adjacent preplanting data research plot, similar conditions existed for the first planting but, following a 0.5-inch rain the corn stand was excellent and apparently the vigorous stand did not prevent the no-til planter from penetrating the soil.

The corn yields from the two fields at Alston in which conventional tillage was used were different from the no-til area. In one case, the Tichner-plowed field had a lower yield than either the Tichner or Bell no-til fields, i.e., 6480 versus 7680 or 7310 kg/ha. However, for the other plowed field (Hoover), the yield was higher (8480 kg/ha) than for the two no-til sites. We did not have a true test as to what the effect of leaving the growing cover crop would have had on yield had we not plowed or resprayed the three fields. It should be pointed out that the yield from the Hoover field was the highest average yield reported in Table 1.

The grain from the small field that was resprayed was not weighed separately from the larger Tichner field in which a good herbicide kill of the cover crop occurred, hence direct conclusions cannot be drawn. Unfortunately, data for the individual strip harvests made by the combine have not been completely analyzed to statistically compare the yields from

the strip harvests and the whole field data[3]. Variations in yield occurred within all fields including the two plowed fields, the resprayed field, and the field where a good kill of the cover crop was achieved. However, the means from the strip harvests were similar to the totals calculated from the whole field approach.

Although some problems were experienced in 1989 with the no-til method of planting corn, as stated earlier, we still believe this method to be generally superior to the conventional method. One additional problem that has been observed on mon-mined land is associated with the cover crop serving as a habitat for small animals, such as mice, that feed on corn seed prior to germination.

An adequate corn plant population is essential for maximizing corn yield. The number of kernels many farmers plant in western Kentucky near our study sites is 64,000 per hectare. However, it has been found that a population 22 to 28 percent lower (i.e., 46,000 to 50,000) gave better yields for restored prime farmland (Powell et al., 1988). This reduced population is especially desirable during years of moisture stress.

**Soil Handling and Land Shaping.**

Handling of soils in a way to minimize compaction and loss of natural soil structure is not an easy task. It has been found by Barnhisel et al. (1987) and Powell et al. (1985) that end-dump trucks which are not driven on the replaced soil is the most effective way of soil replacement. Even soil moved by trucks can be compacted during the leveling phase. The soil structure of wet soil tends to be damaged more by improper handling and leveling than dry soil.

Since scraper pans are often used in soil relocation, the increased compaction needs to be removed by some method. Biological loosening by growing deep-rooted crops such as alfalfa, has been shown by Powell et al. (1985) to be one method of reducing excessive compaction . The more common practice to reduce the adverse effects of excessive bulk density is with deep tillage. The Rome® ripper was used on all sites discussed in this paper, therefore an effect of ripping is not shown in Table 1. A yield response as much as 1250 kg/ha (20 bu/acre) has been observed in research plots[4]. Other methods of soil handling (e.g., bucket wheel excavator), ripping, or subsoiling may also achieve good conditions for rooting, as demonstrated[5] in Illinois by Jansen and associates (Jansen, 1981; Jansen and Dunker, 1987; and McSweeny et al., 1987).

It is highly desirable that the final graded surface have a slope and not be graded flat. The adverse effects caused by a nearly level field may be demonstrated by inspection of data in Table 1. The plowed Tichner field was essentially level. Differential settling of the spoils occurred during the winter of 1988-1989. Small depressions, 10-15 cm deep appeared similar to a micro Karst topography. Even though these low areas were relatively small, they retained water long enough to reduce corn stands and the final yield. It is likely that the effect of micro relief reduced yield as much as 2000 kg/ha (32 bu/acre), the differences between the Tichner- and Hoover-plowed fields. Furthermore, as noted in the methods section, a land leveler was used at this site in an attempt to remove these depressions. However, the spoils continued to settle and this was another reason why we plowed this field in a second try to remove the effects of differential spoil subscidence.

Spoils should be graded to the same proposed final grade of the prime farmland areas. This is necessary to insure sufficient soil depth is achieved on the entire area. If the topography prior to mining had very little relief, the coal operator should provide in his/or her mining plan, some way to provide surface drainage.

Increased bulk density or soil compaction causes restored prime farmland soils to be droughty and thus affects corn growth. This is largely due to the reduced water storage. Increased density has the potential of altering many soil properties, most of which will reduce the yield potential of the soil. Root growth is reduced due to a smaller number and size of soil pores in which to grow and expand. The roots that do exist tend to be more concentrated in the surface where tillage has loosened the soil. If the rainfall distribution is uniform or sufficient, yields may not be affected by excessive compaction except that there is a greater tendency for lodging upon maturity.

In summary, any soil property that affects water infiltration, movement, and storage could adversely affect corn yield. Any management practice that can minimize the effect of reduced root growth, or conversely stimulate root growth, will likely cause an increased probability of meeting target corn yield levels.

## Conclusions

1. The use of end-dump trucks versus scrapper pans may result in less compaction, but in any case, if excessive bulk density were to occur, its removal or reduction is likely to be needed in order to achieve high

---

3 These data will be available when the paper is presented orally or may be obtained by writing the Senior author.

4 Unpublished data of senior author

5 Also includes unpublished data presented at Field Days, 1988 and 1989.

enough corn yields to meet the target yield level for Phase III bond release.

2. Follow the best available agronomic soil and crop management practices available. For example, the no-til corn planting method appears to be superior than the convential plow-seedbed method. However, the advantages of the no-til system may be negated if problems were to occur as were experienced at one site, i.e., failure to kill the cover crop.

3. Selection of proper corn hybrids, seeding rate, and planting date may significantly affect the final yield. Plant one of the top yielding hybrids recommended for the area, preferrably one tested on restored mine land in the area.

4. Utilize soil test data to establish lime and fertilizer application rates and follow proven agronomic practices in applying and incorporating these materials.

5. There is an advantage to reducing the effect of differential spoil settling by preparing the final grade at a slope of 1 to 2 percent. Avoid large areas with final grades less than 1 percent and greater than 4 to 5. percent.

## Acknowledgment

Many of the ideas incorporated into the reclamation concepts presented in this paper were generated from several persons, especially Mr. James L. Powell . Many of these ideas were contained in the various permits prepared by Jim when he was an employee of Peabody Coal Co. Valuable ideas have been suggested by employees of Kentucky Reclamation Association (KRA) who have been contracted to do much of the revegetation, fertilization, etc. associated with these reclamation activities. We wish to acknowledge Mr. Harold Coleman and Mr. Ronnie Newton of KRA.

## Literature Cited

Barnhisel, R.I. 1983. The environmental consequence of burial depth of toxic spoils and of excessive compaction of prime land on the growth of plants. U.S. Department of the Interior (Office of Surface Mining), Division of Technical Services and Research, Washington DC. 175 p.

Barnhisel, R.I. and J.L. Powell. 1985. Reclamation of prime farmland in Kentucky. p. 1-12. In D. Williams and S.E. Fisher (eds.) Bridging the Gap Between Science, Regulation, and the Surface Mining Operation. Proceedings of 2nd Annual Meeting for American Society for Surface Mining and Reclamation. Denver, CO., Oct. 8-10.

Barnhisel, R.I., J.L. Powell, M.J. Bitzer, and W.O. Thom. 1989. Grain sorghum and soybean variety tests on reconstructed prime land--1988. Reclamation News and Views 7(1) March. University of Kentucky Coop. Ext. Ser., Lexington, KY. 5 p.

Barnhisel, R.I., J.L. Powell, and D.H. Hines. 1987. Changes in chemical and physical properties of two soils in the process of surface mining. Am. Soc. Surface Mining and Reclamation 4:87-97.

Barnhisel, R.I., J.L. Powell, J.R. Armstrong, M.L. Ellis, and F.A. Craig. 1988. Effect of soil depth, liming, and subsoiling of reconstructed prime farmland on alfalfa production. p. 266-273. Mine Drainage and Surface Mine Reclamation. Vol II: Mine Reclamation, Abandoned Mine Lands and Policy Issues. USDI, Bureau of Mines. Information Circular/1988 IC 9184.

Barnhisel, R.I., G. Wilmhoff, and J.L. Powell. 1979. Characterization of soil properties of reconstructed prime and non prime land in Western Kentucky. p. 119-122. Proceedings of the Symposium on Surface Mining Hydrology, Sedimentology, and Reclamation. University of Kentucky, Lexington, Ky., December 4-7, 1979.

Fehrenbacher, D.J., I.J. Jansen, and J.B. Fehrenbacher 1982. Corn root development in constructed soils on surface-mined land in western Illinois. Soil Sci. Soc. Am. J. 46:353-359.

Huntington, T.G., R.I. Barnhisel, and J.L. Powell. 1980. The role of soil thickness, subsoiling, and lime incorporation methods on the reclamation of acid surface-mine spoils. p. 9-13. Proceedings of the Symposium on Surface Mining, Hydrology, Sedimentology, and Reclamation. University of Kentucky, Lexington, Ky. December 1-5, 1980.

Jansen, I.J. 1981. Reconstructing soils after surface mining of prime agricultural land. Mining Engineer. 33:312-315.

Jansen, I.J. and R. E. Dunker. 1987. Reclamation for row crop production after surface mining, State-of-the Art. Am. Soc. Surface Mining and Reclam. 4:Paper 4-2. 7p.

Jansen, I.J., R.E. Dunker, C.W. Boast, and C.L. Hooks. 1984. Row crop yield response to soil horizon replacement. Am. Soc. Surface Mining and Reclam. 1:413-430

McSweeney, K., I.J. Jansen, C.W. Boast, and R.E. Dunker. 1987. Row crop productivity of eight constructed minesoils. Reclam. and Reveg. Res. 6:137-144.

Powell, J.L., R.I. Barnhisel, W.O. Thom, M.L. Ellis, J.R. Armstrong, and F.A. Craig. 1985. Reclamation of prime farmland in Kentucky. Am. Soc. for Surface Mining and Research 2:1-12.

Powell, J.L., R.I. Barnhisel, and C.G. Poneleit. 1988. The role of corn variety and population in meeting target yield level for Phase III bond release. p. 257-265. Mine Drainage and Surface Mine Reclamation. Vol II: Mine Reclamation, Abandoned Mine Lands and Policy Issues. USDI, Bureau of Mines. Information Circular/1988 IC 9184.

Powell, J.L., R.I. Barnhisel, and W.O. Thom. 1987. Utilization of soft red winter wheat for reclamation of surface-mined lands in western Kentucky. I. Effects of cultivar selection, soil depth, and ripping. J. Prod. Agr. 1:304-308.

This page intentionally left blank.

# Reclamation for Improved Land Use in Appalachia

This page intentionally left blank.

# Variances from "Approximate Original Contour" Requirements in Central Appalachia: History and Prospects[1]

by
Carl E. Zipper[2]

**Abstract** . The Surface Mining Control and Reclamation Act (SMCRA) requires mining firms to restore mined land to "approximate original contour" (AOC) unless a variance is obtained. In central Appalachian mining regions, AOC variance applications are not common, in spite of the region's needs for flat lands. In adding the AOC variance language to legislation which became SMCRA, it was the intent of Congress to allow surface mining firms to produce flat lands to meet the needs of Appalachian communities. There are many reasons why mining firms do not choose to take advantage of the AOC variance option in more widespread fashion. Community and economic development leaders must take the initiative if AOC variance reclamation is to have an impact on central Appalachia's needs for developable lands. Legal and regulatory changes could lower barriers to AOC variance reclamation. Additional research is needed to develop technologies which will aid development of mined lands.

Additional key words: Surface Mining Control and Reclamation Act.

## Introduction

Since enactment in 1977, the Surface Mining Control and Reclamation Act (SMCRA) has had profound effects on the coal industry and the environment. These effects been strongly felt in the mountains of central Appalachia, the area of the United States that has been most intensively affected by mining.

One result of SMCRA has been the widespread restoration of mined land to "approximate original contour" (AOC). SMCRA Section 515(b)3 requires mining operators to "restore the approximate original contour of the land, with all highwalls, spoil piles, and depressions eliminated." Implementation of AOC, and associated SMCRA environmental protection performance standards, have reduced the environmental effects of surface coal mining.

However, SMCRA's AOC requirements also constrain mining firms from producing level lands to serve the needs of central Appalachian communities. Non-flood prone flat lands are at a premium in this mountainous region, and housing, commercial, and industrial sites are in short supply. This constraint occurs in spite of the "AOC variance" provisions of SMCRA which allow waiver of the AOC requirements in situations where the reconstructed landform will serve an "equal or better economic of public use" than that which preceeded mining [Sec.515(c), 515(d)]. This constraint is perceived as a problem because of the high rates of unemployment and poverty, poor housing conditions, and other quality of life problems which affect central Appalachia. These problems result in part from shortages of developable land.

The purpose of this paper is to present current and historical information regarding the AOC and AOC variance requirements of SMCRA. The paper will place the development of these requirements in an historical context, so as to clearly identify congressional intent. It will also define reasons for mining operators' failure to take more widespread advantage of AOC variance opportunities, and suggest actions that will enhance the ability of mining firms to produce developable lands in central Appalachia.

## The Historical Context

Prior to the 1970's, "shoot and shove" mining was creating exposed highwalls and unstable outslopes throughout Appalachia. The environmental effects of these mining practices received the attention of Congress and the nation, leading to increasingly stringent state laws in the 1970s. These effects also brought about the development of federal legislation providing national environmental protection standards for the surface mining industry. These efforts led to PL 95-87, the Surface Mining Control and Reclamation Act of 1977.

The effort to forge the law that became SMCRA was protracted. The first federal surface mining bill was introduced by Everett Dirksen in 1940 (Dunlap 1975); the legislative effort which culminated in SMCRA began in 1971 (Waters 1979). Over the next six years, the establishment of federal controls over surface mining was a major agenda item in both houses of Congress. Activities included 183 days of hearings, 18 days of House action, three House-Senate conferences, 11 House Committee Reports, two Presidential vetoes (Udall 1979), and additional activity in the Senate.

---

[1] Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990. Publication in this proceedings does not preclude authors from publishing their manuscripts, whole or in part, in other publication outlets.

[2] Carl E. Zipper is Associate Director, Powell River Project, and Associate Director, Virginia Center for Coal and Energy Research, Virginia Polytechnic Institute and State University, Blacksburg, Virginia 24061.

**Approximate Original Contour - Sec.515(b)3**

The AOC concept was advanced in the initial attempt to establish federal jurisdiction over surface mining, the legislative proposal by Everett Dirksen in 1940. This bill required mining operators to perform reclamation operations "as may be necessary to make the contour of the land approximately the same as before the mining operation was begun" (Dunlap 1975).

The forerunner of the modern day AOC legislation was introduced by Congressman Wayne Hays of Ohio (Simpson 1985). House bill H.R. 6482 required operators to "backfill the operations so that the approximate original contour of the land is restored ..." [Sec. 18; Cong. Rec. 118:35035. 11 October 1972]. However, this proposed legislation also allowed for liberal exemptions from the AOC requirement. Where restoration of the original terrain might result in negative environmental effects due to the steepness of reconstructed slopes, the mining operator would be able to "backfill and grade according to a plan of terracing and drainage that will eliminate ... " hazards such as soil erosion, slides, or acid drainage. H.R. 6482 also allowed the Secretary of the Interior to "approve an alternative plan" of reclamation that does not violate the purpose of the section, which was "to restore the area of land affected to the same or an equally useful purpose as before any mining." Such an open-ended variance concept was not repeated in any subsequent legislative packages which received the serious consideration of the Congress.

The 93rd Congress brought considerable discussion of the AOC question to both the House and Senate floors. Substantial reworking of the legislation in both chambers of Congress (Senate Bill S. 425 and House Bill H.R. 11500) led to AOC language closely resembling that of SMCRA Sec. 515(b)3, with the open-ended variance provisions of H.R. 4682 (92nd Congress) eliminated. This language remained in place through subsequent debate, receiving only minor changes. The result is that, today, surface mining operators from throughout the U.S. are required to "restore the approximate original contour of the land, with all highwalls, spoil piles, and depressions eliminated" [Sec. 515(b)3] unless a variance from the AOC requirement is obtained through the permitting process.

A reading of the Congressional Record and various committee hearing documents shows that, without a doubt, the will of Congress was behind the AOC requirements. Numerous attempts to weaken those requirements were defeated. In response to one such attempt, Senator Nelson called the AOC provisions "the very guts of the most important part of the bill, the most significant provision, or one of the two or three most significant provisions, in the measure." [Cong. Rec. 121:33326. 9 October 1975].

The primary rationale of Congress for establishing the AOC requirements was undoubtedly a desire to protect the environment from the effects of minimally-regulated surface mining. Major points of discussion included the following:

1. Major environmental problems were resulting from "pre-law" non-AOC mining, especially in the central Appalachian area. For example, a report by Mathematica, Inc. (1973) documented landslides, mud flows, and rock slides resulting from unstable spoil outslopes. The results of a 1964 aerial survey indicated that an estimated 12% of the total surface mined area in eastern Kentucky was in slides. By 1972, in spite of considerable tightening of Kentucky's regulatory standards, the annual eastern Kentucky slide area had increased to 400 hectares.

   When properly implemented, AOC results in placement of spoils in the most stable place in the mined landscape: the level bench produced by the mining disturbance.

2. Pre-law mining was resulting in "aesthetic disturbances" including exposed highwalls ("scars in the mountainsides") and poorly revegetated outslopes (Mathematica 1973; CEQ 1974). These visible symptoms of pre-law mining were the targets of comments by numerous witnesses at congressional hearings, and they came to symbolize the abuses of uncontrolled Appalachian surface mining (Simpson 1985). AOC reclamation results in coverage of the highwall and elimination of outslope spoils.

3. The "haulback" mining method was being widely cited as the most cost-effective method for achieving AOC in steep-slope terrain. An additional virtue of haulback mining, from an environmental standpoint, is that it allows contemporaneous reclamation. That is, spoil is removed from a current mining area directly to the mined-out pits. Other common surface mining methods of that time required reclamation to be performed as a separate operation (i.e. spoil must be rehandled after mining to achieve reclamation). Thus, adoption of an AOC reclamation standard stimulated mining operators to adopt haulback methods, thus reducing opportunities to abandon surface mining operations after removing coal but prior to initiating reclamation.

4. Congress was under the impression that economic impacts of environmental protection achieved through widespread adoption of AOC would be minimal. The incremental costs of AOC were estimated at $2.00 to $3.00 per ton, on average, although those costs were acknowledged to be greater in steep-slope mining areas such as Virginia, eastern Kentucky and southern West Virginia (ICF 1977). In 1971, the outset of the legislative process which led to SMCRA, average F.O.B. mine prices for coal were approximately $7.00 per ton (US DOE 1980). By 1975, average F.O.B. mine prices had risen to approximately $19.00 per ton. Under these pricing conditions, it appeared that the marginal reclamation costs imposed by AOC would have minimal effects.

**AOC Variance for Mountaintop Removal - Sec. 515(c)**

The proposal to grant variances for mountaintop removal mining was introduced in the 93rd Congress, in Senate Bill S. 425, Section 213(c) [Cong. Rec. 120:33338. 9 October 1974]. The proposed requirements in that legislation were similar in most respects to SMCRA Section 515(c), which governs mountaintop removal variances. However, the one major difference was that no specific post-mining land use requirements were imposed.

In the House of Representatives during the 93rd Congress, the primary surface mining bill (H.R. 11500) also contained an AOC variance provision [Sec. 211(d)]. This was a more general provision, not restricted to mountaintop removal mining. This clause allowed the "regulatory authority to grant appropriate exceptions to the requirements" for restoring the land to AOC, in cases where "an industrial, commercial, residential, agricultural, or public facility development is proposed" for the affected lands, provided the proposed use is deemed to constitute an "equal or better economic or public use" and this use "can only be obtained" through variance from AOC [Cong. Rec. 120:25280. 25 July 1974].

The House-Senate Conference produced a bill [S. 425] which contained language essentially the same as that of todays 515(c), and in essence contained a more restrictive AOC variance provision than either of the two bills advanced by the House and Senate.

In order to receive a variance from the AOC requirements of SMCRA on a mining operation that would "remove an entire coal seam or seams running through the upper

fraction of a mountain, ridge, or hill", a mining operator would have to conform to these general standards:

1. The planned reclamation practices must prepare the land for an industrial, commercial, agricultural, residential, or public post-mining land use that is "deemed to constitute and equal or better economic or public use ... as compared with the premining use." [515(c)3].

2. The planned land use must be compatible with adjacent land uses, and "appropriate assurances" must be presented to show that the planned land use is practical and attainable. Required assurances include documentation of planned investment in public facilities that might be required to support the proposed land use, expected need and market for the products or services to be provided by the planned land use, "commitments by public agencies, where appropriate", and compatibility with adjacent land uses [515(c)3].

3. The reclaimed area will be designed by a professional engineer, will remain stable, and "no damage will be done to the natural water courses" [515(c)4].

## AOC Variance for Steep-Slope Mining - Sec.515(e)

Senate Bill S. 425 (SMCRA of 1974) was passed by majority votes in both houses during the closing days of the 93rd Congress, but was pocket vetoed by President Ford. In a letter to Congress, the President expressed one of his dissatisfactions as being the lack of allowance of "limited variances -- with strict environmental safeguards -- to achieve specific post-mining land uses" [Cong. Rec. 121:5852. 10 March 1975]. In spite of the President's plea, the 94th Congress included no additional AOC variance procedures to the bills which were to become the basis for H.R. 2 and S. 7.

The forerunner of SMCRA's steep-slope mining AOC variance requirement [Sec. 515(e)] was proposed by Senator Ford of Kentucky during the first session of the 95th Congress [Cong. Rec. 123:15705. 20 May 1977]. This proposed amendment to the legislation contained wording nearly identical to SMCRA Section 515(e), with two major exceptions:

1. The proposed amendment would have allowed the variance to be applied to agricultural land uses, as well as commercial, industrial, residential, and public uses.

2. The proposed amendment would have made variances available on all mining; the variance applied to forerunners of two SMCRA clauses: 515(b)3, the general AOC clause, and 515(d)2, which requires AOC in "steep-slope" mining situations. Today's 515(e) grants variances only to Section 515(d)2.

In all other respects, Senator Ford's proposal was identical to today's 515(e). The House bill contained no similar AOC variance provision. The result of the House-Senate Conference was a bill containing the modified version of Senator Ford's amendment that became Section 515(e). Once again, the House-Senate Conference tightened the proposed AOC variance clause.

Under 515(e), a "steep-slope" mining operation (i.e. an operation in an area where the average slope is in excess of 20 degrees, "or such lesser slope as might be defined by the regulatory authority" [515(d)4] ) may receive an AOC variance for an approved (industrial, commercial, residential, or public) land use by meeting the following requirements:

1. The planned operations will "render the land, after reclamation, suitable for" the proposed use, which will be an "equal or better use" than that which preceded mining.

2. The post-mining landform is designed by a professional engineer, and will remain stable.

3. The watershed of the affected area will be improved.

4. No more spoil will be placed off the mine bench than is necessary to achieve the proposed use.

It is notable that Section 515(e) does not require detailed "appropriate assurances" be provided, as in the mountaintop removal variance permitting requirements.

## OSMRE Regulations Governing AOC Variance

The statutes of SMCRA are augmented by the United States Office of Surface Mining Reclamation and Enforcement regulatory program. These regulations contain more detailed requirements for AOC variances authorized by Sections 515(c) [30CFR 785.14 and 30CFR 824] and 515(e) [30 CFR 785.15 and 785.16] of SMCRA. Variances obtained under either of these two sets of requirements must comply with the OSMRE regulation pertaining to alternative post-mining land use [30CFR 816.133(c)], which requires that there be a "reasonable likelihood for achievement" of the planned post-mining land use, and that the use will not be "impractical or unreasonable" nor will it "involve unreasonable delay in implementation."

The regulations in 30CFR 785.15 give definition to the "watershed improvement" requirement of 515(e). This clause states that the watershed will be deemed to be "improved" if "compared with the condition of the waters before mining or with its condition if the approximate original contour were to be restored".

1. Total suspended solids or other pollutants will be reduced, or

2. Flood hazards within the affected watershed will be reduced,

and flows will not vary in a manner adversely affecting local ecology or water use.

## Congressional Intent

This review indicates that without a doubt, Congress intended to allow mining and associated reclamation activities to provide sites suitable for industrial, commercial, industrial, and public land uses which would meet the needs of Appalachian communities. However, Congress wished the variance provisions to be restrictive, not open ended. Reduction of the environmental impacts of mining was maintained as a first priority.

There is ample evidence of Congressional intention to maintain strict controls over AOC variance reclamation. In both the 93rd and the 95th Congress, the House-Senate Conference Committee developed compromise legislation by tightening language defining AOC variance proposals. In 1973, Senator Allen of Alabama proposed amended language to the AOC requirements which called for land to be restored to the approximate original contour unless "another surface configuration is equally effective in controlling erosion, siltation, and rainwater runoff ..." [Cong. Rec. 119:33326. 9 October 1973]. This proposed amendment was soundly defeated, indicating that the Congress was not content to allow variance from AOC on purely environmental grounds. In 1975, in response to President Ford's call for additional variances to achieve specific post-mining land uses, the Conference Report rejected this proposal with the statement, "The Committee believes that unlimited variances would greatly weaken the bill by possibly becoming the rule rather than the exception" [Cong. Rec. 121:5852. 10 March 1975]. In 1977, as Senator Ford proposed the pre-

cursor to SMCRA Section 515(e), his remarks included, "I share your concern on the subject of control. We must guard against a flexible provision that would allow abuses. Obviously, we do not wish to encourage a situation that would allow variances to become the rule" [Cong. Rec. 123:15706. 20 May 1977]. In 1977, the House-Senate Conference Report on House Bill H.R. 2 and Senate Bill S. 7, regarding changes imposed to produce SMCRA Section 515(e), stated: "The variance however is only for developed land uses such as industrial, residential or commercial sites. Agricultural, open space, and similar types of land uses do not qualify" [H.R. Report 95-493, 95th Congress, first session].

There are examples of AOC variance mining procedures being used to prepare lands for improved land use in central Appalachia. For example, near Norton and Wise, Virginia, commercial areas have been prepared on recently-reclaimed mine areas. Near Hazard, Kentucky, commercial development has also occurred on reclaimed mine areas. However, application of AOC variance has not been widespread, although a great need for developable land exists in central Appalachia. When AOC variance reclamation is used to prepare lands for improved uses, such activities generally occur in locales where real estate is highly valued, such as potential commercial areas located next to major highways. There has been very little activity in the remotely-located areas of Appalachia, off the major highways and away from larger towns, where the need for flat land tends to be greatest.

## Reasons for Lack of AOC Variance Activity

Difficulties in meeting the AOC variance requirements have to do with three major areas: planning and permitting, mining operations, and technical constraints. From the standpoint of the mining industry, there are major difficulties inherent in the necessity of submitting plans for implementing an improved (i.e. industrial, commercial, residential, or public) land use with the mining permit application. Especially in cases where the variance is sought under the mountaintop removal variance clause [515(c)], the amount of detail that must be provided can impose substantial costs on the permitting process. Data on markets for a proposed service or product (as required by 515(c)3B) will be particularly difficult to obtain years in advance of land use implementation. This market assessment problem also has implications relative to the fact that a land developer must be able to expect profit from development activities. Again, it will be difficult to gauge the profitability of a proposed investment far in advance. These constraints are most operative in situations where flat land suitable for development is needed most: the land deep in the hills, away from highways, population centers, and sources of employment. In Appalachia, many areas are "off the beaten track" primarily due to transportation and other difficulties caused by terrain.

Other problems result from the fact that the owners of many coal-bearing lands in Appalachia are corporations specializing in mineral and timber development. Some of these corporate interests lack experience in developing land for improved uses, and such activities may not be their primary business objective. The existence of unmined coal below the level of the proposed surface mining operation will constrain surface development, from the landowner standpoint, as modern full-seam coal extraction technologies often cause surface subsidence.

Performance bonding costs can also be increased by an AOC variance application. In many cases, altering the landform so as to increase available level area will also increase the area disturbed by mining by increasing the amount of excess spoil to be disposed in locations other than the mining bench. In such cases, if the performance bond amount is calculated on a per-acre basis, the bonding cost will be increased. More significantly, a "landform alteration" mining regime will, in many cases, entail a longer-term commitment by the mining operator. This can mean a substantial increase in the hypothetical "worst case" reclamation cost used to calculate the performance bond amount.

Although there are situations where a mining operation is rendered more cost effective by an AOC variance mining regime (Zipper et al. 1989), an AOC variance mining operation can also impose operational difficulties. Hollow fills can be costly to implement, as hollow fill construction is closely regulated. [30CFR 816.71-74]. Also, the long-term commitment required can impact the mining operator's ability to respond to negative price impacts by shutting down, or sharply curtailing production, at short notice.

There are also technical difficulties with implementing improved land uses on reclaimed surface mines. In remote areas, especially, waste disposal can be a problem, as septic drainfields are generally not permitted for placement on reclaimed surface mine fills. Also, young fills are prone to settlement, which can damage homes and other buildings if not constructed using methods that protect against structural distortion from differential settlement.

In general, environmental considerations are not a major hindrance to application of AOC variance mining procedures. AOC mining in central Appalachia has some inherent environmental liabilities, especially in steeply-sloping areas (Bell et al. 1989). When properly implemented, AOC variance mining activities can meet the regulatory requirements defining improvement of the natural watershed, by reducing the amount of steeply-sloping mined areas, and by reducing flooding potentials in mountainous watersheds (Zipper et al. 1989). Flat, level areas covered by deep, loose minesoils will, in general, absorb greater rainfall that steep, reclaimed slopes

## Implications

First, community and economic development leaders in the coalfield counties must recognize that the coal industry has little incentive to seek AOC variance for surface mining operations, so as to produce flat lands suitable for community development. AOC variance application can add to permitting time and cost. It can also add to the problems associated with obtaining performance bond while detracting from the operators ability to respond to unanticipated coal price changes and other marketing difficulties.

If AOC variance reclamation is to solve land use problems for Appalachian communities, community leaders and local governments must take the initiative. Where opportunities are recognized to produce flat lands that will aid community development, these opportunities need to be called to the attention of mining operators and landowners. Local governments can offer to help lower the barriers to obtaining an AOC variance by taking the initiative to provide documentation necessary to support AOC variance application, as defined by the relevant federal and state regulatory programs.

Legal and regulatory changes could also lower the barriers to AOC variance without producing "open ended" situations where variances from AOC become the rule rather than the exception. Specifically, SMCRA requires that the reclaimed land be rendered "suitable" for an approved "equal or better economic or public use" [515(c)3 and 515(e)2]. The ability of mining firms to produce lands that would be an aid to community development would be enhanced if criteria were developed for selectively waiving the requirement that the proposed land use actually be implemented at the immediate conclusion of mining. These criteria might include a demonstrated community need for the proposed land use, as certified by an independent third party. Given that a genuine need exists, the "reasonable

likelihood" that the land use for which the site is prepared would actually be implemented is very real. Thus, in communities where housing sites are in short supply, for example, the mining firm could be granted a variance to produce a site that is physically suited for housing (i.e. has access to public roads, potential access to water and waste disposal, and is located on stable ground) without accepting the obligation to arrange financing and to physically produce housing on that site. This change would remove a major impediment to mining firms' willingness to produce beneficial landforms.

Third, additional research is necessary to develop technologies that will aid development of mined lands. Hollow fills are a necessary component of most AOC variance mining plans in steeply sloping Appalachian terrain. Hollow fill construction is tightly regulated, due to the negative, and quite dramatic, environmental impact that would result from fill failure. Mining operators find the tight regulations associated with hollow fill construction, in many cases, to be a reason for avoiding hollow fill construction. Additional research aimed at finding more cost-effective, but environmentally sound, hollow fill construction techniques would aid the ability of mining firms to produce this level land. Regulators cannot liberalize hollow fill construction standards without asssurances that such actions will not compromise the environmental integrity of the resultant structures.

There are also technical difficulties associated with constructing buildings on reclaimed mine sites. Recently reclaimed mines are, essentially, soil and rock fills, subject to settlement with time and changes in moisture status (Krebs and Zipper 1989). Conventional construction techniques, designed for stable ground, would be inappropriate for mined lands of recent origin without costly modifications that detract from the economic flexibility of a mined land development venture. Research to develop cost-effective techniques for construction of buildings that can withstand the effects of settlement will aid mined land development potentials.

Waste disposal can also be a problem for developing mined lands in remote areas, where public sewers are not available. In mountainous Appalachian terrain, natural soils suitable for conventional septic drainfields are scarce, especially in the highlands. Reclaimed mine spoil is generally considered to be unsuitable for on-site waste disposal systems, due to the presence of multiple channels and voids that occurs when earth materials are placed in non-controlled fashion. The development of cost-effective waste disposal technologies for application on fill materials, possibly including controlled placement of soil and/or spoil materials during construction of specific areas designed to host septic drainfield waste disposal systems, would greatly aid mined land development prospects.

## Literature Cited


Bell, James C., W. Lee Daniels, and Carl E. Zipper. 1989. The practice of "approximate original contour" in the central Appalachians: I. Slope stability and erosion potential. Landscape and Urban Planning. (In press).

Council on Environmental Quality (CEQ) . 1973. Coal Surface Mining and Reclamation - An Environmental and Economic Assessment of Alternatives. Prepared for U.S. Senate Committee on Insular Affairs. Serial No. 93-8 (92-43). U.S. Government Printing Office, Washington, DC. 143 p.

Dunlap, Louise C. 1976. An analysis of the legislative history of the Surface Mining Control and Reclamation Act of 1975. Rocky Mountain Mineral Law Institute 21:11-58.

ICF, Inc. 1977. Energy and Economic Impacts of H.R. 13950 (Surface Mining Control and Reclamation Act of 1986). ICF, Inc. Washington, DC. National Technical Information Service PB-274 632.

Imhoff, Edgar A., T.O. Fritz, and J.R. LaFevers. 1976. A Guide to State Programs for the Reclamation of Surface Mined Areas. Circular 731. U.S. Geological Survey, Arlington, VA. 29 p.

Krebs, R.D., and C.E. Zipper. 1989. Foundations for Housing on Reclaimed Mined Lands. Virginia Cooperative Extension Service Publication 460-115. Blacksburg, Virginia. 12 p.

Mathematica, Inc. 1974. Design of Surface Mining Systems in Eastern Kentucky. Volume 1. Prepared for Appalachian Regional Commission and Kentucky Department of Natural Resources. National Technical Information Service PB 242 651. 82 p.

National Research Council (NRC). 1984. Highwall Elimination and Return to Approximate Original Contour as Required in the Surface Mining Control and Reclamation Act of 1977. Committee on Highwalls and Approximate Original Contour, Board on Minerals and Energy Resources, Commission on Natural Resources. National Academy Press. Washington, DC. ISBN 0-309-03448-5.

Simpson, John W. 1985. The emotional landscape and Public Law 95-87. Landscape Architecture 75(3):60-63+.

Udall, Morris K. 1979. The enactment of the Surface Mining Control and Reclamation Act in Retrospect. West Virginia Law Review. 81:553-557.

U.S. Department of Energy (US DOE). 1980. Coal Data: A Reference. Energy Information Administration. DOE-EIA-0064(80). UC-90.

Waters, Robert A. 1979. A summary of the legislative history of the Surface Mining Control and Reclamation Act of 1977 and the relevant periodical literature. West Virginia Law Review. 81:775-783.

Zipper. C.E., W.L. Daniels and J.C. Bell. 1989. Approximate original contour reclamation: An alternative in steep-slope terrains. Journal of Soil and Water Conservation 44(4):279-283.

This page intentionally left blank.

# MINED LAND RECLAMATION FOR WOOD PRODUCTION IN THE APPALACHIAN REGION[1]

J. A. Burger and J. L. Torbert[2]

Abstract. The future economic viability of forest land in the Appalachian coal-bearing region may be threatened by common post-mining reclamation practices. The Surface Mining Control and Reclamation Act (SMCRA) of 1977 allows post-mining conversions of forest land to grass land. This mined land, ostensibly restored for pasture and hay production, is often abandoned from normal management, precluding its timely restoration to forests. Also, even when forest land is intended as the post-mining land use, revegetation procedures are most often geared for establishing the temporary ground cover rather than the permanent forest crop. As a result, mine soils are often compacted, salty, alkaline, heavily fertilized and heavily seeded. These conditions discourage optimum survival and growth of desirable forest tree species.

Additional Key Words: restoration, reclamation with trees, mined-land value

## Introduction

### The Original Forest

Except for the mountainous topography of the central Appalachian coal-bearing region, the most distinctive feature of the landscape is that it is nearly totally in forest cover. The region is covered predominantly by the upland oak-hickory and Appalachian mixed-hardwood forest cover types. The great range of geology, soil, microclimate, and topography results in a range of species composing the forest stand (Burns 1983). White (Quercus alba L.), northern red (Quercus rubra L), and black oak (Quercus velutina Larn.) are found throughout with scarlet (Quercus coccinea Muenchh.) and chestnut oak (Quercus prinus L.) on drier sites. Pignut (Carya glabra (Mill.) sweet), mockernut (Carya Tomentosa Nutt.), and shagbark hickory (Carya ovato (Mill.) K. Koch) are consistent in stands but are a minor component. Species mixtures may also contain yellow-poplar (Liriodendron tulipifera L.), ash (Fraxinus spp.), elm (Ulmus spp.), sugar maple (Acer saccharum Marsh.), red maple (Acer rubrum L.), blackgum (Nyssa sylvatica Marsh.), black locust (Robinia pseudoacacia L.), black cherry (Prunus serotina Ehrh.), American beech (Fagus grandifolia Ehrh.), black walnut (Juglans nigra L.), and birch (Betula spp.) among other less populous hardwood species. In disturbed areas and on some natural sites, pitch (Pinus rigida Mill.), Virginia (Pinus virginia Mill.), eastern white(Pinus strobus L.), loblolly (Pinus taeda L.), and shortleaf pines (Pinus echinata Mill.), eastern hemlock (Tsuga canadensis (L.) Carr.) and eastern red cedar (Juniperus virginiana L.) are found. Common understory species are flowering dogwood (Cornus floridas L.), eastern redbud (Cercis canadensis L.), sassafras (Sassafras albidum (Nutt.) Ness), downy serviceberry (Amelanchier arborea (Michx.)), witch-hazel (Hamamelis virginiana L.), and mountain laurel (Kalmia latifolia L.).

### Forest Use and Potential

Since European man arrived in the region, the forest has acquired a 200-year history of use and abuse. The original forest was logged during 1890 through 1920, and was followed by high-grade logging of second and third growth forests. Logging practices were harsh on the forest site and in some cases abuse of the soil and landscape resulted in a permanent decrease in the productive potential of the land. However, the forest remains today as one of the largest and most valuable hardwood resources in the nation. Its multifaceted values include water quality and

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1989.

[2]James A. Burger is Associate Professor of Forest Soil Science, and John L. Torbert is Forest Soils Research Associate, School of Forestry and Wildlife Resources, Virginia Polytechnic Institute and State University, Blacksburg, VA 24061-0324.

quantity control, wildlife food and cover, recreation, regional biotic stability, and wood supply. Wood supply has traditionally been the most important and tangible forest value. Several studies predict high rates of return for new investments in wood using facilities (Donnell 1987 and McCoy and Wisdom 1984), but non-financial factors such as lack of adequate transportation networks and disaggregated timber ownership must be overcome before these otherwise fruitful investments can be realized (Robles and Wisdom 1989).

Forest Restoration After Surface Mining

Despite past abuses, the forest of the region remains intact, healthy, and valuable. It is extremely resilient and has regenerated and persisted through disturbances such as site-degrading harvesting methods, severe soil erosion, catastrophic fires, severe floods, and intense disease and insect attacks. On an acre-by-acre basis these disturbances are minor, however, compared to the disturbance created by surface mining for coal. In the process of surface mining, both the above and below-ground components of the forest are literally removed.

Reclamation of the land after mining is conducted under the regulatory auspices of P.L. 95-87, or SMCRA. Besides very specific mandates for overburden handling, drainage, erosion and sediment control, maintenance of water quality and final contouring, the law requires restoration of the land to its original use or to one of greater value. The purpose of this paper is to access the effect of current regulations and reclamation techniques on the potential of mined land for wood production. Specifically, conversions to grassland and degraded forest site quality are reviewed.

Forests to Grasslands Conversions

Reforestation can shorten or skip stages of primary forest succession. In its simplest form, forest succession can be described as a natural process whereby, following a disturbance, a community gradually regains the structure and floristic composition of adjacent communities on comparable sites by an orderly and predictable process (Clements 1916). The process entails species replacement and site amelioration. Several successional phases are commonly recognized (Figure 1): They include "pioneer," "building," "mature," and "climax stages." Other divisions and names have been used to describe successional stages, but this oversimplification is sufficient to contrast natural versus induced succession.

Because of steep slopes, shallow soils, and bedrock near the land surface, replaced "topsoils" are most often mixtures of A, E, B, and $C_r$ soil horizons. In many cases no "true" topsoil is recoverable and mine spoils are used as topsoil substitutes. In either case, if left to natural colonization by plants and animals, succession would begin at a primary level. Pioneering herbs, grasses and shrubs would invade and colonize the site followed by nitrogen fixing herbaceous and woody shrubs and trees during the "building" stage. Opportunistic hardwood and pine species would become dominant during the transition between the building and mature stages. Species would include sweet birch (Betula lenta L.), sourwood (Oxydendrum arboreum (L.) DC.), red maple, tulip tree, black cherry, and sycamore (Platanus occidentalis L.), and white, Virginia, and pitch pines. As these species developed into a mature forest, birds and rodents would move the heavier-seeded oaks and hickories onto the site. Over time, the somewhat-shade-tolerant oaks and hickories would persist in the understory and eventually make their way into the canopy as the biotic and abiotic character of the site changed to their advantage.

The rate at which succession would proceed through these stages would be highly variable, depending on the nature of the site and adjoining sources of propagules. It would probably require one to three hundred years for the oaks to dominate if forest restoration were left entirely to nature.

Reforestation procedures as recommended by Torbert et al. (1986 a,b) in Virginia involve hydroseeding acid-tolerant, low-statured grasses and legumes such as perennial ryegrass (Lolium perenne), redtop (Agrostis gigantea), birdsfoot trefoil (Lotus corniculatus), and "appalow" sericea lespedeza (Lespedeza cuneata var. appalow) with moderate to low rates of nitrogen fertilizer in order to provide adequate cover for erosion control, while minimizing the height of the ground cover. Low rates of nitrogen-fixing autumn olive (Elaeagmus umbellata), bristly (Robinia fertilis) and black locust, and European black alder (Alnus glutinosa) are hydroseeded with the ground cover as site-building nurse trees. Finally eastern white pine seedlings are planted as commercially-valuable crop trees.

This reforestation strategy is designed to shorten and "stack" the first three stages of natural succession described earlier. Instead of a slow process of species replacement over time, species representing each of the first three successional stages are sown and planted at once. All are acclimated to biologically depauperate soils, and all are relatively compatible with one another if sown at controlled rates. The grasses, serving as the pioneer vegetation, quickly yield to the legumes when applied nitrogen is minimized. The slow-starting ground-sprawling legumes allow the nitrogen-fixing nurse trees and the planted pines to become established and free to grow before totally covering the site. The legumes enrich the site and eventually give way to the tree cover. Because the nurse shrubs and trees are woody stems prone to pests, the white pines ultimately dominate the site and should become of harvestable age after 40 years on productive sites. During this time other species will invade and be poised for growth and development since the pines could be removed for their economic value. If the pines are harvested, a landowner may allow the site to

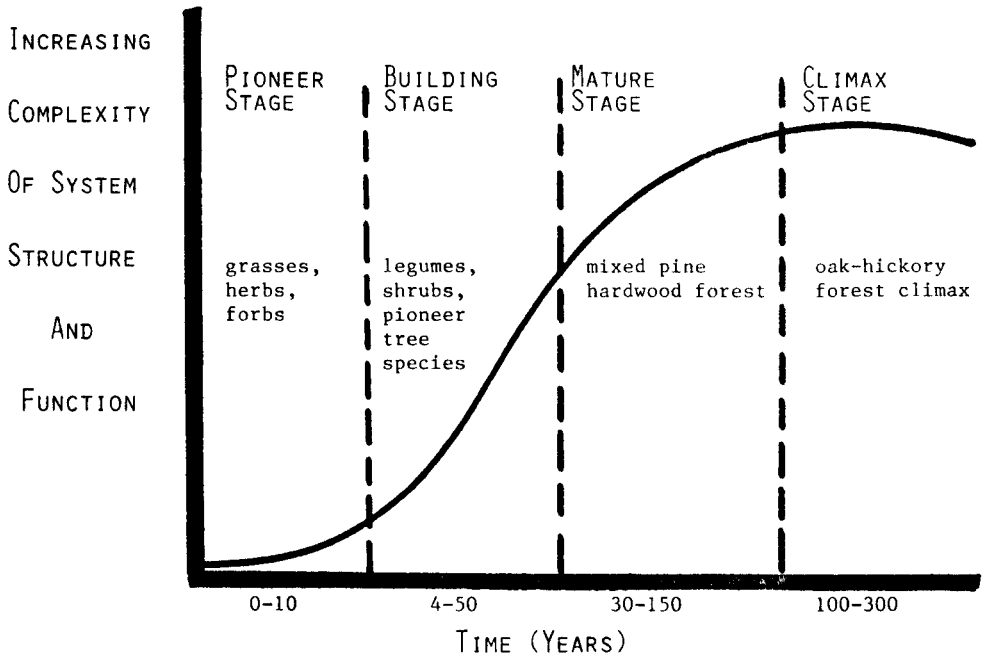



Figure 1. Primary succession on mined land in the Appalachian Region showing likely successional stages if land were not seeded or planted.

regenerate naturally or re-establish another even-aged pine plantation.

A major constraint of timely reforestation using this silvicultural scenario to "compress" natural succession is the option of converting forests to post-mining grasslands. Conversion of forest land to grassland is allowable under Sec. 816.133 of the OSMRE Permanent Program Regulations as "higher or better uses." During the decade of the eighties about 40 percent of the previously-forested mined land in Virginia was converted to grassland. In eastern Kentucky and southern West Virginia the amount was greater than 70 percent. Creation of as much grassland that could reasonably be used for livestock pro-duction in the region is good because it would help diversify the mining-based economy. But grassland established on steep, remote areas with no chance of being used for the intended purpose does not constitute a "higher or better use" than returning the site to its original land use --forestry.

The reality of post-mining land-use patterns in the Appalachian coal fields is that after all possible alternative uses are realized (including forage for livestock production), it is likely that nine out of ten surface-mined acres will return to forest land. This reforestation will occur either by design or by default. Designed reforestation will occur as a result of post-mining procedures that include tree planting such as that described above. Reforestation by default will occur as abandoned grassland is slowly invaded by forest tree species over time.

Compared to designed reforestation, default reforestation can set back the economic potential of the reclaimed land by 50 to 100 years. Grasses and legumes such as Ky-31 tall fescue (Festuca arundinacea, selection Ky-31) and sericea lespedeza are very persistent and yield very slowly to the successional chronology described above. In Virginia, land mined 15 to 25 years ago and revegetated under the auspices of state laws established in 1963 are still dominated by the fescue, lespedeza and black locust community that was established during reclamation. Even the most opportunistic non-leguminous hardwood pio-neers such as red maple, sourwood and black cherry have difficulty emerging. Abandoned grassland remains in the building stage of succession for a longer period of time than if lesser-competitive or native pioneer plants were planted.

## Restoring Forest Site Quality

The productive capability of an Appalachian hardwood forest is hard to measure because it is difficult to define. Theoretically, its value should be defined on the basis of the quality and quantity of water, wildlife habitat, recreation, and wood production. In reality the current post-mining productivity standard for forest land is not defined or measured in terms of any of these forest values; rather it is defined on the basis of the number of surviving woody stems per acre having a certain minimum height after a given period of time.

Unfortunately there is no basis for assuming that a certain number of five-year-old woody stems can be extrapolated to a representation of equal of better post-mining forest-land capability. A reclaimed corn field in the Eastern Interior Coal Region that contains the optimum number of stems at maturity (survival) but produces little or no grain (production) would not meet productivity standards. Reclaimed forest land in the Appalachian coal region that produces tree stems but no useable wood should also not meet productivity standards! It is unreasonable to assume that a stem count is all that is needed for a biologically and economically viable forest. Instead, a forest with a certain species composition growing on a restored mine soil that has the physical, chemical and biological properties "to put wood on those stems" is required.

Forest productivity is more than just wood production; however, when a proper composition of tree species is healthy and growing vigorously it usually results in a forest with higher levels of the other associated values. Therefore, since tree growth is relatively easier to measure, it might serve as an indicator of overall forest productivity. Studies of mine soil productivity in Virginia using white pine as an indicator species show the extent to which productivity varies (Torbert et al. 1988). In fact, the entire productivity spectrum is represented by sites on which trees are unable to survive, to sites on which trees are growing at rates faster than the average for undisturbed natural soils.

Because a forest crop is slow-growing, long-lived, and physically large when compared to an agricultural crop, it is difficult to see, measure and appreciate the different levels of production. In forestry, site quality or productivity is usually measured with an index defined as the average height of the canopy trees at a certain age. The data in Table 1, for example, compare the projected heights of white pines on three sites having different productive potentials due to different mine soil depths (Torbert et al. 1988). Projections are for 50 year old trees; however, since they are growing on sites of different quality due to different mine soil properties, their heights at age 50 are 60, 80, and 100 feet. This means that the sites on which they are growing have site indices (SI) of 60, 80, and 100, respectively. These SI's correspond to fair, good, and excellent for white pine in this region, since a SI of 80 is average for undisturbed soils (Beck 1971).

A mine soil able to produce a white pine stand 100 ft tall at age 50 versus a mine soil able to produce a 60 ft tall stand at age 50 may not impress one as being all that different until stand height is expressed in terms of relative wood volume and the value of that volume. Based on established stand volume tables, a mine soil having a SI=60 would produce 5.0 million board feet (MBF) of wood at a harvestable age of 35, compared to 32.1 MBF of wood on a minesoil of SI=100. Although tree height on the SI=60 site is 60 percent that of tree height on the SI=100 site, the volume is over six times greater on the better site. Because larger trees have more valuable uses, sawtimber versus mine props for example, the disparity between the fair (SI=60) and excellent (SI=100) mine soil is even greater. The product value of the wood from the excellent site is ten times greater ($3,214) than that of the poor site ($308)! If only the stem count was used in this hypothetical case, a SI=60 mine soil would have been judged as productive as the SI=100 mine soil. This is clearly not the case. Final product value of the harvested wood is the reclamation productivity standard that should be used to judge reforestation success. This is analogous to using the value of bushels of corn, or soybeans, or tons of hay harvested from reclaimed agricultural land. Productivity standards for forest land must be based on attributes of mine soils that result in productive forest sites. In Virginia, preliminary studies show that soil strength (exploitable mine soil volume) and chemical toxicities (total salts) (Torbert et al. 1988), and nutritional imbalances caused by pH extremes (Torbert et al. 1989; Moss et al. 1989) most limit the growth and development of forest trees.

Required reclamation practices in the Appalachian region appear to be detrimental to tree establishment and long term growth. Hauling, grading, and "tracking-in" compact mine soils, severely impeding normal root growth and nutrient and moisture uptake. Topsoil substitutes with a near-neutral pH are often selected for the surface to avoid liming and to ensure vigorous growth of ground cover that is also heavily fertilized. Trees, especially coniferous species, do best at a pH between 4 and 6 and require little to no fertilization. Vigorous growth of ground cover during the first two years suppresses tree establishment and often results in poor survival or complete failure. This problem is further addressed by Torbert and Burger (1990) in another paper in this proceedings.

## Summary

Developing economically-viable post-mining forests requires little or no additional effort or expense beyond that required for initial site stabilization. If site requirements and reclamation procedures are specifically geared for this land use, highly productive forest land can be created in a timely manner. If post-mining land uses are permitted based on their probable ultimate use, a higher proportion of the mined acres will be reforested at the outset.

High quality sites and timely reforestation are simple concepts; if these concepts were pursued to their fullest potential, the economic value of mined land and restored forests could be considerably improved. As the regional economy ultimately makes the transition from its coal base, communities will again turn toward the forest as a valuable, renewable resource in which to live, recreate, and derive income.

Table 1. Comparison of mine soil quality and forest production and value.

| Mine Soil Depth[1/] | Site Index | Volume[2/] | Wood Products | Total Value |
|---|---|---|---|---|
| (inches) | (age 50) | (MBF) | | ($/acre) |
| 11 | 60 | 5.0 | mine props/pulp | 308 |
| 24 | 80 | 14.2 | small sawtimber | 920 |
| 43 | 100 | 32.1 | large sawtimber | 3,214 |

[1/]Torbert et al., 1988.

[2/]Million board feet (MBF)

Literature Cited


Beck, D. E. 1971. Polymorphic site index curves for white pine in southern Appalachians. U.S.D.A. For. Serv. Res. Paper SE-80. S.E. For. Exp. Stn., Asheville, NC.

Burns, R. M. 1983. Silvicultural systems for the major forest types of the United States. U.S.D.A. For. Serv. Agric. Handb. 191 p. Washington, D.C.

Clements, F. E. 1916. Plant succession. An analysis of the development of vegetation. Carnegie Inst. Wash. Publ. 242. 512 p.

Donnell, R. D. 1987. A microcomputer program to analyze wood supply and economic feasibility of wood processing facilities. M.S. Thesis. 156 p. Vir. Poly. Inst. and State Univ. Blacksburg.

McCoy, T. S. and H. W. Wisdom. 1984. Feasibility of small mill investments for utliizing diameter hardwood from coal lands in in southwestern Virginia. Forest Products Journal 34:43-48.

Moss, S. H., J. A. Burger and W. L. Daniels. 1989. Pitch x loblolly pine growth in organically amended mine soils. J. Environ. Qual. 18:110-115.

Robles, J. and H. Wisdom. 1989. The economic potential of southwestern Virginia's forests. p. 25-35. In: Powell River Project Symposium. Vir. Poly. Inst. and State Univ. Blacksburg.

Torbert, J. L., J. A. Burger and W. L. Daniels. 1986. The importance of overburden selection for reforestation of mine soils. Virginia Coop. Ext. Ser. Pub. 460-111.

Torbert, J. L., J. A. Burger and T. J. Nichols. 1986. Reforestation seed mixtures for hydroseeding reclaimed mined land. Virginia Coop. Ext. Ser. Pub. 460-112.

Torbert, J. L., A. R. Tuladhar, J. A. Burger and J. C. Bell. 1988. Minesoil property effects on the height of ten-year-old white pine. J. Environ. Qual. 17:189-192.

This page intentionally left blank.

# The Feasibility of Large-Scale Sewage Sludge/Compost Utilization on Central Appalachian Surface Mined Lands[1]

W. Lee Daniels and Kathryn Haering[2]

Abstract The productivity of Appalachian mine soils is frequently limited by organic matter and associated low levels of fertility, aggregation and water holding capacity. Municipal sewage sludge has been used in various forms as a surface amendment in small plot studies in the Appalachian region, and results indicate it to be a superior soil treatment for long-term revegetation success. In April 1989, we initiated a large-scale research/demonstration project utilizing "mine mix" (wood chip compost + sludge cake from Philadelphia) as a surface amendment on a 150 ha active surface mine site in Wise County Virginia. The operation was permitted as a routine revision to a current mining permit. Between June and August 1989, approximately 26,000 wet Mg of mine mix were hauled to the site by rail and spread over the site at an average loading rate of 112 Mg/ha (dry weight as cake). The treatment was incorporated with a chisel plow to a depth of 20 cm. The mine soils were mapped at a scale of 1:3000 before sludge application, and the maps were then used to determine treatment areas, set-backs, hot spots and wetlands. Surface soils were sampled for lab analyses, and the average unlimed pH across the site was 5.7. An intensive water quality monitoring program over the period revealed no impact of the sludge application on surface or groundwater through January, 1990. This paper focuses on the environmental implications, regulatory framework, public relations, and logistical aspects of this project.

Additional key words: Water quality, nitrate movement, mine soil acidity.

## Introduction

The successful long-term reclamation of mined lands is dependent on many factors; one of the most important is building and maintaining mine soil organic matter reserves. Recent research (Roberts et al., 1988a, 1988b) in the Powell River Project watershed in southwestern Virginia has demonstrated the superiority of municipal sewage sludge as a mine soil

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]W. Lee Daniels is an Associate Professor and Kathryn Haering is a Research Associate in the Department of Crop and Soil Environmental Sciences, Virginia Polytechnic Institute and State University, Blacksburg, VA 24061-0404.

amendment. The mining industry faces serious financial and legal consequences if their reclamation efforts are not successful for a full five-year period after mining, allowing reclamation performance bonds (>$10,000 per acre) to be released. Local municipalities and waste water authorities also face an ever-growing cost of sludge disposal. Thus, the development of a mined land sludge utilization program would have obvious economic and environmental benefits for both the private and public sectors.

Despite the obvious advantages of sludge, the implementation of a sewage sludge utilization program on central Appalachian mined lands has been hampered to date by a number of factors. First, the region simply has not produced enough sludge in recent years to justify the economic and regulatory expense of developing, demonstrating, and monitoring an operational-scale sludge utilization project, or a regional mined land mapping/interpretive program. A second factor has been the dispersed nature of the region's treatment facilities, and a third factor has been the lack of carefully documented sludge trials on mined lands in this region. As a result of these factors and others, the protocols and technologies for the utilization of sewage sludge on southwest Virginia mined lands in an environmentally sound manner have not been developed.

We recently completed a five-year study of sewage sludge/plant growth dynamics (Roberts et al., 1988a) on southwestern Virginia mine soils which proved sewage sludge to be a superior long-term reclamation treatment. A major sewage sludge management firm, Enviro-Gro Technologies, indicated interest in the potential of this region for utilization of its "mine mix" sludge product. We have therefore executed a cooperative research/demonstration project among Virginia Tech, The Virginia Center for Innovative Technology (CIT), and Enviro-Gro to develop the necessary technologies and protocols for the environmentally sound use of municipal sewage sludge on Virginia mined lands.

## Objectives

1. To evaluate the logistic, economic, and environmental feasibility of sewage sludge utilization on southwest Virginia mine lands through the development and monitoring of an operational-scale pilot project in the Powell River Project research watershed.

2. To carefully monitor the effect of sludge applications on long-term surface and ground water quality around the utilization area.

3. To determine optimal sludge application rates to insure reclamation success and protect environmental quality.

4. To develop a mined land classification and evaluation system for sludge utilization purposes.

5. To evaluate the potential mined land base in the Virginia coalfields that is suitable for sludge utilization.

## Overall Approach and Cooperators

Considerable scientific research has been conducted to date on the interactions among sewage sludge, mine soil properties, plant growth, and environmental quality. Basically, we believe that as long as the sludge loading rate is appropriate, mine soil pH is maintained above 5.5, and water flow across the site is controlled, the advantages of sludge as a mine soil amendment far outweigh any potential adverse impacts. We have also completed exhaustive small-plot sludge studies on mined lands (Roberts et al., 1988a, 1988b) as have many others in the eastern United States (Sopper et al., 1982). Based on these combined findings, we felt that it was time to move ahead with a field-scale pilot utilization project, with research plots nested within the treated area, and adequate monitoring of the entire area to fully document soil/plant/sludge/water quality interactions.

An active surface mining operation (Red River Coal Co.) within the Powell River Project Research Area agreed to provide approximately 100 ha of newly reclaimed mine soils for the project. Of course, the full approval of the land owner (Penn Virginia Resources Corp.), the mining company, and the various regulatory agencies involved was required for such a venture to be successful. Penn-Virginia agreed in principle to the project, Norfolk-Southern Corp. agreed to haul the sludge compost for a reasonable rate, and Enviro-gro agreed to supply and handle the sludge materials, and to support the research/monitoring effort. The Virginia Division of Mined Land Reclamation approved a permit revision by Red River Coal to substitute a compost/sewage sludge mixture for normal fertilization. This permit revision was also reviewed by the Virginia Water Control Board and the Virginia Department of Health.

The Powell River Project site is ideal for this type of research effort for a number of reasons. First, we have numerous detailed studies on the mine soils, water quality, and land resources within this area over a long period of time (Daniels and Amos, 1981a, 1981b, 1982; Roberts et al., 1988c). The overburden and mine soils in this area are some of the best in the Appalachian region, and are generally non-acidic. Second, due to the active mining operation in the area, we have good surface runoff control, and historical baseline water quality data from sediment ponds, wells, and streams surrounding the site. Third, we have been involved in a cooperative monitoring program at the active mining site, and know the exact composition and characteristics of the spoils across the site.

## Materials and Methods

In order for applied research of this sort to be valid it must be conducted on a large enough scale so that we can truly evaluate the logistic and economic feasibility of handling large amounts of sludge materials in this rugged, remote terrain. For that reason we recommended that approximately 100 ha be the minimum pilot project area. An operation of this size allows us to observe and document the full-scale materials handling operations, and the effectiveness of the spreading techniques. It also permits us to

determine the impact of full-scale compost/sludge application on both short- and long-term water quality in the surrounding sediment ponds, receiving streams, and groundwater.

## Sludge Properties and Handling Procedures

The sludge material used for this project originated in Philadelphia, and was a 50:50 mixture of composted municipal sewage sludge (aerated static pile composting method) and stabilized anaerobic sludge cake. The average characteristics from 16 sieved samples taken in Philadelphia 1987 and 12 bulk samples taken in 1989 from the applied material are detailed in Table 1.

Table 1. Characteristics of Sludge "Mine Mix" Used in Study.

| | 1987 | 1989 |
|---|---|---|
| | Sieved (< 0.7 cm) | Unsieved |
| pH | 7.7 | 7.0 |
| %Solids | 53.00% | 46.00% |
| Total N | 1.90% | 1.20% |
| Total P | 1.70% | 0.70% |
| Total C | 28.00% | -- |
| K | 0.17% | 0.30% |
| Mg | 1.07% | .55% |
| Cd | 11 ppm | 3 ppm |
| Cr | 133 ppm | -- |
| Cu | 425 ppm | 294 ppm |
| Mn | 317 ppm | 455 ppm |
| Ni | 53 ppm | 73 ppm |
| Pb | 213 ppm | 159 ppm |
| Zn | 749 ppm | 562 ppm |

The actual spread material is approximately two-thirds composted wood chips, and one-third partially composted sludge cake on a dry weight basis. The mix was shipped by Norfolk-Southern via rail. A total of 635 coal cars were used to haul the mine mix to Norton, Virginia. The material was unloaded from the rail cars using a Hertzog (tm) loader (Fig. 1), and transported to the site via covered truck. The trucks were tarped and brushed off before they entered the public road. The mine mix was stockpiled on the reclamation site and surrounded by berms of spoil until spread with specially designed sludge spreaders (Fig. 2).

The mine mix materials were incorporated by chisel plowing (20 cm) and heavy disking immediately after spreading, and were seeded immediately (Fig. 3). Sludge was not spread within five m of outslopes, drainage ditches, sediment ponds, etc. Equipment time was carefully documented in an effort to determine interactions among surface mine location, topography, cost of application, and the overall practicality of large-scale sludge application in these landscapes.

Based on our small plot work, and on experience elsewhere (Sopper et al., 1982) we assumed that an average rate of 112 Mg/ha (as sludge) could be effectively incorporated into these spoils. Our actual average loading rate was approximately 100 Mg/ha as sludge, along with about 200 dry Mg of composted wood chips. We ran trials at the onset of spreading operations to verify this assumption. A replicated (four) loading rate study was installed during the spreading operation as discussed below. We sub-sampled the materials just before application for analysis and calculation of total loading rates.



Figure 1. The "mine mix" materials were unloaded from coal cars using specialized front-end loaders into haul trucks. Care must be taken in the selection of unloading facilities and hauling routes to avoid adverse public reaction.

## Description of Long-Term Research Program

Before the sludge was applied, we mapped the mine soils of the entire area in detail. This will allow us to determine the interactions of sludge and mine soil type over time. Every year, we will sample and analyze the standing vegetation throughout the treated area. We will concentrate on intensive soil and plant tissue sampling and analyses from research plots nested within the treated area. In these replicated plots we varied the sludge loading rates from 0 to 175 dry Mg per ha. Soil and plant tissue samples will be taken annually and analyzed for all plant macro- and micro-nutrients and all heavy metals of environmental concern. Additional soil-plant analyses will be performed as agreed to or specified by the regulatory agencies involved.

We are particularly interested in the influence of sludge on the more acidic spoil materials, since considerable research has indicated that it will buffer pH, and reduce iron and manganese availability. While the vast majority of spoils within this area are non-acidic, the operator does face a pre-existing acid discharge problem from previous mining activities, and some limited areas of acid spoils will be delineated and treated for the sake of the research effort.

One of the major concerns associated with large-scale sewage sludge utilization programs is the potential contamination of surface and

groundwaters, particularly by nitrate. Our water quality monitoring program is cooperative with, and in some cases in addition to, the regular monitoring program that Red River Coal carries out on the permit. Sediment ponds are sampled weekly, groundwater wells biweekly, and surface water instream sites monthly. A total of 12 sites are currently being sampled, and more will be added as mining is completed over the next year. A broad range of parameters have been analyzed at each location including pH, hardness, acidity and alkalinity, total-Fe and Mn, heavy metals, C, P, $NO_3^-$, and $SO_4^{2-}$.

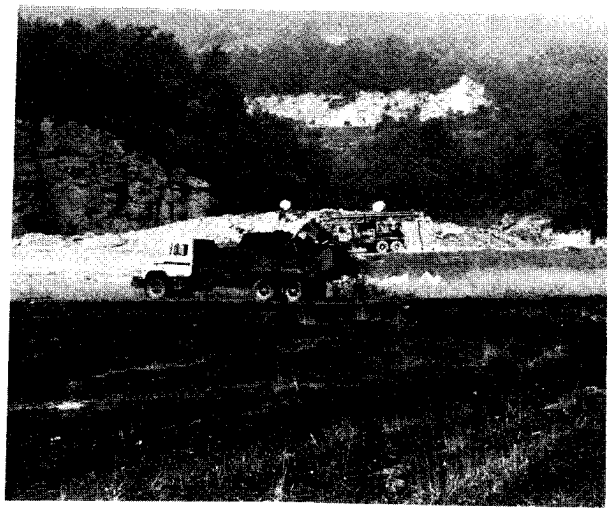


**Figure 2. The "mine mix" was applied by either spreader truck as shown or with a spreader unit pulled behind a farm tractor. The total loading rate was 350 wet tons per acre of mix, or 112 Mg/ha as sludge cake. The depth of the material was approximately 12 cm before deep incorporation.**

## Results and Discussion

Over the course of the summer of 1989, approximately 26,000 wet Mg of the mine mix product were applied to approximately 40 ha of the site. All incorporation for the 1989 season was completed by late October, and the area was seeded to a conventional hayland-pasture mix with a winter rye cover crop. The balance of the sludge was stockpiled, to be spread the following spring (1990). During this period of time we learned quite a bit about the logistics, mechanics, and potential pitfalls of large-scale sludge application and utilization in this kind of landscape. The balance of this paper will address our first-year findings and insights.

## Mine Soil Mapping and pH

Before sludge application, we mapped all areas at a scale of 1:3000 on standard black and white aerial photos. Contrasting mine soils were delineated based on rock type composition, slope class, depth to hard rock, wetness, and surface configuration. Further detail on the mapping criteria employed is given by Daniels and Amos (1981a, 1981b). These maps were then used to calculate the amount of suitable area across the site for sludge application. We found that mapping at this scale was essential to accurately delineate suitable treatment areas, isolate hot spots and wetlands, and delineate varying slope or roughness classes. We evaluated the use of larger scale (1:4800 and 1:6000) imagery for the site-specific mapping, but found it generally inadequate.

Since the total mined area was fairly large, we subdivided it into "treatment cells" which were labeled alphabetically, and the exact area of each cell was determined with an acreage grid. Large bulk samples of the mine soils were taken on a 50 m grid across the entire area to a depth of 15 cm. These samples were taken before sludge application, and will be taken every year thereafter for characterization of chemical, physical and mineralogical properties. The average soil pH of the areas treated in 1989 is given in Table 2.



**Figure 3. The material was incorporated with a chisel plow to approximately 20 cm+, and then disked with a heavy off-set disk. A hayland/pasture seeding mix was broadcast over the site after the final disking.**

The majority of overburden mined at this site is non-acid-forming, and tends to stabilize at pH values above 5.0. However, areas do exist across the site which contain varying amounts of acid-forming materials. These range in size from several hectares to several meters in diameter. The average pH values for areas D and F were pulled down by several very acidic hot spots which were sampled intentionally for characterization, while the pH of area E was uniformly low. Area E was limed at a rate of 10 Mg/ha prior to sludge application, and all hot spots within the entire application area were spot limed at a similar rate. We will continue to monitor soil pH across the site over time, but do not expect overly acidic conditions to develop. Soil pH will be maintained above 5.5 by liming if necessary.

## Water Quality Monitoring and Treatment Effects

Our water quality monitoring program will need to continue for several years before the long-term influence of the sludge treatment can

be adequately assessed, particularly the potential deep movement of nitrate to groundwater. The summer of 1989 was one of the wettest on record, however, and we were encouraged by the fact that under exceptionally heavy rainfall conditions we did not detect any surface run-off of sludge-related contaminants from stockpiles or spreading areas (Table 3).

---

Table 2. Average soil pH before application of sludge for various sites.

| Area | # of Samples | Average pH |
|---|---|---|
| A | 10 | 5.65 |
| B | 13 | 5.96 |
| C | 19 | 5.81 |
| D | 21 | 5.24 |
| E | 7 | 4.65 |
| F | 8 | 5.18 |
| G | 5 | 5.42 |

---

The total-N loading applied to this area was quite high in an effort to supply organic matter and plant nutrients for the five-year bond release period and beyond. However, the overall C:N ratio of the material is above 20:1 due to the large component of wood chips, and we do not anticipate major nitrate leaching. Other mined land work in Pennsylvania (Sopper et al., 1982) with heavy loading rates has shown minimal or no off-site movement of nitrogen. We believe that large amounts of the total-N loading will also be lost via volatilization and denitrification pathways, and we will attempt to verify this hypothesis. We also believe that the moderate native pH of these spoils will limit any potential heavy metal uptake into plant tissue. Again, we will verify this assumption over time.

Logistical Concerns

The concept of using empty railroad cars to back-haul sludge materials from urban areas to the coalfields is certainly attractive, but is not without its pitfalls. First, the waste treatment site must have a rail siding adequate for loading the sludge into rail cars; many do not. Second, an adequate siding must be located within a relatively short hauling distance of the application site, and it must be large enough to allow for many cars to be processed without interfering with normal coalfield traffic. We incorrectly assumed that sidings associated with coal prep plants near our site would be adequate for this function, but they were not large enough to handle our traffic without affecting coal traffic to and from the tipple. The siding we finally used was associated with an old coal crushing facility approximately seven miles from the site, and was not an ideal location. The loading operation in Philadelphia was capable of filling approximately 15 cars per day, which were combined into units of 40 or more and shipped in batches. This process, combined with normal delays in shipment and bad weather made it difficult to coordinate the short-haul trucking to the site with any regularity. For this and other reasons, the short-haul trucking accounted for almost one-third of the total cost of transporting the materials. Despite these difficulties, between June 6 and August 25, we were able to completely move 635 rail cars of material to the site. The total cost of transportation is proprietary information at this time, but the costs did fall within the expected range of the sludge management firm involved (Enviro-Gro Technologies).

---

Table 3. Levels of nitrate, pH and total iron at 3 water sampling locations typical of the project area. Site GW-2 is discharge from a hollow fill, Pond 2 is a large sediment pond which receives surface drainage, and the Powell River location is just downstream from the treatment area, on a major perennial stream draining the site. All values are a mean of 5 observations.

| | ------sampling date (1989)------ | | | | |
|---|---|---|---|---|---|
| Location | Pre-Sludge | Jun | Jul | Aug | Sep |
| | ----- NITRATE (PPM) ---- | | | | |
| GW-2 | 4.35 | 2.34 | 1.88 | 2.29 | 2.21 |
| Pond 2 | 1.92 | 2.19 | 1.71 | 1.73 | 1.20 |
| Powell R. | nd | 0.58 | 1.80 | nd | -- |
| | ---------- pH ---------- | | | | |
| GW-2 | 6.5 | 6.6 | 7.5 | 7.4 | 7.0 |
| Pond 2 | 6.9 | 6.8 | 7.4 | 7.4 | 7.6 |
| Powell R. | 6.8 | 7.5 | 7.5 | -- | -- |
| | ----- TOTAL-FE (PPM) ----- | | | | |
| GW-2 | 1.9 | -- | 0.2 | 0.7 | 0.6 |
| Pond 2 | 0.6 | 0.8 | 0.6 | 0.2 | 0.2 |
| Powell R. | 0.7 | 0.5 | 0.1 | -- | -- |

---

By mid-June 1989, sufficient sludge had been hauled to the site for application procedures to begin. The spring and summer of 1989 were very wet, however, and the imperfectly drained benches would not support the spreading vehicles for a large part of the summer. This experience pointed out the great importance of detailed mapping, particularly the accurate delineation of seasonal wet spots and ephemeral drains. Once the benches dried sufficiently to support spreading operations, up to 1,000 wet tons of mine mix could be spread per day (Fig. 2).

Incorporation was completed much more quickly than the spreading, and the areas usually received two to three passes with the chisel plow (Fig. 3), and two passes with a heavy off-set disk. A 4WD 85 horsepower tractor was used for incorporation operations. Through experimentation, we found that the mine mix could be spread and incorporated on up to a 15% slope on highwall backfills, and down slopes in excess of 20% when equipment could be driven to the top of a fill slope and then down.

Public Relations and Regulatory Issues

The land application of sewage sludge

materials is often controversial, regardless of the locale of operations. Despite the fact that this project was heavily publicized in the media, several citizens along the direct-haul truck route were not aware of the project until the actual unloading and hauling began. These individuals became quite irate and complained vigorously to all regulatory agencies involved. Interestingly enough, the citizens were not nearly as concerned about the environmental and water quality implications of the land application program as they were about human health issues such as the potential for AIDS transmittal. Obviously, in order for such a research-demonstration operation to be successful, the local community must be completely informed of the objectives, methods, and safeguards employed. Beyond these measures, however, we also recommend that great efforts be made to keep all unloading and hauling away from public roads whenever possible.

From a regulatory perspective, there are obvious advantages to incorporating sewage sludge utilization into active mining permits since the mining agencies generally have jurisdiction over NPDES discharge permits and other water quality programs. The routine sampling associated with the water quality monitoring program for an active permit can be easily modified to include nitrate and other parameters of interest such as heavy metals. However, this is not the case when sludge is used as a treatment on abandoned or bond-released mined lands. In this instance, a "conventional" sludge application permit must be obtained through the relevant state agencies, and the cost of permitting and monitoring is much higher. It should also be noted that currently proposed EPA regulations for land application of sludge materials could severely limit the loading rate to mined lands, and would severely affect the economics of operations such as this.

## Conclusions

Based on our experience with this project, we believe that the widespread use of municipal sewage sludge materials is logistically and economically feasible in this region. Detailed mapping and characterization of the site before application are essential to correctly specify the total suitable acreage, and to identify areas which will pose limitations to spreading equipment. To date, no influence of the sludge application effort has been detected in ground or surface waters around the site, but long-term monitoring is essential for a project such as this. Additional monitoring of soil and plant tissue nutrient and heavy metal levels must be continued for multiple seasons before the overall environmental impact of the treatment can be adequately assessed. If the sludge is to be back-hauled to the site in rail cars, careful attention must be paid to the development of an adequate unloading facility which is as close to the spreading site as possible. Hauling on public roads should be avoided whenever possible, and local citizens must be kept constantly informed of project objectives, methods, and monitoring efforts.

The implementation of full-scale sewage sludge utilization on southwest Virginia mined lands could lead to markedly improved reclamation and associated improved local water quality. The use of sewage sludge will provide an immediate financial benefit to the mining industry through reduced fertilizer, lime, and reseeding costs, and a significant long-term economic benefit through greatly enhanced probability of timely release of reclamation performance bonds. The project will also be of great value to the expanding sludge utilization industry. Furthermore, the completion of this effort will supply local waste water authorities with an economically viable alternative for the use of their sludges.

## Acknowledgements

This project was funded by Enviro-Gro Technologies Inc. and the Virginia Center for Innovative Technology. We also want to express thanks to Penn-Virginia Resources, the Blackwood Land Company, Red River Coal, Environmental Monitoring Inc., the Norfolk Southern Corp., the Virginia Division of Mined Land Reclamation, the Virginia Dept. of Health, the Virginia Water Control Board, and the Philadelphia Water Department for their support and cooperation. Finally, we thank Jon Rockett for his countless hours of negotiations on our behalf, and Barry Stewart for assisting with the preparation of this manuscript.

## Literature Cited

Daniels, W. L. and D. F. Amos. 1981a. Mapping, characterization and genesis of mine soils on a reclamation research area in Wise County, Virginia. p. 261-275 *In:* Proc. 1981 Symposium on Surface Mining Hydrology, Sedimentology and Reclamation. (Univ. of Ky., Lexington, KY)

Daniels, W. L. and D. F. Amos. 1981b. Minesoil Maps and Legend and Mapping Unit Interpretations of the Powell River Project Reclamation Research Project. Department of Agronomy, VPI & SU, Blacksburg, VA. 43 p.

Daniels, W. L. and D. F. Amos. 1982. Chemical characteristics of some SW Virginia minesoils. p. 377-381 *In:* Proc. 1982 Symposium on Surface Mining Hydrology, Sed. and Reclamation. (Univ. of Ky., Lexington, KY)

Roberts, J. A., W. L. Daniels, J. C. Bell and D. C. Martens. 1988a. Tall fescue production and nutrient status on Southwest Virginia mine soils. J. Environ. Qual. 17:55-62.

Roberts, J. A., W. L. Daniels, J. C. Bell and J. A. Burger. 1988b. Early stages of mine soil genesis as affected by topsoiling and organic amendments. Soil Sci. Soc. Am. J. 52:730-738.

Roberts, J. A., W. L. Daniels, J. C. Bell and J. A. Burger. 1988c. Early stages of mine soil genesis in a SW Virginia mine spoil litho-sequence. Soil Sci. Soc. Am. J. 52:716-723.

Sopper, W. E., E. M. Seeker, and R. K. Bastian. 1982. *Land Reclamation and Biomass Production with Municipal Wastewater and Sludge.* 524 p. The Penn State Univ. Press, State College, Pa..

# OCCURRENCE OF GROUND WATER IN MINE SPOIL, A RENEWABLE RESOURCE: STAR FIRE TRACT, EASTERN KENTUCKY[1,2]

by

**James S. Dinger, David R. Wunsch, and John E. Kemp[3]**

**Abstract.** The Star Fire site is a large surface mining operation in eastern Kentucky owned and operated by Cyprus-Southern Realty. Approximately 10,000 acres of reclaimed, usable land will result from the ongoing mining operation by the year 2010. Ultimately, development on the tract will depend on the availability of an adequate water supply utilizing both surface and ground waters. The water resources of the first 1,000-acre tract of mine spoil have been investigated. Areas of ground-water recharge and discharge have been identified at the site. Major recharge enters the spoil by way of disappearing streams, ground-water flow from bedrock that is in contact with the mine spoil, and specially designed infiltration basins. Ground water discharges predominantly from springs and seeps along the western outslope of the spoil. Monitoring wells drilled into the spoil indicate a saturated zone ranges from 8 to 25 feet in thickness. Data from water-level measurements, dye-tracing, and geochemical analyses of water samples indicate that a compacted haul road acts as a low-permeability barrier to ground-water movement.

Water samples from surface and ground water taken from the site show that all waters are a Ca-Mg-sulfate type, differing only in the total concentration of these constituents. Specific conductance measurements of samples collected at the site ranged from 257 microsiemens in surface runoff to 3,725 microsiemens in a monitoring well situated in the interior of the spoil. The pH ranged between 6.23 and 6.66 for ground water sampled at the site, and between 7.67 and 8.28 for surface water.

Additional Key Words: water supply, ground water, spoil aquifer, infiltration basin, base flow, surface water, ground-water quality

## Introduction

In 1988 Kentucky produced 161 million tons of coal, ranking number two in the nation. However, the work force generating this production has been reduced approximately 30 percent since 1980, and unemployment among the general work force is estimated to be in the 10 to 20 percent range. A principal cause for this high unemployment rate is the lack of economic diversification. Two of the major factors that hamper economic diversification are the lack of water supplies and rugged topography, particularly in the Eastern Kentucky Coal Field, where local relief ranges from 300 to 2,500 feet. Water supplies sufficient to support economic development are restricted to the larger streams, and even these supplies must be enhanced by construction of expensive reservoirs. Rugged terrain also makes it difficult to pipe water from the source of treatment to outlying areas.

Many coal companies operating in eastern Kentucky have no real interest in these limiting factors because they generally own only the mineral rights to the properties they mine, not the land surface. Cyprus Mountain Coals, a subsidiary of Cyprus Minerals, Inc., is unique in that they own, fee simple, the 17,000 acres at the Star Fire surface mine and therefore have considerable interest in post-mine development of the property. It is estimated that 10,000 acres of gently rolling land will be created by the year 2010 through the mining of coal by mountaintop removal techniques, thus providing a site for new land uses and future economic development.

This paper discusses the concepts of developing a water supply for the site. It documents the initial planning and construction phases of building and recharging an aquifer in the spoil material generated during mining, and it reviews initial ground-water quantity and quality data measured at the site between March and September 1989.

## Hydrogeologic Setting

The Star Fire Mine encompasses portions of Breathitt, Perry, and Knott Counties and is located approximately 5 miles northeast of Hazard, Kentucky, near the Daniel Boone Parkway (KY-80) (Fig. 1). Regional geology of the site is mapped on the Noble (Hinrichs, 1978) and Vest (Danilchik and Waldrop, 1978) 7.5-minute geologic quadrangle maps. The coal zones being mined are the Hazard Nos. 7, 8, 9, and 10 beds (Fig. 2), all of which are included in the Breathitt Formation of Pennsylvanian age. These beds range in thickness from 3 to 7 feet, and several of the zones contain rider coals that are also mined. Overall, these coals can be characterized as high-grade, low-sulfur, bituminous. The overburden at the site consists of calcareous sandstones and shales. Mine permit data indicate that the potential for acid-mine drainage is low because of the high net neutralization potential of the overburden lithologies at the site.

---

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]Publication in these proceedings does not preclude authors from publishing their manuscripts, whole or in part, in other publication outlets.

[3]James S. Dinger, Ph.D., Head of Water Resources Section, and David R. Wunsch, hydrogeologist with the Kentucky Geological Survey; John E. Kemp, graduate student at the Department of Geological Sciences, University of Kentucky, Lexington, Kentucky 40506-0107, (606) 257-5500.

Figure 3 is a map showing the major drainages and the sequence of mining through the years 1989, 1995, 2002, and 2007 for Jobs 5 and 7 within the Star Fire Mine. The years indicate the completion date for that section of the mine; each job represents approximately 5,000 acres. The computer model Sediment, Erosion, Discharge by Computer Aided Design (SEDCAD+) (Schwab, 1987; Schwab and Warner, 1987) was used by Dinger and others (1987) to estimate annual total runoff for the site in 1995 and 2007 (these dates represent the completion of mining for major sections at both job sites). The model estimated an upper limit of 2,770 and 2,960 acre-feet of annual runoff would be generated in those years, respectively.

Mountaintop removal mining methods are being used at each job site, with primary emphasis presently being placed at Job 5. At Job 5, truck-and-shovel techniques are recovering the No. 9 and No. 10 coals, and a 64-yard$^3$ bucket dragline is employed to mine the No. 7 and No. 8 seams. In the process of mining, a 200- to 300-foot-thick spoil backfill is being created.

Three major watersheds drain the 17,000-acre site: Buckhorn Creek (8,019 acres), Long Fork (5,129 acres), and Lick Branch (2,049 acres) (Fig. 3). Field observations indicate that these streams are flashy and have low flows during the summer months. These characteristics do not lend themselves to a reliable long-term water supply upon which significant economic development can depend.

## Ground-Water Considerations

### Introduction

In order to substantially increase the amount and seasonal reliability of available water at the site, ground water from within the spoil material is being investigated. If a substantial aquifer can be developed within the spoil material, it may serve two purposes: (1) provide a unique ground-water supply, and (2) provide base flow to the major drainage basins and subsequent surface reservoirs during drought conditions, which generally prevail in eastern Kentucky between May and October.

Several elements of the spoil have been studied at Job 5 since August 1987, in the section of the mine completed between 1981 and June 1989 (Fig. 3). These elements are: (1) creation of coarse-rock zones within the spoil for the recharge and movement of ground water (i.e., the creation of an aquifer framework), (2) design and construction of infiltration basins to enhance recharge to the ground-water reservoir, and (3) characterization of ground-water quality and quantity in the spoil material. Consideration of these elements comprises the remainder of this paper.




Figure 1. Location of the Star Fire Mine.

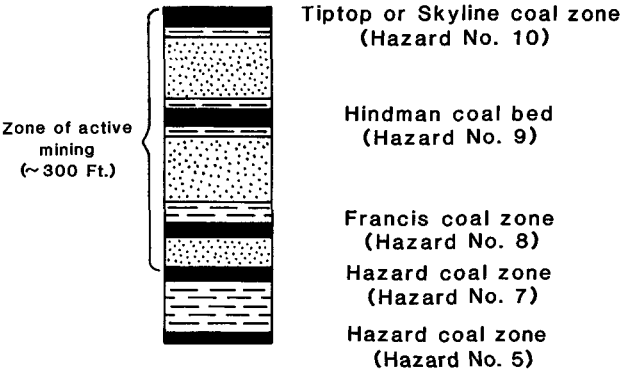



Figure 2. Schematic geologic column of site.

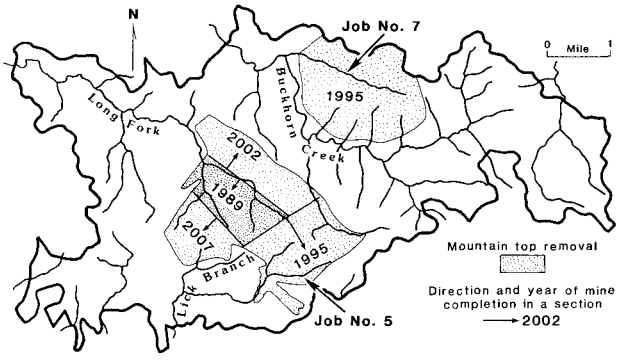



Figure 3. Map of the drainage basins and sequence for mining Job 5.

## Spoil Aquifer Framework

Dinger and others (1988) provided a detailed description of the spoil material and its potential as aquifer material. Coarse rubble zones (Fig. 4, features A, B, C, and F; Fig. 5) provide avenues for ground-water movement. The finer sized or compacted material between the coarse rubble (Fig. 4, features C, D, E, and G) may provide for ground-water storage once saturated or may even act to significantly retard the movement of ground water within the spoil.

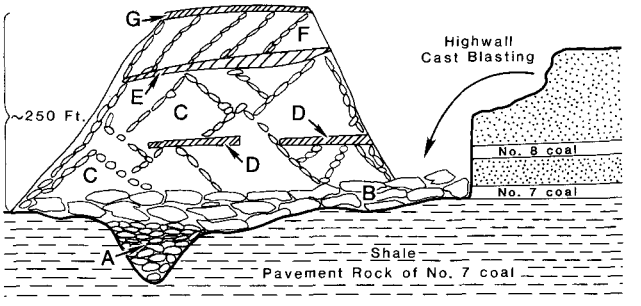



Figure 4. Schematic cross section showing components of spoil significant to the development of an aquifer framework: A = hollow fill, B = cast-blasted rubble, C = dragline spoil, D = dragline pad, E = temporary haul road, F = end-dumped spoil, and G = final graded land surface.

### Infiltration Basins

The final graded land surface is a compacted continuous surface (Fig. 4, feature G) that has been observed to prohibit surface water from infiltrating into the spoil material. In addition, future development of the site might include construction of impervious structures

such as paved roads, parking lots, and buildings that would substantially hinder recharge to the ground-water system. With these conditions in mind, infiltration basins constructed in the spoil may be the best way to facilitate ground-water recharge. Therefore, special spoil-handling techniques are being pursued to capture surface runoff from the land surface for recharge to the ground-water system within the spoil material.

Two alternative infiltration basin designs were proposed for the Star Fire Project (Dinger and others, 1988). The two designs differ only in the depth of the rock chimney, one being shallow (approximately 70 feet deep) and the other deep (approximately 250 feet deep). The selection of the shallow alternative was made based upon existing mining operating procedures that leave an open space approximately 70 feet deep between spoil cones (Fig. 5). A selectively filled rock chimney to this depth may be sufficient to bypass the potential near-surface confining layer within the spoil (Fig. 4, features E and G). This alternative may prove to be as effective in recharging the spoil as the deeper option, but more economical to construct.

Selection of the deep option was based on the premise that such an infiltration basin will function as a direct connection to the rubble zone resting on top of the No. 7 coal underburden (Fig. 6). This rubble zone is created by blast casting the interburden rock from the highwall into the pit from which the No. 7 coal bed has been removed. The extensive rock chimney constructed to this zone bypasses all intermediate compacted zones within the spoil that might tend to perch percolating ground water. To date, only this deep basin has been installed, but the final surface grading has not been completed. Therefore, the basin is receiving only limited recharge from surface runoff.

Evaluation of these designs will provide data to evaluate a tradeoff between recharge capability and construction costs. Over the next several years this research should reveal many interactions among infiltration basin design, ground-water recharge rate and quality, required contributing watershed area, number and location of infiltration basins, and cost of installation.

## Water-Monitoring Methods

In order to characterize the hydrogeology of the spoil aquifer, several research methods were utilized. These included: (1) precipitation measurement, (2) discharge measurement on streams and springs, (3) installation of monitoring wells in spoil, (4) dye tracing in spoil, and (5) water-quality analysis.

### Precipitation Measurement

Daily precipitation data from the Robinson Forest Camp Gage was provided by the University of Kentucky, Department of Forestry. The camp gage is a standard 8-inch, non-recording gage located 3.5 miles from the study area (Fig. 3).

### Discharge Measurement

Instantaneous discharge measurements were made at three springs (S#1, S#2, and S#3) discharging from the spoil, and where Chestnut Gap Branch flows into the base of the spoil (swallet) in a manner similar to that of a disappearing stream in karst areas (Fig. 7). These springs were monitored to characterize the variability of discharge in relation to recharge events, determine the sources of recharge to the spoil, and determine the nature of ground-water flow through the spoil. Chestnut Gap Branch was monitored to determine the amount of recharge it contributes to the spoil aquifer.

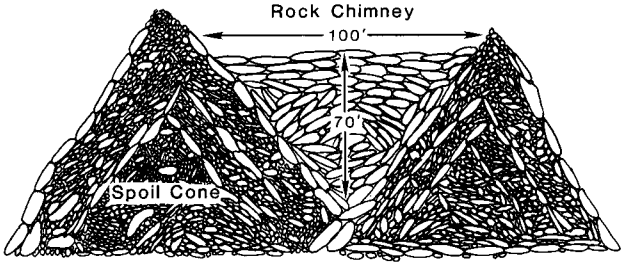



Figure 5. Schematic cross section of 70-foot-deep, V-shaped rock chimney for infiltration basin.

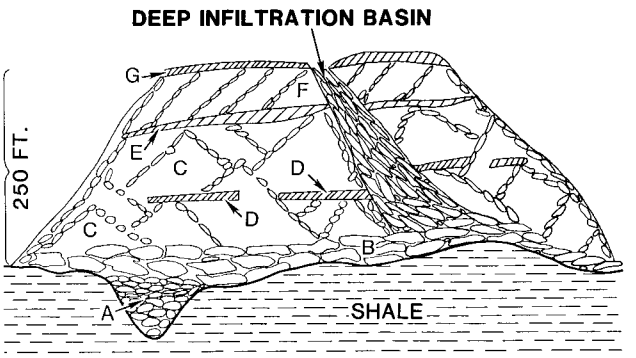



Figure 6. Schematic cross section showing components of spoil significant to the development of an aquifer framework: A = hollow fill, B = cast-blasted rubble, C = dragline spoil, D = dragline pad, E = temporary haul road, F = end-dumped spoil, and G = final graded land surface.

## Monitoring Well Measurements

Three monitoring wells (MW#1, MW#2, and MW#3) have been installed in the spoil to study the development and fluctuations of the water table in the spoil, characterize the ground-water quality, determine the effectiveness of the deep infiltration basin, and determine the hydraulic properties of the spoil (Fig. 7).

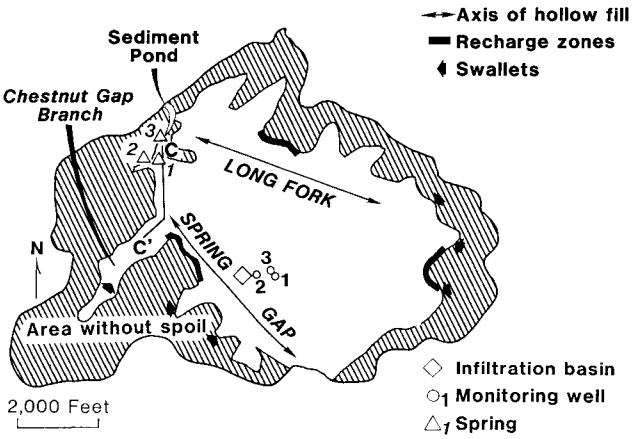



Figure 7. Map showing locations of ground water related to features.

## Dye Tracing

Dye traces using Rhodamine WT were conducted in an attempt to define flow paths through the spoil, and to determine travel times along these flow paths. Rhodamine WT has been widely employed in the study of karstic carbonate aquifers, and to a lesser extent in granular aquifers (Aulenbach and others, 1978). It exhibits many properties favorable for ground-water tracing, including detectability at very low concentrations, low toxicity, a distinct peak-emission wavelength, chemical stability over a wide range of pH values, photochemical and biological stability, and a low rate of adsorption (Smart and Laidlaw, 1977). For this study, the most critical of these factors is thought to be adsorption, because the dye is assumed to flow through an aquifer matrix rich in clays, organic shales, and ferric hydroxide. Ferric hydroxide was found to adsorb significant quantities of Rhodamine WT in an experiment designed to test fluorescent dyes for use in ground-water tracing in underground coal mines (Aldous and Smart, 1987).

Two dye traces were conducted; one trace was initiated at the Chestnut Gap Branch swallet and the other at MW#1 (Fig. 7). The three springs served as monitoring points using charcoal detectors. An additional detector was placed in Chestnut Gap Branch upstream from the dye introduction point in order to test for interference from organics. The elutriant from the detectors was analyzed with a Turner Model 10 filter fluorometer.

## Water-Quality Sampling and Analysis

Water samples from the three spoil springs, three monitoring wells, Chestnut Gap Branch swallet, and the deep infiltration basin were collected to establish water-quality characteristics and determine changes in water quality that occur between recharge and discharge points. Samples have been collected on a monthly basis since May 1989 for the springs and sporadically from well water as the monitoring wells were completed through the summer of 1989. Some samples were taken from springs and wells to study recharge events. Variables determined in most samples were temperature, specific electrical conductance, pH, sulfate, bicarbonate, calcium, magnesium, sodium, potassium, manganese, and iron. In some cases chloride and nitrate were measured.

# Results and Discussion

## Recharge

Field reconnaissance of the study area revealed numerous places where streams and storm runoff recharge the spoil aquifer (Fig. 7). Several streams were observed to flow directly into swallets at the toe of spoil slopes. The largest of these streams is Chestnut Gap Branch, a first-order stream with a watershed area of 0.32 mile$^2$.

A number of recharge zones (specific points in some cases) were observed on the spoil during storm runoff events. These zones occur at places where the spoil adjoins highwalls or natural bedrock slopes, or in places where spoil handling resulted in boulder zones being locally exposed at the surface. The largest of this type of recharge zone is located in the northeastern part of the spoil, where a depression approximately 150 feet deep has been left open since 1982 (Fig. 7, point A). Storm runoff was observed to flow into this pit and then rapidly disappear into the spoil. Likewise, the deep infiltration basin has been partially functional in this regard, although its present watershed is limited at this time.

Recharge also occurs when precipitation falls directly on ungraded dragline-cast spoil cones. The size of this area varies depending on the amount of grading that has occurred, but an extensive area, often 2 million feet$^2$ or larger, is always present.

In addition to these major points of recharge, numerous small cracks and fissures in the spoil surface were observed to capture lesser amounts of storm runoff. Another source of recharge to the spoil aquifer is the contribution from the bedrock aquifer in places where it is in contact with spoil.

Infiltration through the spoil surface is not thought to account for a significant amount of recharge because of the compacted nature of the graded spoil. Drilling and excavation have shown that the spoil is dry within a few inches of the surface, and storms have been observed to quickly form a thin sheet of mud on the surface, which indicates that rainfall does not easily infiltrate the spoil. This situation may change as vegetation re-establishes itself and aids in the development of soil profiles, resulting in a more porous surface structure.

## Ground-Water Movement

Knowledge of ground-water movement has been gained by examining discharge hydrographs and dye tracing to the springs, through water-level measurements in the spoil monitoring wells, and through ground-water quality determinations made at both the springs and wells.

**Spring Discharge.** The most significant area of observed discharge is a group of three springs located at the northern toe of the Spring Gap Branch hollow fill (Figs. 7-8). This area has remained swampy throughout the time of field investigation. Spring No. 1 (S#1) is the largest of the three springs, and discharges at an approximate elevation of 1,040 feet. The discharge point is located at the toe of a 130-foot-thick lift of end-dumped sandstone spoil from the recent mining operation (1981-1989). This spoil joins and partially overlaps a much lower permeability hollow-fill material from previous mining (1950-1960's) and a 45-foot-thick lift of end-dumped shale (1982), which is considered to have a lower permeability than the sandstone spoil (Fig. 8). During times of extremely high discharge, a number of small springs have been observed to crop out along the toe of this lift at an elevation equivalent to S#1. Spring No. 2 (S#2) boils up from a small hole located directly in the middle of the 45-foot lift, about 200 feet from S#1 (Fig. 7), at an approximate elevation of 1,030 feet. Spring No. 3 (S#3) discharges from a boulder zone along the outslope of the 45-foot lift at an approximate elevation of 1,010 feet (Fig. 8).

In contrast to the Spring Gap hollow fill, discharge was not observed from the toe of the Long Fork hollow fill. It is believed that ground water is flowing through this hollow fill, but is discharging directly into the sediment pond at a point below the water level of the pond (Fig. 8).

Ground water has also been observed to discharge from the spoil into the active dragline pit when the pit is at the level of the No. 7 coal bed. During the past 6 years, ground water has been discharging from the spoil into the active pit at a rate high enough to require pumping on a daily basis. On occasion, pumping rates have reached an estimated 0.56 cfs (360,000 gallons per day). Discharge measurements range from 1.10 to 6.05 cfs for S#1, 0.13 to 0.41 cfs for S#2, and 0.58 to 1.37 cfs for S#3 (approximately 1 to 5 million gallons per day) (Fig. 9). Spring No. 1 is by far the largest of the three springs.

Flow at S#3 seems to correspond to S#1, whereas S#2 generally flows at a steady rate, independent of recharge events. The rapid response of S#1 and S#3 to precipitation and their location at the base of the mine spoil are good indicators that these springs are discharge points for ground water moving through permeable spoil. Because of its location close to the bedrock valley wall and its steady flow, it can be inferred that S#2 is ground water coming primarily from the bedrock through the low-permeability shale spoil that was dumped in this general location in 1982 (Fig. 8).

The difference in discharge between S#1 and S#3 also provides evidence for the effectiveness of the low-permeability shale fill in reducing ground-water discharge from S#3. Spring No. 3 is at a lower elevation than S#1, but has a lower rate of discharge. If a flow barrier did not exist, the majority of discharge would be expected to occur at S#3 instead of S#1.

Spring No. 1 was observed to respond within an hour after the 1.37-inch storm event started, and after only 5 hours its discharge was measured at 70 percent of peak flow (Fig. 10). The peak dis-

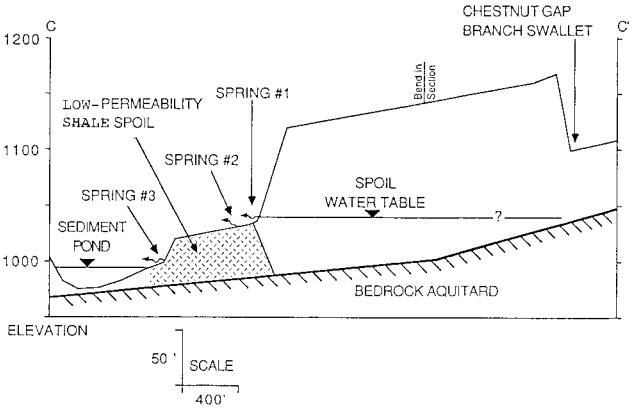



Figure 8. Cross section C-C' showing the configuration of the springs, spoil, and bedrock along the axis of the Spring Gap Branch hollow fill.

charge occurred between 29 and 50 hours after the storm began. This indicates that the spring responds quickly to recharge from a storm event. Such rapid flow through the aquifer suggests that the storm pulse moves along discrete high-porosity conduits composed of loose boulders similar to those described by Carrucio and Geidel (1984).

Figure 10 also indicates that the discharge recession for this storm event is slow and extended, in contrast to the rapid rise and peak in discharge. Measurements recorded on June 11, 4 days after the peak discharge, show that S#1 was discharging at a rate equal to 80 percent of the peak. The fact that the spring does not recede as fast as it rises indicates that the spoil has the ability to store a significant amount of ground water and release it over an extended period of time. A similar hydrograph response was observed at S#1 after a 3-day storm event that began on June 13 (Fig. 9).

The source of recharge to the spoil aquifer can be analyzed by studying the relationship between S#1 and Chestnut Gap Branch, the largest stream recharging the aquifer. Figure 11 shows that two storms in June produced similar hydrographs at both gaging locations, reflecting the close relationship between recharge from surface runoff and peaks in spring discharge. However, the volume of water discharged from S#1 consistently exceeds the input from Chestnut Gap Branch, indicating that this stream is not the sole source of recharge to the spring.

Through the process of hydrograph separation and developing recharge:area ratios from the spring hydrographs, it was determined that 90 percent of the flow at S#1 can be accounted for by groundwater recharge from Chestnut Gap and Spring Gap Branch basins. This indicates that the large area of mine spoil to the east is not contributing significantly to the spring's discharge.

**Monitoring Well Measurements.** In an effort to study the ground water in the large mine spoil area to the east of the springs, a field of monitoring wells has been proposed. Presently, three wells have been drilled (Fig. 12). Water levels in the wells verify that a water table does exist within a

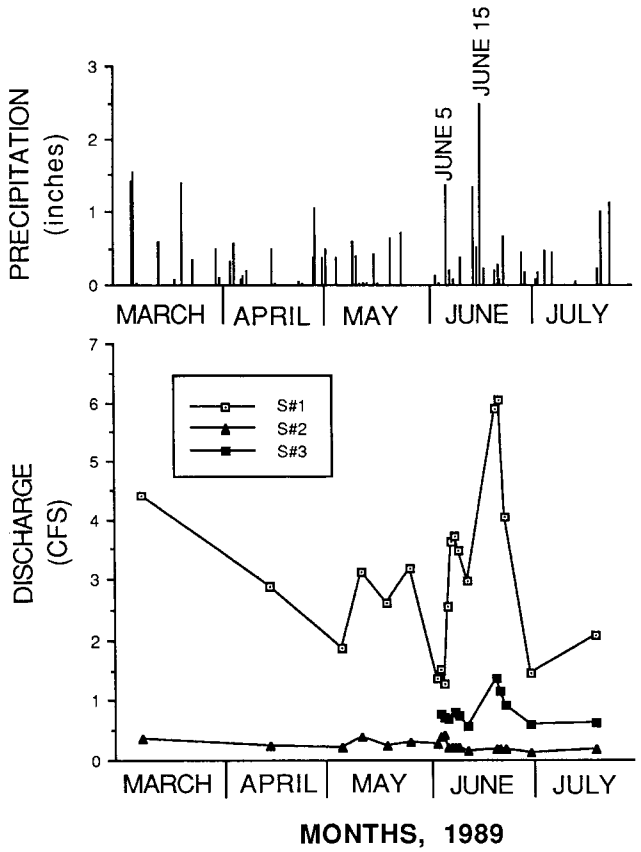



Figure 9. Precipitation and discharge hydrographs for Springs #1, #2, and #3 (March 1-July 25, 1989).

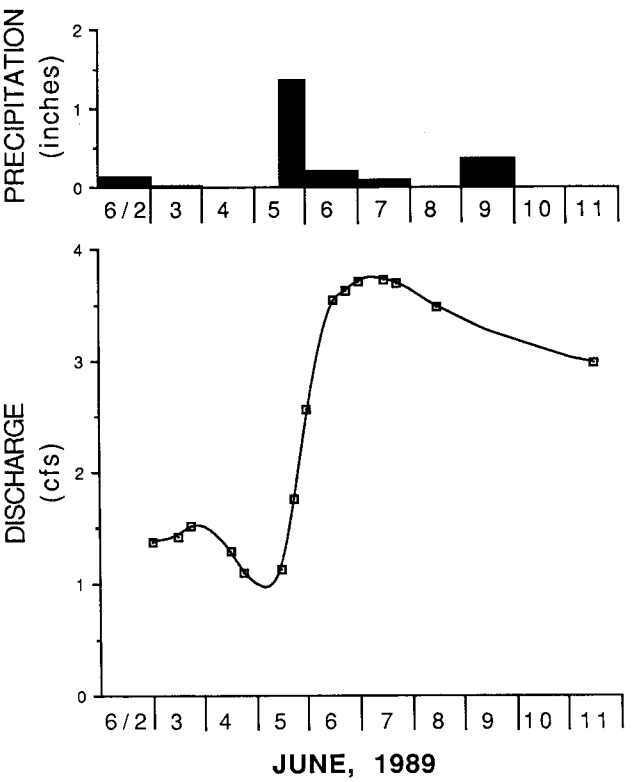



Figure 10. Precipitation and discharge hydrograph for Spring #1 (June 2-11, 1989).

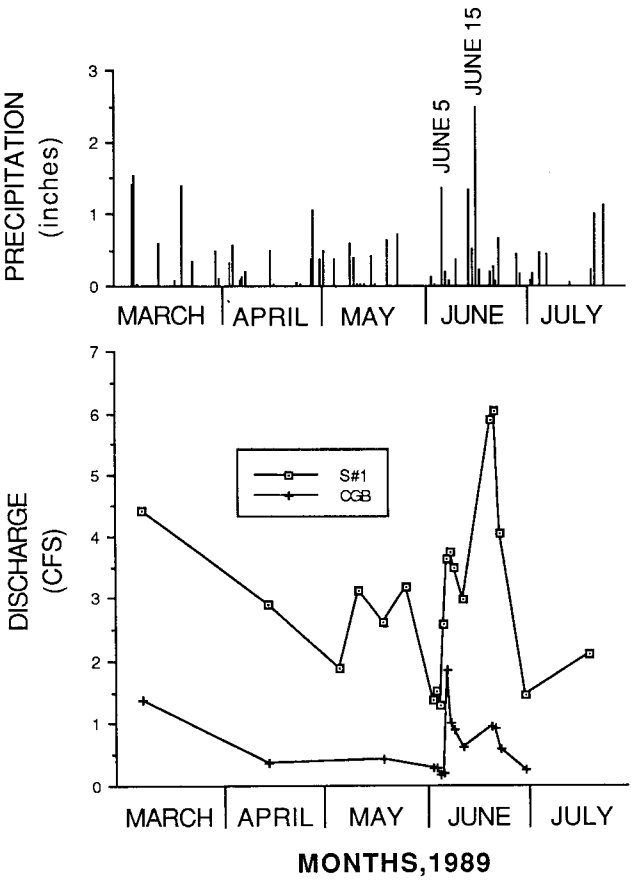



Figure 11. Precipitation and discharge hydrograph for Spring #1 and Chestnut Gap Branch (March 1-July 25, 1989).

portion of the area containing blast-cast and dragline-cast spoil. The water-level elevations in all three wells average nearly 1,130 feet, which is approximately 90 feet above S#1. The thickness of the saturated zone in the spoil is a function of the elevation of the bedrock aquitard below the spoil: the higher the bedrock elevation, the thinner the saturated zone (Fig. 13). The saturated thickness varies from 25 feet at MW#2, to 16 feet at MW#1, to only 8 feet at MW#3, based on measurements recorded on July 18, 1989.

The existence of a water table within this area of the spoil indicates that a significant body of ground water has accumulated. Assuming a constant saturated thickness of 15 feet and an estimated porosity of 20 percent, there would be approximately 3,000 acre-feet (980 million gallons) of water stored within this 1,000 acres of spoil.

Hydrographs for the monitoring wells reveal some important behaviors of the ground-water system (Fig. 14). Response of water level to a 4.55-inch storm event was measured at MW#1 and MW#2 between June 13 and June 17. Water levels in the wells rose approximately 1 foot, peaking within 10 days of the storm's end, whereas the recession to equilibrium levels took an additional 21 days. This behavior is similar to the response measured in the springs and reinforces the concept that the spoil has the ability to store recharged water and release it over an extended period of time.

Monitoring well No. 2 is located on the periphery of the deep infiltration basin (Fig. 12). The water level in MW#2 responds more rapidly to recharge compared to the other two monitoring wells (Fig. 14). One idea is that the deep infiltration basin, although not completely functional, is allowing significant recharge to the spoil, which is raising the water level in MW#2. Based on limited data, the water level in MW#2 averages a consistently higher level (approximately 1.5 feet) compared to the water levels recorded in MW#1 and MW#3.

One explanation for this elevated water level is that a barrier to flow exists west of this well, preventing the movement of ground water toward the spoil outslope and toward the springs previously discussed. Additional supporting evidence for the presence of a ground-water flow barrier is that 90 percent of the spring discharge can be accounted for by drainage from the Spring Gap and Chestnut Gap Branch watersheds, which are located on the opposite side of this barrier.

Field observations indicate that the coal haul road constructed and maintained along the western outslope of the sandstone spoil created by mining between 1981 and 1989, which was continuously compacted by heavy equipment, acts as a low-permeability barrier to ground-water flow (Fig. 13). The effectiveness of this barrier was observed during the active mining of the east-west-oriented pit. Sumps were created against the eastern edge of the haul road at the bottom level of the mine to collect and pump out ground water discharging from the spoil into the pit (Fig. 12).

**Dye Tracing.** Two dye traces have been conducted at the site: trace No. 1 consisted of introducing dye at the Chestnut Gap Branch swallet and trace No. 2 was conducted by introducing the dye into MW#1. In both

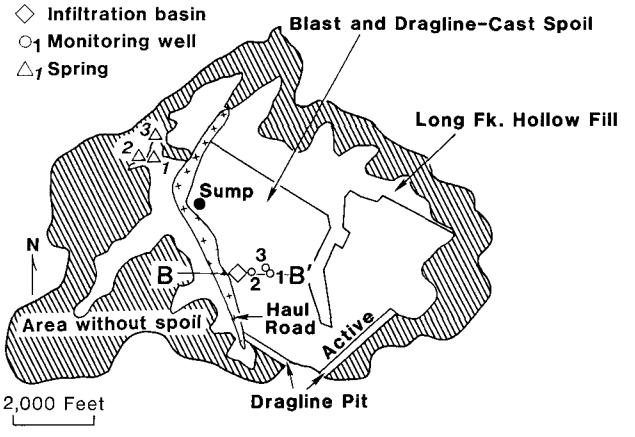



Figure 12. Map showing hydrologic features associated with the monitoring wells.

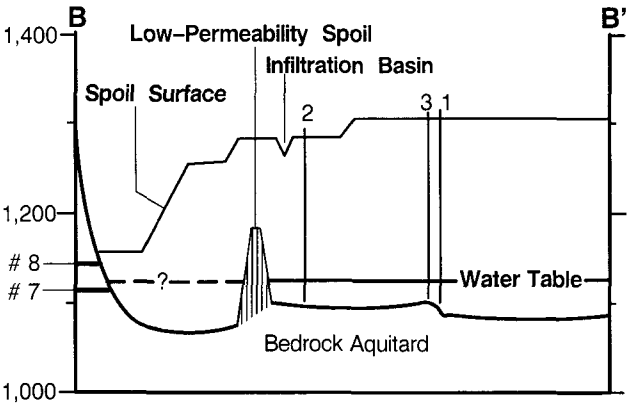



Figure 13. Schematic showing relationship between water table, base of spoil, and thickness of saturated zone. Cross section B-B' located on Figure 12.

traces the springs were monitored for dye discharge. Visual confirmation of dye discharge was overshadowed in the elutriant by a tea-colored interference fluid. Similar coloration has been reported in the literature, and is attributed to the presence of organic matter (Quinlan, 1986). Although visual inspection was unsuccessful, subsequent analyses of the elutriant with a fluorometer produced positive readings. These analyses were reviewed with the knowledge that the intensity could be affected by the interference fluid, which fluoresces at a wavelength close to Rhodamine WT (Smart and Laidlaw 1977). Dye concentrations at S#1 indicate that dye definitely arrived at the springs 49 to 93 hours after injection (Table 1).

Table 1: Trace No. 1 Dye Concentrations

| Date | Hours after Start | Dye Concentration (µg/L) S#1 | S#2 | S#3 |
|---|---|---|---|---|
| 2/22 to 2/27 | -118-0 | 0.00 | 0.00 | 0.00 |
| 2/27 to 3/1 | 0-49 | 0.35 | 0.00 | 0.00 |
| 3/1 to 3/2 | 49-74 | 29.00 | 0.55 | 1.75 |
| 3/2 to 3/3 | 74-93 | 40.00 | 2.80 | 3.20 |
| 3/3 to 3/13 | 93-33 | 2.10 | 0.45 | 1.30 |
| 3/13 to 3/16 | 334-411 | 1.80 | 0.20 | 0.30 |
| 5/11 to 5/18 | 1749-1917 | 0.00 | 0.00 | - |

Breakthrough time for trace No. 1 was used to calculate an apparent velocity ranging from 0.014 to 0.009 feet/second based on a straight-line travel distance of 2,400 feet and a travel time between 49 and 73 hours. This range is close to the flow velocity of 0.002 feet/second reported by Ladwig and Campion (1985), who used an unidentified tracer in spoil at a mountaintop removal operation in Pennsylvania.

Dye injected at MW#1 for trace No. 2 was not recovered at any of the springs. Three months after dye injection, water samples from MW#1 showed high concentration of Rhodamine WT, indicating that ground water in the vicinity of the well flows slowly through saturated, porous media. This result is in contrast to the rapid flow through boulder zones in the hollow fill characteristic of Chestnut Gap Branch, as demonstrated by dye trace No. 1. This slower movement of ground water in this part of the mine spoil and the lack of dye emerging from the springs also suggest that a low-permeability barrier exists between this area and the lower elevation hollow fill in Spring and Chestnut Gap Branches.

## Hydrogeochemistry

The range of values and arithmetic means calculated for a limited set of data for both the surface water and ground water are listed in Table 2. Figure 15 is a tri-linear diagram showing the water types and the relative amounts of dissolved solids found in water samples taken from S#1, MW#1, MW#2, Chestnut Gap Branch (CGB), and surface runoff that was flowing into the deep infiltration basin (IB). All five of

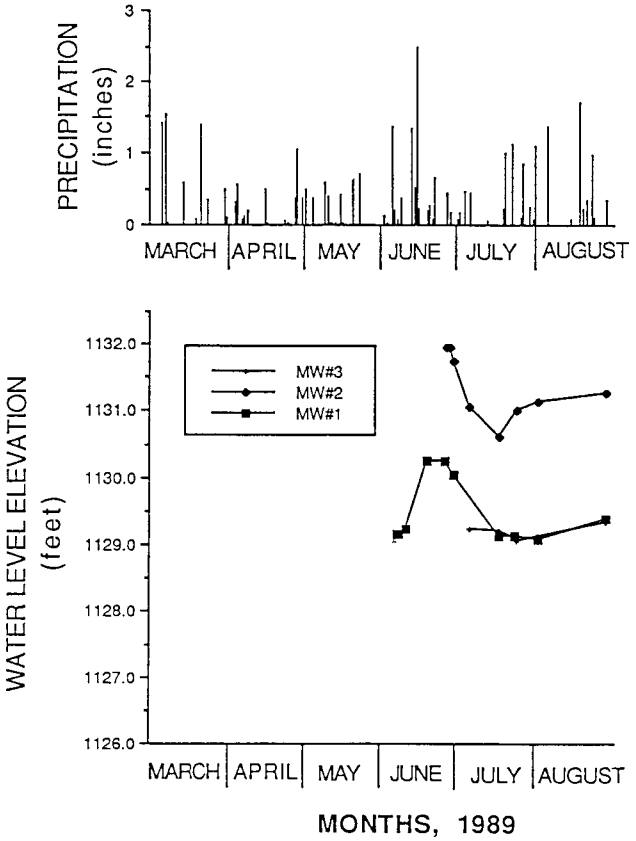



Figure 14. Precipitation and hydrograph for Monitoring Wells #1, #2, and #3 (March 1-August 29, 1989).

the samples plot on Figure 15 as a calcium-magnesium-sulfate type water, the only major difference being in the quantity of total dissolved solids. As used here, the area of the circles plotted on the diagram is proportional to the total dissolved solids of the samples.

The samples with the least amount of dissolved solids are the two surface-water samples, IB and CGB. S#1 shows a dramatic increase in dissolved constituents compared to the surface waters that are recharging the spoil, but considerably less than the samples derived from the monitoring wells at the site. MW#2 consistently has lesser amounts of dissolved solids compared to MW#3. MW#2 is located on the periphery of the infiltration basin; thus, its chemical signature is most likely a result of dilution by surface water as it moves down into the spoil by way of the infiltration basin. This observation is substantiated by the water-level data discussed previously, which indicate that the water level in MW#2 is consistently higher than the level in MW#3 (Fig. 14), and that the water level will exhibit a greater response in MW#2 after a recharge event.

Based on limited data, the total dissolved solids in the water recharging spoil at CGB is significantly lower (as estimated from measurements of specific conductance) than that of the water emerging from the springs at the discharge zone. Discharging ground water has a total dissolved solids content nearly three times the content of the water entering the spoil at Chesnut Gap Branch swallet.

This dramatic increase in mineralization of waters discharging from the springs may result from several scenarios. Two worthy of discussion, are: (1) the recharging water, although only in contact with the spoil material for a very short time (as evidenced by travel times determined by dye-tracing), is reacting with the spoil material, resulting in an increase in dissolved solids; or (2) the relatively fresh water from the stream flowing into and recharging the spoil is mixing with the more mineralized water from stagnant or isolated areas of the spoil (as evidenced by the high conductance values obtained from MW#1 located in the interior section of the spoil), resulting in discharging ground water that is a function of the mixture of the more dilute water entering the spoil at Chestnut Gap Branch swallet and the more mineralized water flowing from within the spoil. Based on preliminary chemical data, dye-trace data, and the fact that the majority of the total discharge measured from the springs can be accounted for by the estimated recharge from the Spring Gap Branch and Chestnut Gap Branch watersheds, it appears that the reaction of water and spoil materials encountered along the ground-water flow path is the most important factor affecting water quality observed at S#1.

Conductivity measurements taken from MW#3 are nearly twice as high as the conductance values of the water discharging from S#1. Increased mineralization is most likely the result of the longer contact time between the ground water and spoil. The monitoring wells are located near the interior of the spoil, somewhat isolated from the majority of fresh water recharge that is thought to be entering near the boundaries of the spoil. Slower ground-water movement is thought to be occurring in the interior section of the spoil, as evidenced by dye-trace data from the monitoring wells. Even if the hydraulic conductivity of the spoil near the monitoring well field is assumed to be the same as that of the Chestnut Gap Branch hollow fill, ground water recharging the spoil at the eastern edge of the spoil (Fig. 12) would have a much longer flow path, resulting in increased contact time and mineralization.

In general, the pH of all the surface and ground water at the site is favorable. The range of pH values for the surface water at the site is 7.62 to 8.28. The pH's of ground water sampled at the site were slightly acidic, but consistently ranged above 6.0 (Table 2). The high quality of the coal, non-acidic overburden, and high neutralization potential of the spoil provide for favorable conditions and minimal occurrence of acid mine drainage. This aspect of the site is integral for the construction and maintenance of an aquifer that may someday provide a usable and beneficial water supply.

Only one sample of water has been taken from the storm water that flowed into the infiltration basin. This sample had a relatively low value of conductance (257 microsiemens) and a pH of 7.67. Future plans include installation of an automatic sampler at the inlet to the infiltration basin to collect water samples for measurement of temporal variation in the runoff quality during recharge events.

## Summary and Conclusions

Major recharge to the spoil appears to be limited to swallets in the border of the spoil, where spoil adjoins higher bedrock outcrops at highwalls and natural slopes, and areas where the spoil has not been graded and compacted. Local recharge can occur where isolated boulders crop out in the graded spoil. Future land use may result in the creation of many structures, including roads and buildings, which could hamper ground-water recharge. Therefore, a deep infiltration basin has been constructed into the spoil. This basin, although not entirely operational at this time, may be responsible for raising the water table approximately 1.5 feet in its vicinity.

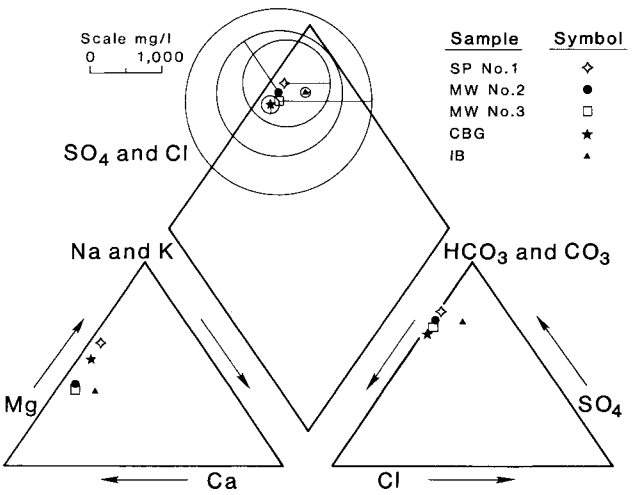



Figure 15. Tri-linear diagram showing the water types and relative amounts of dissolved solids from samples SP#1, MW#1, MW#2, CBG, and IB.

Table 2. Geochemical Data of Samples from Selected Sampling Sites

| | | Sample Location | | | | |
|---|---|---|---|---|---|---|
| | | I.B. (n=1) | CGB (n=3) | S#1 (n=3) | MW#2 (n=3) | MW#3 (n=3) |
| pH | range | 7.67 | 7.96-8.28 | 6.37-6.39 | 6.23-6.41 | 6.42-6.66 |
| | mean | - | - | - | - | - |
| Ca | range | 31.4 | 44.2-54.6 | 201-256 | 291-339 | 490-520 |
| | mean | - | 50.8 | 237 | 312 | 501 |
| Mg | range | 13.6 | 35.8-42.0 | 111-134 | 124-149 | 213-231 |
| | mean | - | 39.9 | 126 | 136 | 222 |
| Na | range | 10.1 | 7.9-13.1 | 32.0-46.0 | 31.3-44.5 | 46.9-54.2 |
| | mean | - | 11.9 | 39.6 | 37.1 | 51.0 |
| K | range | 0.72 | 3.04-4.02 | 7.62-10.5 | 8.7-10.3 | 18.8-20.7 |
| | mean | - | 3.51 | 9.03 | 9.5 | 19.8 |
| Cl | range | <10 | <10 | 5.6-7.0 | 5.3-7.0 | 9.2-11.0 |
| | mean | - | - | 6.2 | 6.1 | 10.4 |
| $SO_4$ | range | 85.0 | 145-240 | 717-894 | 884-994 | 1520-1650 |
| | mean | - | 180 | 827 | 939 | 1600 |
| $HCO_3$ | range | 27 | 110-146 | 286-411 | 492-539 | 695-805 |
| | mean | - | 123 | 338 | 524 | 732 |
| Cond. | range | 257 | 590-720 | 1686-2048 | 1927-2434 | 3061-3635 |
| | mean | - | 610 | 1911 | 2133 | 3310 |
| TDS* | | 180 | 427 | 1337 | 1493 | 2317 |

*TDS estimated by conductance value multiplied by 0.7

Dissolved solids date in mg/L; Cond. = conductivity in microsiemens; I.B. = infiltration basin; CGB = Chestnut Gap Branch, n = number of samples.

Observed ground-water discharge is taking place at the spoil outslope as springs that have emerged only in the past year. Discharge has varied from approximately 1 to 5 million gallons per day, and measurements indicate that the springs react quickly to precipitation events. The rapid rise in discharge within a few hours of the beginning of rainfall indicates that coarse rubble zones are behaving as conduits, whereas the extended recession curve of spring hydrographs indicates that the finer sized spoil material is releasing ground water over a period of days. Recharge:area runoff estimates and dye tracing indicate that the springs are being fed primarily through the older hollow fills in Spring Gap and Chestnut Gap Branches. The extensive spoil created by the present mining, which began in 1981, does not appear to be contributing significant ground water to the springs.

Data from three monitoring wells and water levels measured in the active pit indicate that this more recent spoil (approximately 200 feet thick) has a water table approximately 90 feet above the springs and a saturated thickness ranging from 8 to 25 feet, depending on the structural contour of the underburden of the lowest coal bed that is being mined. The large difference in elevation between the water table and the springs, dye tracing and chemical data from a swallet and monitoring wells, and the discharge:area runoff ratios indicate that the extensively used (heavily compacted) haul road along the western edge of the present mine site retards the flow of ground water from the active mine site to the springs. Changes in spoil compaction or overburden type affecting ground-water movement is also observed in the hollow fills from which the springs emerge. In this location shaly spoil placed into the hollow has forced the ground water to the land surface. The effects of compaction and spoil lithology on ground-water movement is an important observation when developing the concept of building aquifers in spoil to meet water-resources needs. There is also a need to coordinate mining with reclamation plans in order to develop the most advantageous water supplies for post-mining development.

Ground-water quality for both the springs emerging from the hollow fill and the monitoring wells in the upper spoil can be classified as a calcium-magnesium-sulfate type. Spring water has a specific electrical conductance mean of 1,911 microsiemens, whereas that of the monitoring well away from the infiltration basin averages over 3,300 microsiemens. This higher value reflects the longer travel time and concomitant mineralization of ground water flowing through the areally extensive upper spoil. The monitoring well near the infiltration basin has a reduced conductance value of 2,133 microsiemens when compared to the other two wells, which indicates that better quality surface water is infiltrating to the ground water and improving its quality. The pH of all springs and wells fell into a favorable range of 6 to 7, whereas sulfate was high, ranging from 717 to 1,652 mg/L.

Based on the initial water-quality and -quantity data measured at the Star Fire Mine, it is apparent that the mining techniques can provide the physical framework to construct an aquifer in the extensive mine spoil. Development of a useful water supply within the spoil will be a key factor in future land use and economic diversity of the site.

## Literature Cited


Aldous, P. J., and P. L. Smart. 1987. Tracing ground-water movement in abandoned coal-mined aquifers using fluorescent dyes. Ground Water 26 (2): 172-178.

Aulenbach, D. B., J. H. Bull, and B. C. Middlesworth. 1978. Use of tracers to confirm ground-water flow. Ground Water 16 (3): 149-157.

Caruccio, F. T., and G. Geidel. 1984. Induced alkaline recharge zones to mitigate acidic seeps. In Proceedings, 1984 Symposium on Surface Mining, Hydrology, Sedimentology, and Reclamation, University of Kentucky. p. 43-47.

Danilchik, W., and H. Waldrop. 1978. Geologic map of the Vest Quadrangle. U.S. Geological Survey, Geologic Quadrangle Map GQ-1476.

Dinger, J. S., R. C. Warner, and J. E. Kemp. 1988. Building an aquifer in coal-mine spoil: Concept and initial construction at the Star Fire tract, eastern Kentucky. In Proceedings, 1988 Symposium on Surface Mining, Hydrology, Sedimentology, and Reclamation, University of Kentucky. p. 85-93.

Dinger, J., R. Warner, P. Schwab, and J. Kemp. 1987. Water resources of the Star Fire tract, a first look. In The Star Fire Project, a report on work started during 1987 and directions for research and development 1988, 89. Kentucky Geological Survey, University of Kentucky, Lexington, Kentucky. p. 42-66.

Hinrichs, E. N.. 1978. Geologic map of the Noble Quadrangle. U.S. Geological Survey, Geologic Quadrangle Map GQ-1411.

Ladwig, K. J., and P. Campion. 1985. Spoil water quality variation at two regraded surface mines in Pennsylvania. In Proceedings, 1985 Symposium on Surface Mining, Hydrology, Sedimentology, and Reclamation. University of Kentucky. p. 121-130.

Quinlan, J.F., ed. 1986. Water-tracing techniques in karst terranes. In Course manual: Practical karst hydrogeology, with emphasis on ground-water monitoring. National Water Well Association, 898 p.

Schwab, P. 1987. SEDCAD—Sediment, Erosion, Discharge by Computer-Aided Design. M.S. Thesis, Agricultural Engineering Department, University of Kentucky, Lexington, Kentucky.

Schwab, P., and R. Warner. 1987. SEDCAD+ user's manual. Civil Software Design, Lexington, Kentucky.

Smart, P. L., and I. M. S. Laidlaw. 1977. An evaluation of some fluorescent dyes for water tracing. Water Resources Research 13: 15-33.

This page intentionally left blank.

# INNOVATIVE DESIGN FOR IMPROVED MINING ECONOMICS AND LAND USE[1]

by

Timothy A. Keeney and George A. Hall[2]

**Abstract.** There are two primary purposes for good engineering design: (1) public safety and welfare; and (2) economy of construction. This paper describes a specific application of an innovative design to achieve these purposes in the surface mining of coal. Fundamental mine design considerations include site specific topography, as it affects excess spoil disposal and drainage control, and the physical and chemical characteristics of the overburden as they pertain to earth moving, spoil disposal, and postmining land use. A unique application of the design purposes to these design considerations is described for a surface mining operation in central West Virginia where site conditions did not fit conventional surface mining methods.

Additional Key Words: Engineering, Regulations, Surface Mine Reclamation

## Introduction

There are conspicuous variations in geologic conditions from one geographic area to another, and these variations are very important to mining operations. For example, eastern Kentucky, southern West Virginia, and western Virginia are generally characterized by sandstone dominant strata forming narrow valleys and very steep hillsides. On the other hand, eastern Ohio, southwestern Pennsylvania, and northern West Virginia are frequently characterized by shale dominant strata, often easily weathered to clay, which have formed moderate slopes and wider valleys. These examples are broad generalizations, and there are diverse local variations. Central West Virginia is replete with such variations.

The topography, the chemical and engineering properties of the overburden, and other site specific conditions, along with the intended postmining land use, actually delimit the mine design. Of course, inherent in all design must be consideration of public safety and welfare. Any feasible design which fits these requisite conditions should be evaluated, and the most suitable (usually from an economic viewpoint) should be selected. Unfortunately, this is seldom the approach used in the design of surface mining, especially with respect to spoil disposal and reclamation.

For the most part, spoil disposal, drainage and sediment control, and reclamation are dictated by governmental regulations, not by site specific conditions and sound engineering design. The broad spectrum of geologic conditions must be met with a few approved "proven" designs--anything else is, by definition, "experimental."

This paper describes an instance where the geologic conditions and the intended postmining land use did not lend themselves to application of conventional spoil disposal and reclamation methods. Fortunately, the West Virginia Department of Energy accepted an alternative design under a "variance" provision in their regulations. The alternative engineering design which resulted in more fav-

[1]Paper presented at the 1990 Mining and Reclamation Conference, Charleston, West Virginia, April 23-26, 1990. Publication here does not preclude publishing this manuscript, in whole or in part, elsewhere.

[2]Timothy A. Keeney is President of Level Land Construction Company, Inc., 1217 Virginia Street East, Charleston, WV 25301; George A. Hall is President of Terra Engineers, Inc., 1624-1/2 Kanawha Boulevard East, Charleston, WV 25311.

orable mining economics and improved postmining land use potential is described.

## Site Description

### Fundamental Mine Design

The design of a surface mining operation must utilize earthmoving procedures which will avoid degradation of surface and ground water by either acid drainage or suspended sediment. This begins with an assessment of the physical and chemical properties of the overburden to be disturbed, followed by development of a suitable approach to disposal of excess spoil which recognizes constraints of site specific topography. Finally, cost-effective measures to provide drainage and sediment control for all disturbed areas must be provided.

## Site Topography and Geology

### Description of the Problems

The problems associated with mining of coal reserves in this area were varied. First, the surface land owners were insistent that the postmining landscape be capable of greater utility than before mining, since premining slopes precluded uses more intensive than forestland. Second, as can be seen from the geologic description above, quality durable rock was scarce to nonexistent during the initial mining cuts. Third, the expensive hauldown and special handling required with conventional excess disposal in valley fills would have to be avoided to maintain mining feasibility. Additionally, surface landowners were adverse to the construction of open rock cores across the resulting fill areas.

The operation in question is located in Nicholas County, West Virginia, approximately 10 miles north of the city of Summersville. A series of surface mining operations had previously been conducted along the hillsides of an elongated valley in the headwaters of Little Beaver Creek of Gauley River. The hillsides were traversed by periodic natural drainage channels. The drainage channels were nearly perpendicular to the stream in the valley, although some channels had branches in their upper extremities. Slopes were moderate, but somewhat steeper in the higher elevations. As shown in Figure 1, natural drainage channels were quite narrow, with side slopes averaging 50%.

The overburden to be disturbed included predominantly fine-textured strata overlying four coal seams. In descending order, the seams included the Cedar Grove Rider, Cedar Grove, Alma, and Peerless. Shale and mudstone overlie the Cedar Grove Rider, and the Cedar Grove Rider and Cedar Grove are separated by an interval of 45 ft of shale and mudstone. The Cedar Grove to Alma interval was made up of 20 ft of mudstone, while the Alma to Peerless interval was comprised of a 10 to 12 ft interval of fine-textured sandstone underlain by 18 ft of mudstone.

## Materials and Methods

### Characterization of Overburden Materials

The geologic materials to be disturbed were analyzed by Acid-Base Accounting in accordance with EPA Manual No. 600/2-78-054. The strata were shown to contain an overwhelming dominance of alkaline materials, ranging from 10 to 40 tons/1000 tons $CaCO_3$ Equivalent. In addition, the materials were analyzed for available plant nutrients and found to contain high to very high levels of plant available Ca, Mg, and K. Phosporous levels were low, but were superior to native soil levels. Using these data and supported by successful applications on surrounding operations, a topsoiling variance was obtained. This eliminated costly rehandle, lowering earthmoving costs.

## Results and Discussion

### Excess Spoil Disposal-Design to Fit Conditions

Excess spoil disposal is generally accomplished in one of three ways: 1) Downslope placement, when slopes are less than 20 degrees (36 percent); 2) Valley Fills, in which a rock core extends along the axis of the valley all the way from the valley floor to the surface of the fill; or 3) Durable Rock Fills, in which at least 80% of the spoil must be durable rock.

As previously described, the overburden materials at the site in question were dominated by fine-textured materials. Excess spoil disposal was further complicated by the fact that initial cuts resulted in the generation of spoil materials dominated by soil and weathered rock. Durable rock core materials would not be released until mining of the 2nd and 3rd cuts. Additionally, the surface owners requested that the rock core not be left open across the top of the fills so as not to interfere with the utility of the reclaimed land.

Under these constraints, the approach was developed to design the valley fills with rock underdrains which would pass the peak discharge from a 100 year, 24 hour storm event. This would accomplish the necessary drainage and fill stability, as well as compensate for the scarcity of durable rock in the development phase of the operation by decreasing the minimum required rock volume.

As a further means of eliminating unnecessary mining costs as well as to compensate for the dominant volume of nondurable material, a waiver was obtained to allow placement of fill material in 50 ft lifts rather than the required 4 ft lifts. This procedure eliminated the costly proposition of hauling material down to the lower lifts by dumping them from a higher elevation, which also allowed for the required durable core rock to segregate by gravity at the base of the dumped slope.

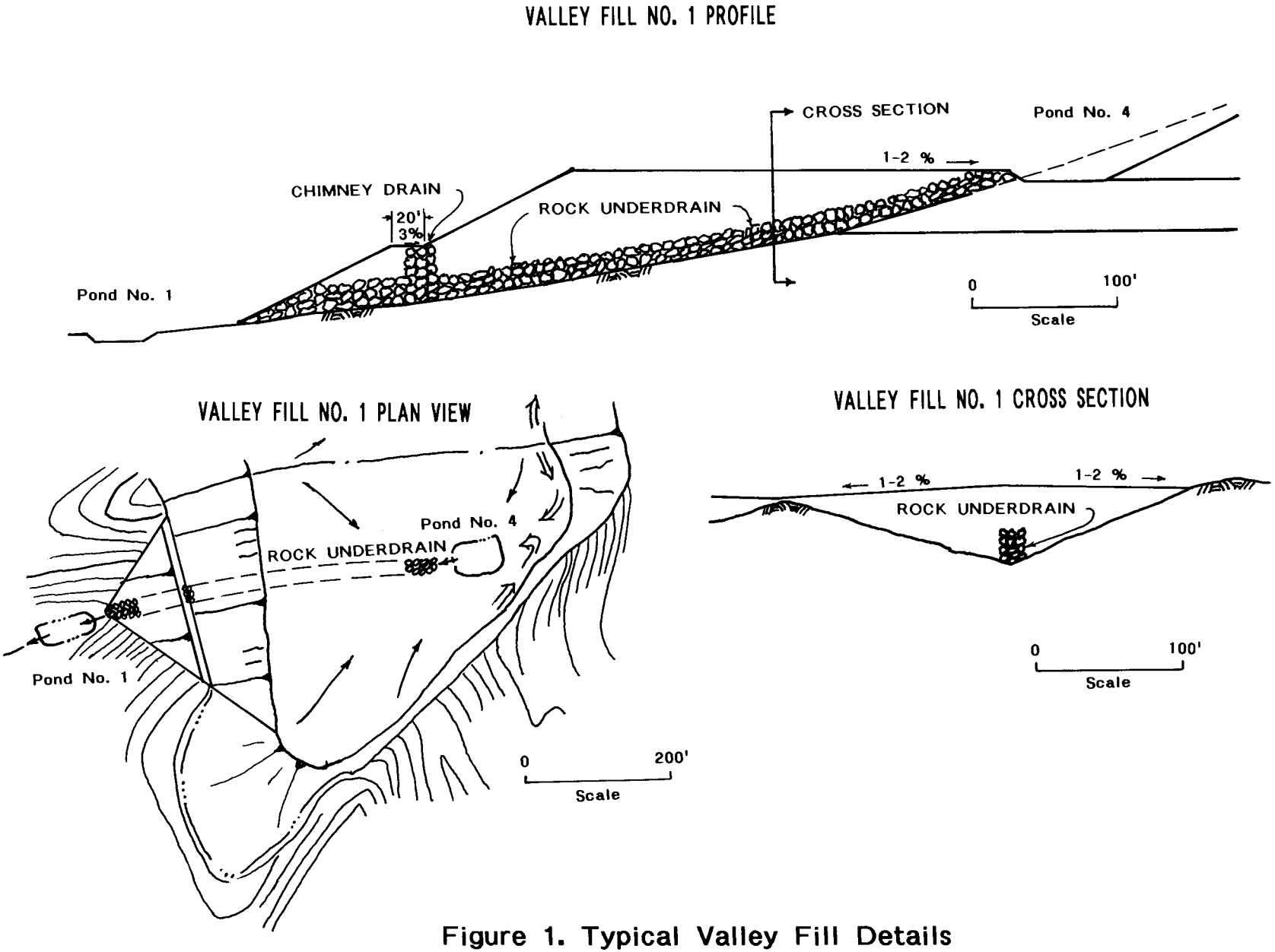



Figure 1. Typical Valley Fill Details

Some further details of the drainage design and construction and their effect on slope stability require delineation. It is these details which allow the design to satisfy the design purposes. Firstly, the rock drain cross section sized for a 100 year, 24 hour flood was the minimum allowable cross section. The fills were constructed by dumping predominantly nondurable material along the sides of the hollows, leaving a channel in the middle for the underdrain. Boulders of durable rock tended to roll to the bottom. Many of these were intentionally separated from the fill and pushed over the edge of the fill to roll into the underdrain area; a dozer with a rake attachment was used for that purpose. Of course, the sandstone stratum was the major source of rock for the underdrain. Some of the shale encountered was relatively durable, and boulders of this material were used to cover the drain. The net effect of this careful construction was underdrains which were larger than the required minimum at virtually all locations.

Secondly, specifications required an enlargement of the drain cross section under the downstream face of the embankment for two reasons: bench drainage was passed to the rock drain by means of chimney drains (see Figure 1); an enlarged drain in the toe area prevents the possibility of pressure flow during an extreme flood, thereby preventing an adverse effect on slope stability.

## Postmining Land Use

The requests of the surface owners to construct a postmining landscape that was conducive to a higher level of land management was a major consideration with the design of the operation. The concept of the rock underdrains and valley fills, coupled with the documentation of the presence of highly favorable chemical and physical properties of the overburden materials, allowed for the construction of a wide terrace-like bench area along the contour. Slopes were graded to drain to the back of the fills into sediment ponds. The ponds are situated behind the surface opening of the rock underdrains so that when filled with storm drainage, the ponds will discharge into the underdrains. This procedure would appear to offer an added benefit of water conservation by augmenting both normal and storm drainage volumes.

The mixed overburden material was used as the topsoiling medium for plant growth on the entire operation due to the alkaline nature and fine-textured characteristics of the material. This procedure has been used on adjacent areas since 1982 with highly favorable results. Approximately 30 acres of alfalfa and timothy hay crops are currently in production.

The terrace or "Level Land" design approach has been to construct landscape capable of supporting a variety of uses. As reported above, the main use to date has been hayland. The manageable topography can be conceived to be also favorable for livestock enterprises. However, one residence has now been constructed by a surface owner on land mined and reclaimed in this manner. It is apparent that uses such as forestland would be enhanced due to the ability to use more intensive management techniques as well as mechanical harvesting on the gentle slopes.

## Conclusions

The project described in this paper illustrates one instance where site specific conditions and postmining land use required some innovative design. The innovative design accomplished the intended goals of both public safety and economy of design. Careful construction practices improved on the design to increase the safety of the fills.

It is of paramount importance that government regulations be sufficiently flexible to allow alternative design practices. Many designs are possible which rely on sound principles, are not "experimental," and are equal or superior to "accepted" practice. If the coal industry is to survive, these designs must be encouraged.

It is of equal importance that the industry use care in constructing innovative projects. The construction described herein required a little extra time and effort, but the results will benefit posterity.

# THE LAND USE PLANNING PROCESS AS APPLIED TO A MOUNTAINTOP REMOVAL COAL MINE SITE IN EASTERN KY[1]

by

Thomas J. Nieman and Diane S. Meshako[2]

**Abstract** -- Pro-active planning is a classic case of "form following function" as it encompasses the spectrum of land uses and land forms. The planning model being developed for mountaintop removal coal mining sites is designed to transform inaccessible, unusable, mined land into land suitable for industrial, residential, recreational and a multitude of other uses. The mine site in question is unique in that the land area is contiguous, it is owned fee-simple by one company, and all of the coal is removed. Unlike many surface mines, the site will never be remined. Thus any reclamation and subsequent development can be considered permanent. It is the intent of this project to provide a model for economic and social development through planned utilization of "developable" land resulting from mining and reclamation.

## Introduction

Comprehensive land use planning, when approached from the physical resource perspective, typically responds to existing physiographic characteristics. Factors such as slope, aspect, soil, and vegetation influence many existing land use decisions and present conditions. However, in the case of mountaintop removal mining sites, all existing physiographic characteristics are obliterated, thus leaving the planner with no existing topographical factors on which to apply developmental considerations. Land forms and site conditions can be developed as a result of land use decisions made prior to the mining and reclamation process.

One concern is the "carrying capacity" of the possible land forms, including topped mountains and filled valleys. Developmental experiences on mined lands have led to the conclusion that "all made land consisting of valley fill in surface mine areas of the Southern Appalachians should be considered as potentially unstable ground, subject to subsidence, even where normal engineering precautions for good stabilization have been taken in the placement of the fill" (Krebs 1987).

Subsidence or creep and collapse settlements are current topics in geotechnical research. Creep settlement refers to the routine settlement of loose earth materials over time as the materials consolidate under forces imposed by their own weight. The rate of creep settlement varies directly with the fill depth, decreases exponentially with time, and varies greatly with the degree of compaction during placement. In contrast, collapse settlement, which is dependent upon water and expected in most fills over 20 feet, may be both substantial and rapid. This hydroconsolidation is accelerated as the water softens the point-to-point rock contacts (Krebs and Zipper 1989). Additionally, "angular distortion from vertical movement should be expected closer to the side than to the centerline" of valley fills (Krebs 1987). These shifts in the earth are often exacerbated by water percolation from leaky pipes or runoff from paved areas.

Additional physical concerns include problems associated and compounded by standard developments. They include sewage disposal, an adequate water supply, and access. However, planning and innovative techniques can contribute to a successful development. Constructed wetlands sewage systems (Goldstein 1989) have been effective as well as economical to build and maintain in rural settings. Similarly, a computer program, IWR-MAIN, a water use forecasting system (U.S. Army Corps of Engineers 1988), may be used to realistically assess the needs and estimate future water demand of an area. Community sectors are disaggregated into individual categories such as metered and sewered residences, commercial establishments, and three-digit SIC manufacturing categories. Using this tool, a planner can predict long-term impacts of water demand management or water conservation practices. Regional access has been greatly improved due to the efforts of the Appalachian Regional Commission.

However, to be successful, planning also needs to be approached from the sociocultural perspective. Historically, in eastern Kentucky, the local economy

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]Thomas J.Nieman is a Professor and Diane S. Meshako is a research specialist, Department of Horticulture and Landscape Architecture, University of Kentucky, Lexington, KY 40546.

has been dependent upon fluctuations in the international energy markets; therefore, the coal industry has not generated the kind of income stability that would provide for public and individual investments in the future. Local politics, land ownership patterns, an inadequate tax base and until recently, poor access have limited public efforts to search for alternative economic development. When confronted with low and uncertain incomes, individuals and families have had little incentive or means to invest in their own futures through educational and other improvements. With all of these various needs and concerns identified, the integration of the issues will permit a planning model and suitable development to take place.

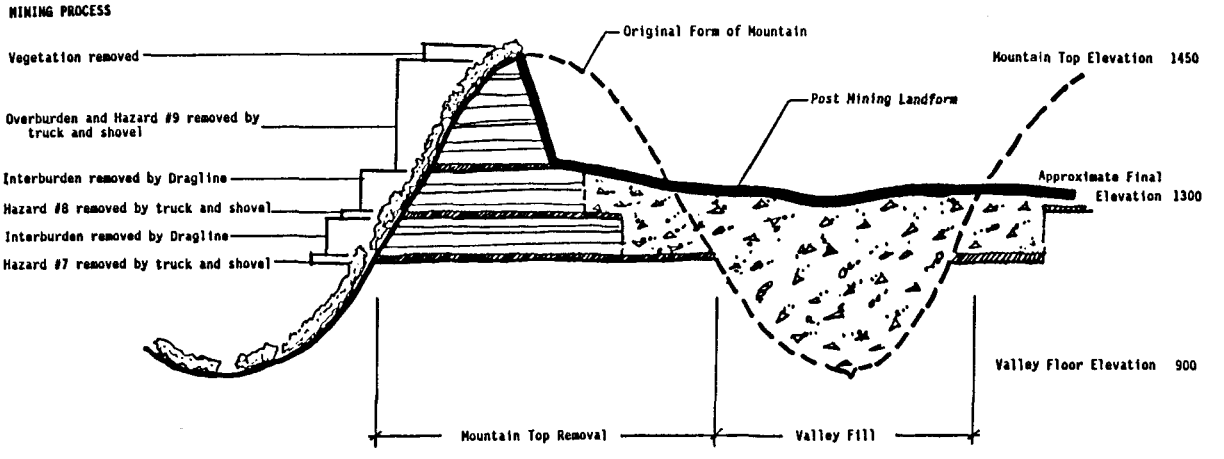
## Site Description

The Cyprus Minerals Corporation is the sole owner of the mineral and surface rights of Star Fire, a 17,000 acre site in eastern Kentucky. In response to their concern for an ecologically sound post-mined land use, they have supported efforts to develop a comprehensive plan to demonstrate a commitment to stewardship of the land. The mountaintop removal/valley fill mining process (Figure 1) includes the excavation of the entire top of a coal-bearing mountain while the coal is removed seam by seam. At Star Fire all of the overburden above the Hazard #9 seam is removed by the standard truck and shovel technique, with the overburden being placed in an adjacent valley. After the 9th seam of coal is removed, the interburden between the Hazard #9 and #8 seams is "moved over" by "cast blasting" and a 65 yard capacity Marion drag line. Similarly, the interburden between the Hazard #8 and #7 seam is "moved over;" however, it remains on the mined out area of the mountain. The result is a newly created landscape devoid of vegetation. The mountain per se is dramatically changed, all of the coal has been removed, and the site will never be remined. What were once valleys may now be the highest areas of the site.

At the present rate of mining, approximately 4 million tons per year, by the year 2010 over 5000 acres of reformed "land" will have been created. Currently the mining process and the reclamation program at Star Fire is focused on the creation of flat land that meets the reclamation standards. A Canada goose lake and wildlife refuge, which was a short-term goal, has been realized. While the normal reclamation process with regard to grading side slopes, smoothing out spoil cones and hydroseeding the area was taking place, a 35 acre lake was created in 1988 and 1989 on 250 feet of unconsolidated spoil. This newly created ecological niche on a "mountain-topped" site is being monitored to aid in the understanding of the dynamics and problems associated with building on massive fill areas.

## Methods

While short-term goals such as the Canada goose lake are being achieved, a planning process is evolving that will define long-term objectives and establish the models needed to accomplish these objectives. Investigation is centering on the intersection of the realm of the mining process and the resulting potential land forms, the economic feasibility of future development including the criteria critical for land uses, and the social acceptability of future plans (Figure 2). The model and the intersection of the elements will continue to change as more information is integrated into the system. The process is dynamic, and since there is little historical reference to use as a basis, much of the investigation evolves around the dynamics of the process rather than pre-established planning and development criteria.




MOUNTAIN TOP REMOVAL / VALLEY FILL MINING PROCESS

Figure 1

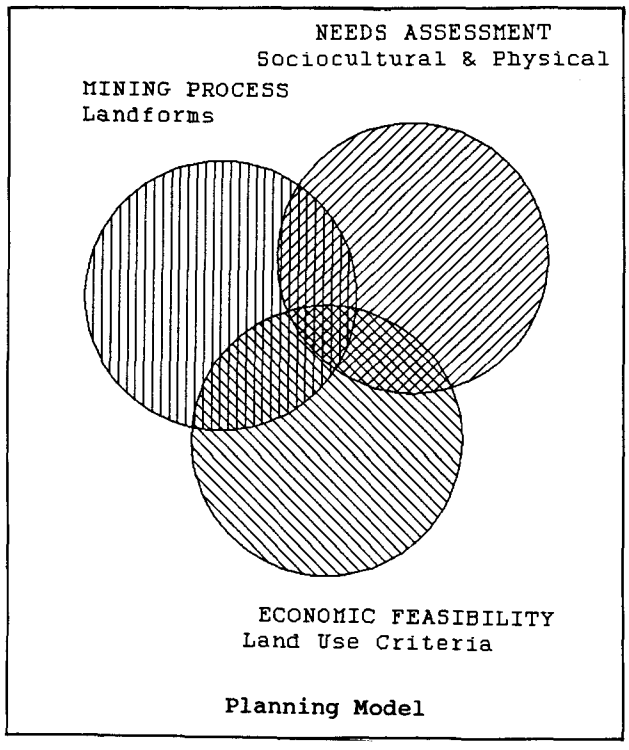



Figure 2

## Land Forms

Three proposed land forms and their relationship to the original land form are illustrated (Figure 3). In each sketch, the fill on the original mountain is very gently rolling or flat; however, in (A) the fill on the valley is at an elevation that is less than the elevation of the fill on the mountain and it can be shaped to capture maximum solar radiation (Olgyay 1973); in (B) the fill is gently rolling at the same elevation as the fill on the mountain; and in (C) the fill is significantly higher than the fill on the mountain, which again can be shaped to capture and maximize solar radiation. The ridge lines should be parallel to the original valley; however, they can be moved to create long gentle slopes with the desired orientation. Land forms as well as vegetation can be used to channel the strong north west winter winds and buffer pedestrian activities. These hypothetical land forms are presented to emphasize the choices that are available when making final land form determinations.

## Sociocultural

The existing social factors of the region, such as eastern Kentucky's historical dependence on the coal industry for the majority of its income, its land ownership patterns, and its low educational attainments and high unemployment levels, comprise a third area of this study. The scarcity of developable land in eastern Kentucky is related as much to these sociocultural factors as it is to the physiographic conditions of the region.

Existing law (federal - Public Law 95-87 1977 and state - 405 KAR 1978) and regional land use plans (KRADD 1978) were reviewed to ascertain which reclamation efforts would be consistent with established social perceptions. In addition, existing land forms and land use patterns were analyzed in order to understand their cultural implications for future development. The intent of a new comprehensive master plan is to satisfy the long term goals, to enhance what already exists and to integrate new ideas.

## Economic Feasibility

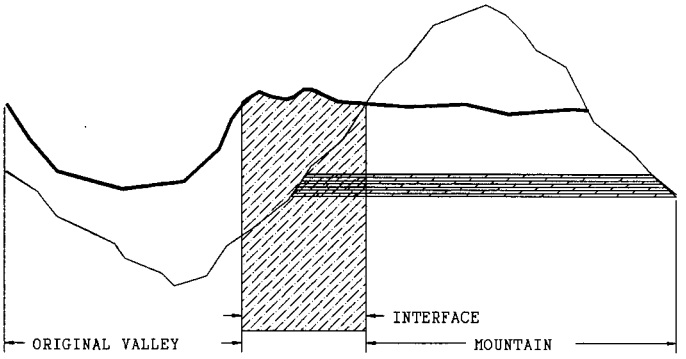
Heavy industry, light industry, public services, residential, agriculture, impoundments of water and wildlife, alone or in combination, are the land uses categorized by 405 KAR. Their potential for development on Star Fire is the second realm examined from the perspective of use intensity and carrying capacity in order to establish influencing criteria (Nieman et al. 1989). These criteria will then be used to develop scenarios depicting the range of each land use intensity and to serve as an aid to communicate potential design concepts and various alternatives. The number of physical site related factors which appear to be influencing development decisions are surprisingly few but very complex. They include macroclimate (wind, solar orientation and rainfall), drainage, soils, slope, and infrastructure such as access and utilities. Once influencing physical factors are established, acceptable parameters can be proposed that will determine which criteria can be satisfied within the realm of economic feasibility.

## Results and Discussion

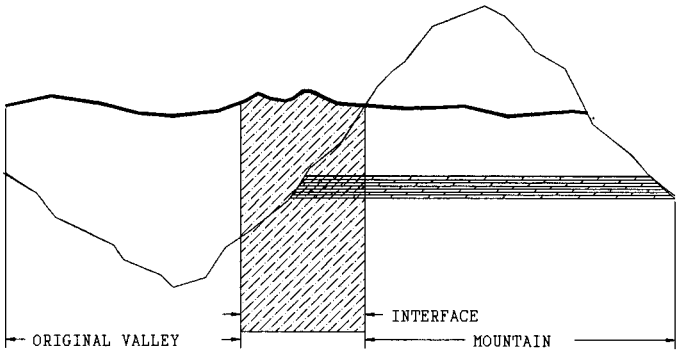
Using the integration of the mining process, the economic feasibility and the sociocultural studies, it was decided to examine the original topography of the Star Fire site: the 1200 foot contour was traced, since this is the approximate elevation of the seventh seam (where it exists) (Figure 4). The line delineating the opaque (original mountain) and the light (valley fill) areas is the interface zone. This is the area in which there is the greatest likelihood for maximum differential movement and maximum angular distortion. For this reason, the area along the interface is not an optimum building site and should be left as a buffer between land forms and/or land uses. It could be a transition zone accommodating recreational land uses, such as golf courses or wildlife refuges. These areas or "cone zones" would require minimal grading efforts and would act as a delineator of land forms in the landscape.

Each of the three areas of the model is being investigated independently. The results are being integrated, and potential scenarios are being developed and illustrated. The first scenario is a residential neighborhood conjoined with industry (Figure 5). The residences are situated on a slope which has a southeast orientation and is supported by all valley fill. The industry is sited on the spoils overlaid on the original mountain, with the interface zone left as a buffer. This buffer could be planted for aesthetics and as a wind break for

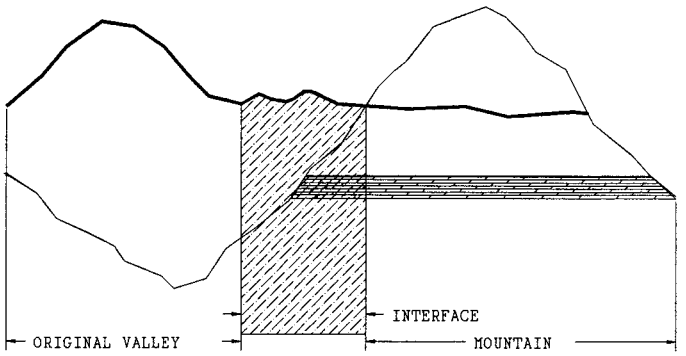
A.




B.




C.




HAZARD #7 -- 1200 FT

PROPOSED FINAL GRADE -- 1300 FT

ORIGINAL CONTOUR -- 900 TO 1400 FT

**Figure 3**

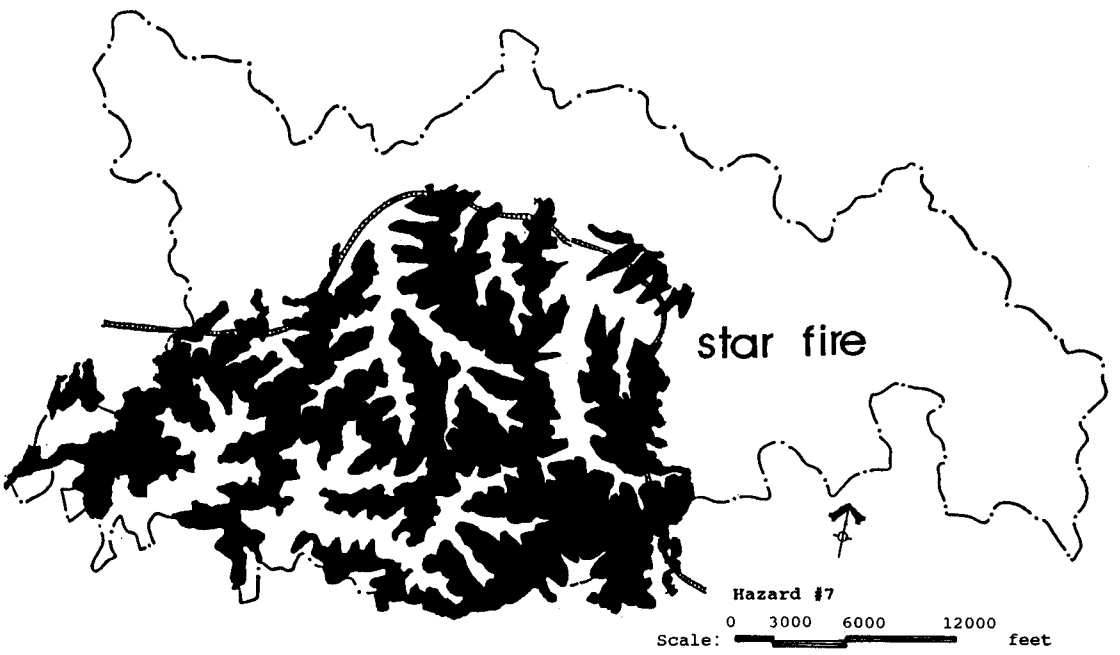



Figure 4

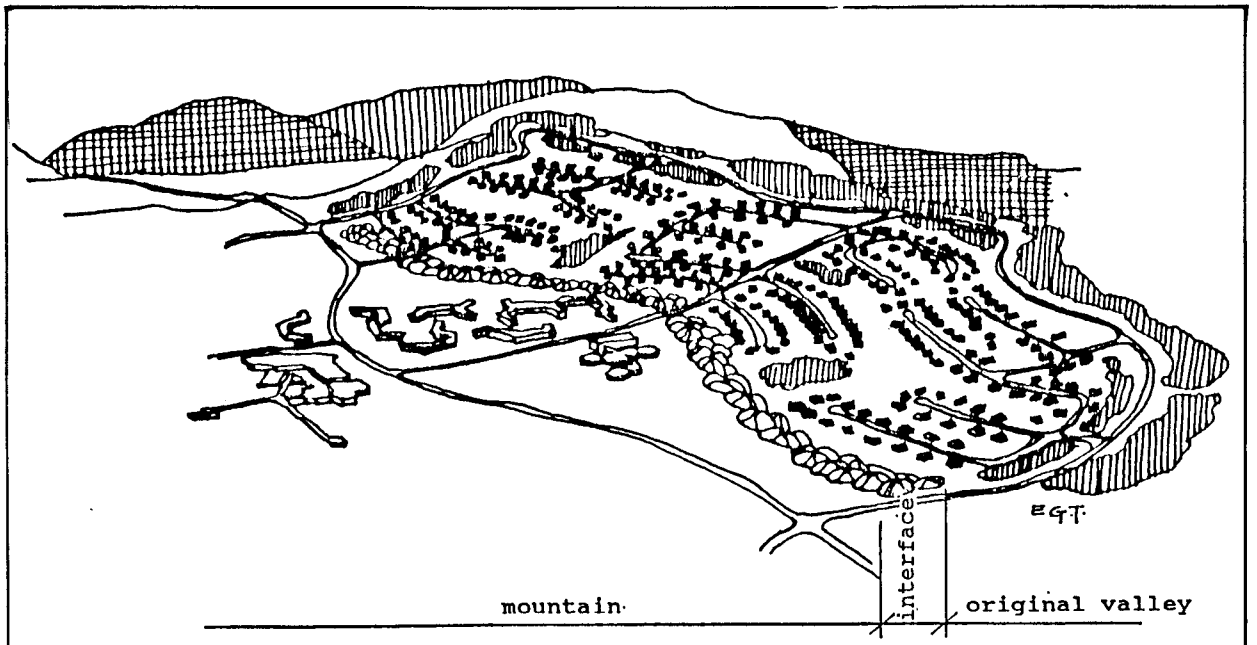



Figure 5: Industry conjoined with residential landuse. The industry is sited on the original mountain and the residences on valley fill. Each benefits from the use of the interface zone as a buffer.

industry. In this scenario the valley has been filled to an elevation higher than the fill on the original mountain in order to capitalize on the macroclimate. In addition, the residences are also oriented parallel to the valley centerline, which should minimize foundation adjustments. Both the solar orientation of the structures and the buffered northwest winter wind will provide efficient and comfortable dwellings and employment sites. Other scenarios including agriculture, public services, recreation, and wildlife development are being investigated and considered in various combinations and intensities.

## Conclusion

Star Fire has proven to be a very large, dynamic, and challenging laboratory that integrates the broad realm of physical, social, and economic decision making. Its uniqueness relative to location, process and support establishes Star Fire as a potential demonstrative model that can impact the changing pattern of development in eastern Kentucky far into the future. This is a unique situation, where past planning experience provides little help in making land form determinations or development decisions. All of the existing physiographic characteristics have been altered as a result of the mining process. Thus physiographic elements are being newly created rather than acting as influencing determinants. The culture of the region, which traditionally has been an enigma, may become the most critical aspect of the planning process. The land form that is being constructed is deceiving -- it neither reflects the cultural character of the area nor does it reveal the underlying substructure of the previous mountain/valley configurations. The plains evoke a false sense of stability, which conveys a mixed message. To an observer who is familiar with the shortage of developable land in the area, Star Fire must look like a gold mine. To a professional who is familiar with the previous land configuration it presents a challenge. If people truly prefer to live in an environment that is similar to the one in which they were reared, it is incumbent upon the planning process to sift out and provide the physical elements that are determined to be salient in providing an environment that will be acceptable to the people in question. Time and commitment on the part of the coal company, the various levels of government, and those involved in the planning process will make the difference.

## Literature Cited


Congressional Research Service, Library of Congress. 1977. Public Law 95-87.

Commonwealth of Kentucky Department for Natural Resources and Environmental Protection, Bureau of Surface Mining Reclamation and Enforcement. 1978. Regulations for the Strip Mining of Coal and Surface Effects of Underground Coal Mining and Surface Effects of Non-Coal Mining, (405 KAR).

Goldstein, Bruce. 1989. Sewage Treatment -- Naturally. Renewable Resource Journal, 7(2).

Kentucky River Area Development District (KRADD). 1978. Land Use Plan.

Krebs, Robert D.. 1987. Residential Foundations on Reclaimed Land. In Powell River Project Symposium and Progress Reports, Virginia Polytechnic Institute, Clinch Valley College, Wise, VA.

Krebs, Robert D. and Carl Zipper. 1989. Foundations for Housing on Reclaimed Mined Lands, Virginia Cooperative Extension Service, Pub. 460-115.

Nieman, Thomas J., Diane Meshako and Barbara Rackers. 1989. Star Fire -- Needs Assessment and Preliminary Development Report.

Olgyay, Victor. 1973. Design With Climate. Princeton University Press, Princeton, NJ.

U.S. Army Corps of Engineers. 1988. IWR-MAIN Water Use Forecasting System, Version 5.1 Program and User's Manual.

# Reclamation Policy and Regulatory Issues

This page intentionally left blank.

# EVALUATION OF METHODS TO DETERMINE PLANT PRODUCTIVITY FOR BOND RELEASE[1]

J. G. Skousen[2]

Abstract. State regulatory authorities (SRAs) must evaluate the amount of 1) plant cover and 2) production on revegetated areas and compare them to reference areas or a technical standard to determine if acceptable vegetation exists for reclamation bond release. While most SRAs have adopted a method to evaluate ground cover by plants, only a few SRAs have established a method for determining production. In this study, six methods for predicting aboveground biomass were used to determine if one or more methods would provide an accurate way of assessing aboveground biomass. Correlations were determined between five methods (visual estimation methods were quadrat cover, estimated yield, and average height; indirect methods were disk height and probe) and the dry weight of forage clipped from small (0.25 $m^2$) and large (62 $m^2$) plots on eight reclaimed sites in West Virginia. Forage weight from small plots was also compared to forage weight from large plots. The visual estimations were made by three individuals and their estimates were analyzed for variance. Results showed significant differences among the observer's estimates indicating that these estimation methods would not be dependable for determining revegetation success. Disk height, and averages of average height and estimated yield were significantly correlated with forage dry weight from small and large plots ($r = 0.74$ to $0.87$), while cover and probe readings had r values of 0.43 and 0.51, respectively. Small plot clipping was highly correlated to field yields ($r = 0.85$) and especially on five out of eight sites, but overestimated field yields by 35%. Disk height and small plot clipping are the only two methods used in this study that can be recommended for evaluating aboveground biomass for revegetation success.

Additional Key Words: Revegetation, reclamation, aboveground biomass determination, phase II bond release

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]J. G. Skousen is Assistant Professor, Division of Plant and Soil Sciences, West Virginia University, Morgantown, WV 26506-6108. This article is a contribution from the West Virginia Agric. Exp. Stn. Journal Article No. 2146. This research was primarily supported by the Office of Surface Mining Reclamation and Enforcement, Contract No. EC68-36306, and also by funds appropriated under the Hatch Act.

## Introduction

Standards for determining successful revegetation of mine lands require that both cover and production

of living plants on revegetated areas shall not be less than 90 percent of a reference area or other success standard approved by the regulatory authority (USDI 1983). Most states with approved regulatory programs have adopted some method for evaluating ground cover and, in a recent study by OSMRE (1987), eight of these methods were compared and gave similar results. However, determining production or aboveground biomass of the ground cover on reclaimed land for bond release is more difficult for state regulatory agencies (SRAs), and few SRAs have adopted a method.

Aboveground biomass can be determined by mowing an entire field, baling the material, and calculating the weight of the forage. More commonly, assessment of aboveground biomass is done by clipping the plant material in a number of small plots, drying, and weighing. By clipping numerous plots, an average weight per unit area and a standard error can be calculated. This method has long been an accepted practice for estimating aboveground biomass in hay fields, pastures, and rangeland (Stoddard et al. 1975). Three disadvantages of the method cited by SRAs are: 1) the time required to clip a number of plots is excessive, 2) the labor involved is intensive, tiresome, and tedious, and 3) special facilities are needed to dry and weigh the plant material.

A quicker method involves visually estimating the fresh weight of aboveground biomass in a plot followed by clipping and weighing some of the plots (Pechanec and Pickford 1937). Typically, the worker estimates the weight of ten successive plots, then after the visual estimate is made on the tenth plot, the worker "calibrates" his estimates by clipping and weighing the aboveground plant material in the tenth plot. Many individuals become proficient in making reasonably accurate weight estimates while others do not make reliable estimates (Mueller-Dumbois and Ellenberg 1974, Stoddard et al. 1975).

Double sampling techniques measure plant characteristics and correlate these attributes with clipped weights. Ground cover has been shown to range from no correlation ($r = -0.16$) to good correlation ($r = 0.70$) with biomass (Griggs and Stringer 1988, Pasto et al. 1957). Sward height of alfalfa in Pennsylvania was highly correlated ($r = 0.89$) with aboveground biomass (Griggs and Stringer 1988), and a correlation coefficient of 0.81 was determined between sward height and biomass of a mixed sward in Australia (Michalk and Herbert 1977). Similar relationships have been found in other studies with height, and when combining cover and height as a variable (Alexander et al. 1962, Evans and Jones 1958, Heady 1957, Pasto et al. 1957, Shrivastava et al. 1969, Whitney 1974).

The disk meter is another method of measuring aboveground biomass without clipping many plots (Baker et al. 1981, Bransby et al. 1977) and has shown a high correlation with height and density (bulk) of the vegetation. It is necessary to clip a limited number of plots and develop regression equations between disk height and aboveground biomass on each site for each season (Karl and Nicholson 1987, Palazzo and Lee 1986). Baker et al. (1981) reported that the slopes of the regression lines for disk height and biomass on 40 mixed swards in West Virginia were not significantly different. Palazzo and Lee (1986) found that disk height correlations to biomass on reclaimed sites were best within individual sampling days. Linear correlation (r) between disk height and biomass was 0.61 for sericea lespedeza (Lespedeza cuneata (Dumont) G. Don.) stands and 0.78 for tall fescue (Festuca arundinacea Schreb.) stands.

Capacitance meters (probes) are based on electrical principles and have been used for biomass estimates for many years (Neal and Neal 1973). Crosbie et al. (1985) compared a single probe capacitance meter with other methods of measuring biomass and reported that the probe readings were sensitive and could delineate between pasture grazing treatments (Richardson 1984). Currie et al. (1987), however, found the values obtained from a single probe capacitance meter to be quite variable and the probe appeared to respond to vegetation attributes other than surface area. They also found that a single regression equation did not relate probe readings to aboveground biomass as had been suggested by others (Vickery et al. 1980).

Because methods for determining production to evaluate revegetation success on mine sites have not been established, this research study compares several methods for estimating aboveground biomass on eight reclaimed surface mine sites in West Virginia. The objectives were to determine the accuracy of six techniques for predicting aboveground biomass for use by SRAs in determining revegetation success for bond release.

## Study Sites and Methods

Eight surface mines in West Virginia were selected as research sites (Table 1). All sites had been seeded with a pasture seed mix composed of tall fescue, orchardgrass (Dactylis glomerata L.), ryegrass (Lolium perenne L.), birdsfoot trefoil (Lotus corniculatus L.), and clovers (Trifolium spp.); they

Table 1. Characteristics of eight reclaimed surface-mine sites in West Virginia.

| Characteristic | SITES (CM) Camden | (DP) Dippel | (GR) Garlow | (GS) Gumspring | (KK) King Knob | (LR) Laurita | (HD) Hodrag | (HO) Hobold |
|---|---|---|---|---|---|---|---|---|
| Location (County) | Lewis | Monongalia | Monongalia | Monongalia | Monongalia | Monongalia | Boone | Boone |
| Coal seam | Redstone | Waynesburg | Sewickley | Freeport | Waynesburg | Sewickley | Stockton | Stockton |
| Overburden composition[1] | SH = 50%<br>SS = 30%<br>MS = 15%<br>LS = 5% | MS = 30%<br>SS = 70% | SH = 50%<br>SS = 40%<br>LS = 10% | SS = 80%<br>SH = 20% | SS = 80%<br>SH = 20% | SH = 50%<br>SS = 40%<br>LS = 10% | SS = 50%<br>MS = 50% | SS = 50%<br>MS = 50% |
| Date mined | 1979 | 1978 | 1973 | 1982 | 1978 | 1979 | 1983 | 1980 |
| Mine method[2] | B - L | B - L | Drag | B - L | B - L | B - L | Drag | B - L |
| Date reclaimed | 1981 | 1980 | 1976 | 1985 | 1980 | 1981 | 1985 | 1982 |

[1]LS = limestone, MS = mudstone, SH = shale, SS = sandstone
[2]B - L = bulldozers and front-end loaders, Drag = dragline.

had not been grazed or mowed prior to the study. Lime and fertilizer had been applied according to soil test recommendations. Most mining companies wait until near the end of the growing season to obtain the maximum forage accumulation before requesting a revegetation evaluation by the SRA for bond release. Therefore, all sites in this study were sampled in September 1987.

Four transects of 100 m were randomly located on each site. Every 10 m along the transect line, visual estimates were made by three observers. Each observer carried a separate data sheet for recording his/her estimates, and no discussion took place during visual estimates. The observers were two agronomy graduate students and a professor. They all had experience in making vegetation evaluations before the project started. After estimates were made, probe and disk readings were taken, followed by clipping. To reduce error, the same individual performed the same task on all the sites (i.e., one person placed the quadrat, stick, and disk; another person took all readings with the probe; and one individual did all the clipping). The measurements were conducted in the following order these specifications:

1) Quadrat cover

A quadrat measuring 50 x 50 cm (0.25 $m^2$) was placed on the left side of the transect line. Each observer visually estimated the percent of ground cover by plants inside the quadrat.

2) Estimated yield

Each observer visually estimated the fresh weight (g) of the plant material inside each quadrat. (After the material in the quadrat was clipped as described in Method 6, the fresh plant material in every third quadrat was weighed with a portable metric scale and observers were able to "calibrate" their visual estimated yields).

3) Average height

A meter stick was placed in the center of the quadrat. Each observer used the meter stick to estimate the average height (cm) of the vegetation in the quadrat.

4) Probe

Probe readings were taken by one individual in 10 systematic places within each quadrat that gave an average corrected meter reading (CMR) for each quadrat.

5) Disk

A disk meter (50 cm diameter and 1.9 kg) was placed in the center of the quadrat and dropped onto the vegetation by one individual. The height of the disk (cm) was read.

6) Quadrat weight

After all estimates and readings had been taken, all biomass 1 cm above ground level in the quadrat was hand clipped, dried at 60°C for three

Table 2. Significance of differences among three observers in their visual estimates of quadrat cover, estimated yield, and average height on eight reclaimed sites in West Virginia.

| Method | Site | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | CM | DP | GR | GS | HD | HO | KK | LR |
| Quadrat cover | NS | * | ** | NS | NS | NS | ** | NS |
| Estimated yield | ** | ** | * | * | ** | NS | * | ** |
| Average height | ** | ** | * | ** | ** | ** | ** | ** |

NS = non significant, * = $P < 0.05$, ** = $P < 0.01$.

days, and weighed.

Visual estimates of cover, yield, and height were analyzed by analysis of variance to determine the variability among observers. The three visual estimates were then averaged for each quadrat, combined into a data set with probe and disk height readings, and linear regression was used to fit the relationships between quadrat weights and values obtained by the five methods.

Forage from large areas (65 $m^2$) along the side of each transect on each site was also harvested with a sickle-type mower, raked together, and placed in large gunny sacks. The material from the large plots was also dried and weighed, and linear regression was used to compare the values obtained from the five methods and small plot forage weights to large plot weights. Correlation coefficients for each relationship were also determined (Ray 1982).

## Results and Discussion

### Variation Among Observers

Estimates made by three observers for quadrat cover, estimated yield, and average height were significantly different on most study sites (Table 2). The estimates of quadrat cover showed the least variation among observers in our study. Variation in visual estimates was expected but not to the degree that was found in the study. Reasons for such high variation among observers are not totally clear, especially when considering that the observers were looking at the same small plot at the same time and making their judgments. For example, when estimating average height of vegetation in the quadrat, the observer needed to account for all vegetation in the quadrat and assign one average height value. It was evident that different types of vegetation, bare areas, and the observer's location in relation to the quadrat surely introduced subjectivity into the estimate. Estimated yield was also very subjective, and the estimates made by the observers were often not accurate compared to the weight of clipped forage as shown in the examples in Table 3. These results demonstrate that visual estimates, even when made on the same vegetation in a small plot, were highly variable among individuals. Considerable variation existed even when the observers had frequent chances to "calibrate" their estimates and experience that should have helped them make accurate estimates.

The fact that the estimates made by the observers resulted in significantly different values suggests that methods which require estimation to predict aboveground biomass for bond release are not very reliable. Reliability and repeatability are two criteria that are necessary when selecting a method to determine revegetation success. In order to assess the accuracy of these estimation methods in predicting clipped forage weights, the average of the three observer's values were calculated for each quadrat and compared to the clipped weights.

### Comparison Among Methods

Relationships between each method of measuring aboveground biomass and the dry weight of clipped forage in small 0.25 $m^2$ plots are shown in Figure 1 using the data from all sites ($n = 320$). Average height ($r = 0.81$), disk height ($r = 0.76$), and estimated yield ($r = 0.74$) showed the highest correlations ($p < 0.01$) with the dry weight of clipped forage in small plots. Quadrat cover and the probe showed significant but lower correlations ($p < 0.05$) with quadrat weight ($r = 0.43$ and 0.48, respectively). The same trends were evident when the data was correlated to the forage weight from large plots ($n = 32$) (Figure 2).

Our results compared favorably to

Table 3. An example of the estimated yields of forage in 0.25 $m^2$ quadrats made by three observers, and the average of the three estimates compared to the measured fresh weight after clipping of selected quadrats on two reclaimed sites in West Virginia.

| Location | Estimated yield Observer 1 | 2 | 3 | Ave | Actual fresh weight |
|---|---|---|---|---|---|
| S-T-Q[1] | g/.25$m^2$ | | | | |
| CM-1-3[2] | 280 | 340 | 285 | 301 | 440 |
| 1-6 | 220 | 280 | 100 | 200 | 140 |
| 1-9 | 200 | 180 | 185 | 188 | 180 |
| 2-3 | 260 | 130 | 140 | 177 | 165 |
| 2-6 | 180 | 190 | 130 | 167 | 160 |
| 2-9 | 120 | 100 | 140 | 120 | 130 |
| 3-3 | 90 | 125 | 90 | 102 | 110 |
| 3-6 | 130 | 110 | 130 | 123 | 130 |
| 3-9 | 140 | 120 | 130 | 130 | 120 |
| 4-3 | 80 | 80 | 95 | 85 | 85 |
| 4-6 | 110 | 140 | 100 | 117 | 105 |
| 4-9 | 80 | 50 | 100 | 77 | 70 |
| Ave | 158 | 154 | 135 | 149 | 153 |
| HO-1-3 | 50 | 25 | 50 | 42 | 35 |
| 1-6 | 40 | 35 | 45 | 40 | 60 |
| 1-9 | 90 | 110 | 95 | 98 | 130 |
| 2-3 | 40 | 35 | 80 | 52 | 30 |
| 2-6 | 160 | 130 | 120 | 139 | 150 |
| 2-9 | 210 | 280 | 325 | 272 | 310 |
| 3-3 | 300 | 230 | 275 | 268 | 140 |
| 3-6 | 200 | 290 | 220 | 237 | 200 |
| 3-9 | 240 | 160 | 190 | 197 | 105 |
| 4-3 | 100 | 75 | 60 | 78 | 60 |
| 4-6 | 180 | 165 | 130 | 158 | 120 |
| 4-9 | 60 | 45 | 150 | 85 | 125 |
| Ave | 139 | 132 | 145 | 137 | 126 |

[1]Site-Transect-Quadrat

[2]Differences among observers were significantly different on CM but not on HO.

many studies in the literature using similar techniques. Ground cover by plants was not closely correlated to clipped weights in this study which agreed with other studies (Griggs and Stringer 1988, Pasto et al. 1957), while sward height has generally shown a much better relationship to aboveground biomass (Michalk and Herbert 1977). Probe readings were not highly correlated to clipped forage in this study, and Currie et al. (1987) found correlation coefficients ranged from 0.50 to 0.80 between probe readings and forage weights on Montana rangelands. It is not clear why the probe did not predict aboveground biomass better in this study, but it should be noted that the probe showed higher correlations with clipped weights on sites where the plant material was less than 15 cm tall.

The disk meter has been used in many places and under many conditions to predict aboveground biomass, and our results revealed that the method worked reasonably well on surface mine sites. Table 4 gives the linear regression equation and correlation coefficient for our eight sites in West Virginia. Correlation of disk height with forage weight from small plots resulted in accurate estimates of aboveground biomass on almost all of our sites. Note that the correlation coefficients were all above .50 except for the KK site.

When comparing disk height measurements to the forage weight obtained from large plots, the same pattern was evident (Table 4). Disk heights from three of the sites did not

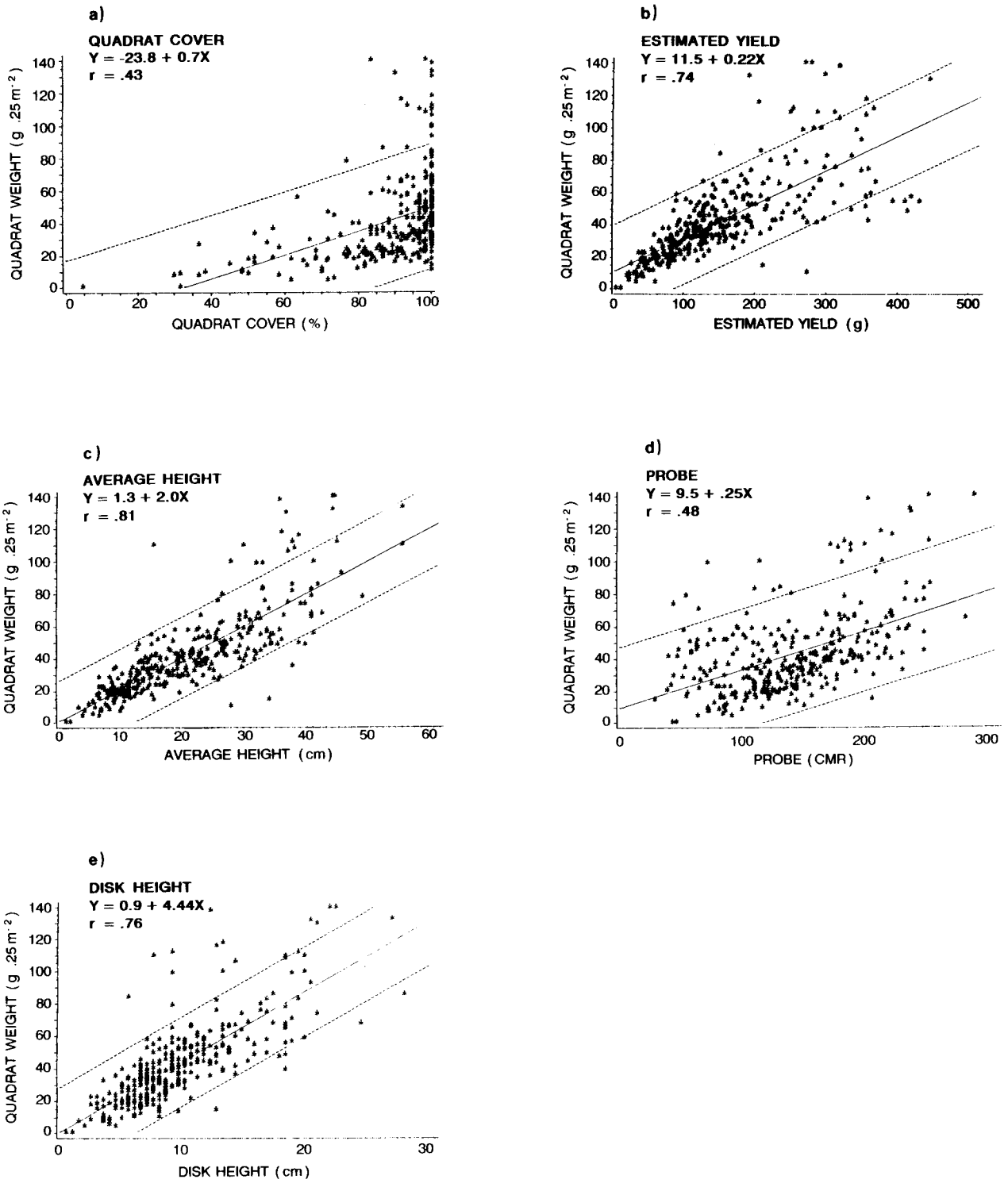



Figure 1 a-e. Relationship between clipped forage weight in small plots quadrat weight) and a) quadrat cover, b) estimated yield, c) average height, d) probe, and e) disk height (n = 320). Dashed lines indicated 90% confidence intervals.

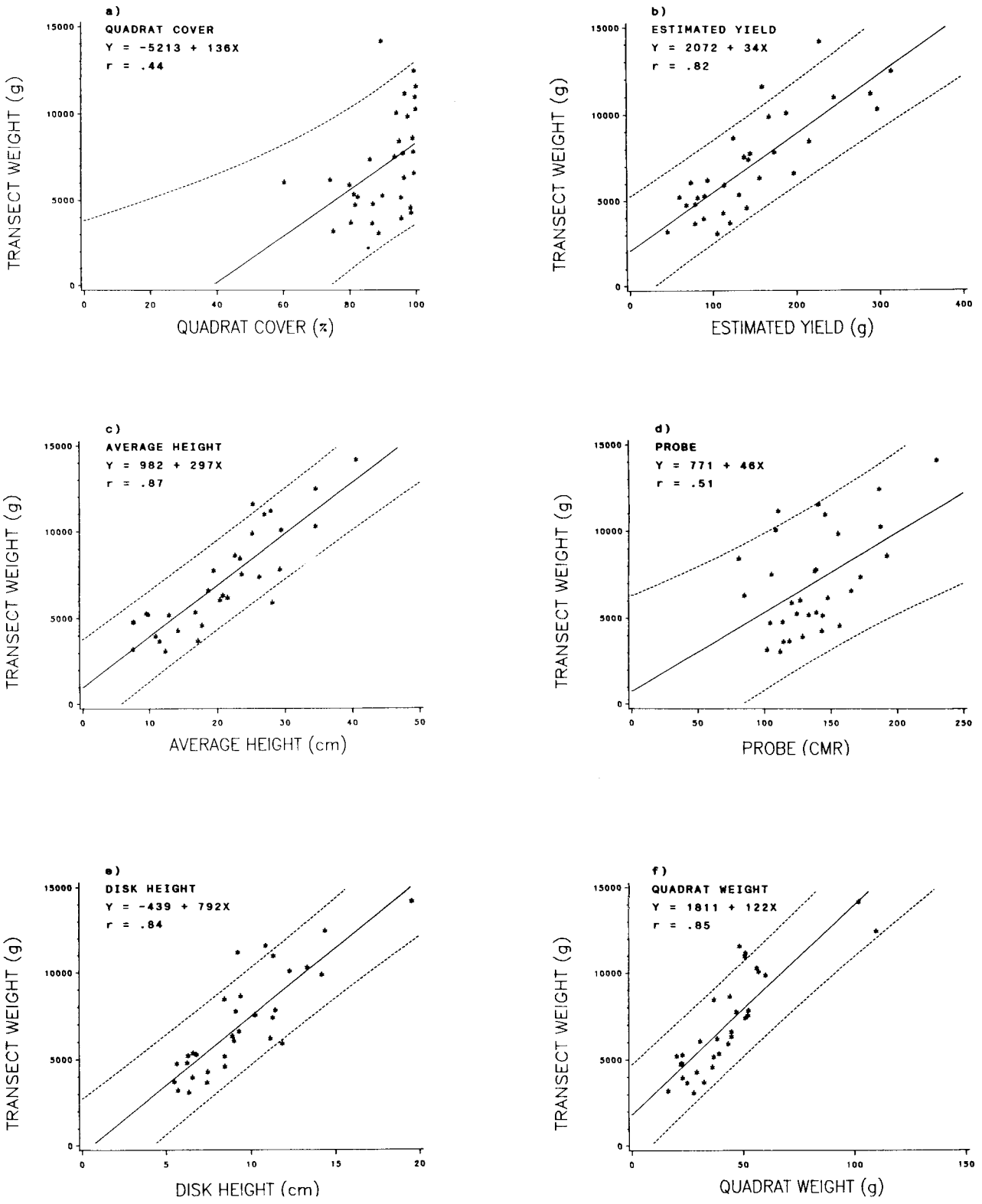



Figure 2 a-f. Relationship between clipped forage weight in large plots (transect weight) and a) quadrat cover, b) estimated yield, c) average height, d) probe, e) disk height, and f) quadrat weight (n = 32). Dashed lines indicate 90% confidence intervals.

Table 4. Linear regression equation and correlation coefficient between disk height (X) and dry weight of forage (Y) in small plots (0.25 $m^2$) and large plots (65 $m^2$) for each surface-mined site in West Virginia. Each site had n = 40, while all sites had n = 320.

SMALL PLOTS

| Site | Equation | r |
|---|---|---|
| CM | Y = -9.2 + 5.3X | .87 |
| DP | Y = -1.4 + 4.1X | .81 |
| GR | Y = 17.7 + 3.0X | .51 |
| GS | Y = 7.2 + 4.5X | .71 |
| HD | Y = 20.5 + 3.6X | .57 |
| HO | Y = 0.8 + 4.4X | .78 |
| KK | Y = 12.4 + 1.6X | .34 |
| LR | Y = 16.8 + 3.0X | .64 |
| All Sites | Y = 0.9 + 4.4X | .76 |

LARGE PLOTS

| Site | Equation | r |
|---|---|---|
| CM | Y = -2151 + 881(X) | .88 |
| DP | Y = 3908 + 160(X) | .72 |
| GR | Y = 3624 + 519(X) | .33 |
| GS | Y = 17 + 730(X) | .97 |
| HD | Y = -11892 + 1626(X) | .78 |
| HO | Y = -1971 + 816(X) | .98 |
| KK | Y = 2984 + 124(X) | .10 |
| LR | Y = 7806 + 226(X) | .40 |
| All Sites | Y = -439 + 792(X) | .84 |

correlate well with large plot weights (GR, KK, and LR) while the GS and HO site were highly correlated. Across all sites (n = 320), the overall correlation coefficient between disk height and small plot forage weight was 0.76, while the r value between disk height and large plot forage weight was 0.84. The use of the disk meter for predicting aboveground biomass shows good potential as a method for evaluating revegetation success. The disk method still involves calibration for correlative analysis on each site.

Small plot clipping has been the standard method for estimating aboveground biomass in many places and under many conditions. As expected, forage weights from small plots correlated well to the forage weights from large plots (r = .85), especially on five of our eight sites (Table 5). However, two of the sites showed no correlation, while the other site showed only a slight correlation.

There has been some concern expressed by SRA's that the weight of forage from small plot clipping does not translate directly into actual field yields that a landowner might obtain with commercial machinery. Converting the average forage dry weight from small plots and from large plots to an equal unit of measurement (i.e., kg/ha) showed that our large plot forage weights only averaged 65% (range of 52 to 80% across sites) of the forage weight calculated from small plots (Table 6). This discrepancy occurs because workers who clip by hand generally remove more forage closer to the ground and also collect more forage than commercial hay mowing and harvesting equipment. Therefore, the amount of material harvested by hand clipping will almost always be greater than the amount harvested from the same area by commercial equipment. In cases where a reference area is not available for a forage weight comparison, the forage weight clipped from small plots must be compared to county averages under specified management. If forage weights from small plot clipping are compared to county averages, correlative analysis should be done to compensate for the overestimation of forage weight from small plot clipping. Across our sites, the compensation values averaged 35%, or about one-third of the material that was hand harvested was not harvested with our sickle-type mower or was not collected with our rake. This estimate of approximately one-third loss of

Table 5. Linear regression equation and correlation coefficient between the forage weight obtained from small plots (X) and the forage weight from large plots (Y) on eight reclaimed sites in West Virginia. Each site had n = 40, and all sites had n = 320.

| Site | Equation | r |
|---|---|---|
| CM | Y = 945 + 117.2(X) | .92 |
| DP | Y = 4527 + 17.8(X) | .48 |
| GR | Y = 10877 - 40.7(X) | -.08 |
| GS | Y = -2350 + 191.4(X) | .99 |
| HD | Y = 4183 + 80.4(X) | .85 |
| HO | Y = 1797 + 119.7(X) | .99 |
| KK | Y = 4827 - 46.7(X) | -.24 |
| LR | Y = 4104 + 125.1(X) | .82 |
| All Sites | Y = 1811 + 122.2(X) | .85 |

material may be a reasonable value to use in comparing the forage weight from hand clipping of small plots to actual field yields obtained by farmers.

Visual estimations for predicting aboveground biomass were significantly different among three observers. In spite of wide variation among observers, the average values for estimated yield and average height showed high correlation to forage weight from small and large plots. Average height and estimated yield, while showing good correlation after averaging, are very subjective among individuals and would not provide reliable estimates of aboveground biomass to evaluate revegetation success. The probe was not accurate in predicting forage weight in this study, while the disk provided good correlation and accurate estimates of aboveground biomass. Forage weight from small plots showed good correlation to large plot forage weights in five out of eight sites. The disk meter and small plot clipping are the only methods used in this study suitable for measuring aboveground biomass. These two methods showed reasonable accuracy and reliability in estimating forage clipped weights and are recommended as methods that could be used by SRAs in evaluating plant production. Small plot clipping has been the standard method for estimating aboveground biomass for many years, but should be calibrated to field harvesting. In this study, approximately 35% (or about one-third) of the weight of the material from hand clipped plots should be subtracted to equate to actual field forage weights.

## Acknowledgments

The author acknowledges the valuable assistance of Silva Strazds and Wael Elmosly for their help during this research project. Appreciation is extended to the following individuals

Table 6. The average forage weight from small plots (0.25 $m^2$) and large plots (65 $m^2$) and these weights converted to a standard unit of measurement on eight reclaimed sites in West Virginia.

| Site | Dry Weight of Forage | | | |
|---|---|---|---|---|
| | Small Plot | | Large Plot | |
| | g | kg/ha | g | kg/ha |
| CM | 33.3 | 1,330 | 4,842 | 745 |
| DP | 26.1 | 1,044 | 4,992 | 768 |
| GR | 48.0 | 1,920 | 8,925 | 1,373 |
| GS | 42.2 | 1,686 | 5,717 | 880 |
| HD | 67.5 | 2,700 | 9,560 | 1,471 |
| HO | 55.5 | 2,219 | 8,440 | 1,298 |
| KK | 22.4 | 895 | 3,782 | 582 |
| LR | 48.8 | 1,950 | 10,202 | 1,570 |
| All Sites | 43.0 | 1,720 | 7,047 | 1,084 |

and companies for the sites sampled in this study: Charlie Miller, Grafton Coal Company; Frank Dippel, Dippel and Dippel Coal Company; John Garlow; Brian Osborn, Laurita Energy Corporation; Brownie Stuart, King Knob Coal Company; Larry Emerson, Hobet Mining Company.

Appreciation is also given to William Bryan, Barton Baker, and John Sencindiver, Division of Plant and Soil Sciences, West Virginia University, and to Hugh Cunningham and Eric Perry of OSMRE for reviewing this paper and providing valuable comments and suggestions.

## Literature Cited

Alexander, C. W., J. T. Sullivan, and D. E. McCloud. 1962. A method for estimating forage yields. Agron. J. 54:468-469.

Baker, B. S., T. V. Eynden, and N. Boggess. 1981. Hay yield determinations of mixed swards using a disk meter. Agron. J. 73:67-69.

Bransby, D. I., A. G. Matches, and G. F. Krause. 1977. Disk meter for rapid estimation of herbage yield in grazing trials. Agron. J. 69:393-396.

Crosbie, S. F., B. M. Smallfield, H. Hawker, M. J. S. Floate, J. M. Keoghan, P. D. Enright, and R. J. Abernethy. 1985. Exploiting the pasture capacitance probe in agricultural research - A comparison with other methods of measuring herbage mass. Invermay Agricultural Research Center, Ministry of Agricultural and Fisheries, Mosgiel, New Zealand.

Currie, P. O., T. O. Hilken, and R. S. White. 1987. Evaluation of a single probe capacitance meter for estimating herbage yield. J. Range Manage. 40:537-541.

Evans, R. A., and M. D. Jones. 1958. Plant height times ground cover versus clipped samples in estimating forage production. Agron. J. 50:504-506.

Griggs, T. C., and W. C. Stringer. 1988. Prediction of alfalfa herbage mass using sward height, ground cover, and disk technique. Agron. J. 80:204-208.

Heady, H. F. 1957. The measurement and value of plant height in the study of herbaceous vegetation. Ecology 38:313-320.

Karl, M. G., and R. A. Nicholson. 1987. Evaluation of the forage-disk method in mixed grass rangelands in Kansas. J. Range Manage. 40:467-471.

Michalk, D. L., and P. K. Herbert. 1977. Assessment of four techniques of estimating yield on dryland pastures. Agron. J. 69:864-868.

Mueller-Dombois, D., and H. Ellenberg. 1974. Aims and methods of vegetation ecology. John Wiley and Sons. New York.

Neal, D. L., and J. L. Neal. 1973. Uses and capabilities of electronic capacitance instruments for estimating standing herbage. J. Brit. Grassl. Soc. 23:81-89.

Office of Surface Mining Reclamation and Enforcement. 1987. An evaluation of methods for measuring ground cover for reclamation bond release. Unpublished report. OSMRE, Eastern Field Operations, Pittsburgh, PA.

Palazzo, A. J., and C. R. Lee. 1986. Biomass determination at reclamation sites using a weighted disk meter. Recl. and Reveg. Res. 4:307-312.

Pasto, J. K., J. R. Allison, and J. B. Washko. 1957. Ground cover and height of sward as a means of estimating pasture production. Agron. J. 49:407-409.

Pechanec, J. F., and G. D. Pickford. 1937. A weight estimate method for determination of range and pasture production. J. Am. Soc. Agron. 29:894-904.

Ray, A. A. 1982. SAS user's guide: statistics. 1982 edition. SAS Institute, Inc. Cary, NC.

Richardson, M. A. 1984. Notes for pasture probe users. Batchelar Agricultural Centre, Ministry of Agricultural and Fisheries, Palmerston North, New Zealand.

Shrivastava, J. P., D. A. Miller, and J. H. Jackobs. 1969. Estimating alfalfa yields. Agron. J. 61:649-651.

Stoddard, L. A., A. D. Smith, and T. W. Box. 1975. Range management. 3rd ed. McGraw-Hill Book Company. New York.

United States Department of Interior. 1983. Revegetation of regraded areas and other lands disturbed by surface coal mining operation; final rule. USDI, Office of Surface Mining Reclamation and Enforcement. Federal Register 48(172):40140-40162.

Vickery, P. J., I. L. Bennett, and G. R. Nichol. 1980. An improved electronic capacitance meter for estimating herbage mass. Grass and Forage Sci. 35:247-252.

Whitney, A. S. 1974. Measurement of foliage height and its relationship to yields of two tropical forage grasses. Agron. J. 66:334-336.

Mineral Resources Policies and the Construction Aggregate Industry. Anthony M. Bauer, Associate Professor, Landscape Architecture Program, Michigan State University, East Lansing, Michigan.

The extraction of construction aggregates is on of the largest mining activities in the United States. It is also an urban land use. While general policies governing the protection and extraction of these resources exist at the state level, typically the lead agencies responsible for planning, permitting and regulating aggregate mining are local units of government. However, few local agencies have established policies and procedures aimed at the protection and exploitation of these resources. This urban proximity and the lack of local mineral resources policies has resulted in the sterilization of significant reserves in regions of the United States where they are in highest demand. Concerned about the loss of these reserves for future exploitation a few state and local agencies have enacted policies for their protection. It appears there are three conditions common to these resources protection policies: 1) an accurate resources inventory; 2) some form of local agency control over the administration of the resources protection policy; and 3) incorporation into the local planning and zoning documents some form of mineral resources districts within which aggregate extraction is the primary use and other uses are permitted only under conditions that do not sterilize existing reserves. A review of these documents suggests that the protection of aggregate reserves can be an effective and compatible part of the local comprehensive planning process.

ADDITIONAL KEY WORDS: Mineral resources districts; Mineral resources inventories; Resources protection policies; California; Ontario, Canada

# RECLAMATION PRACTICE, PERCEPTIONS AND POLICY

## THE AUSTRALIAN EXPERIENCE

David L Whitrow
Executive Officer - Environment & Services
Australian Mining Industry Council

The tide of environmental debate has been running strongly against the Australian mining industry for some two decades, with the most obvious result being a huge increase in the area to which access for mineral exploration and development is denied or severely restricted.

An analysis of a string of increasingly important defeats for the industry indicates that it faces the following inhibitions in the battle for public opinion:

- It lacks a political constituency, due to its capital intensive nature and consequent low number of voting employees.
- The environmental degradation caused by historical mines and the generally unattractive nature of current operations is more evident than current reclamation practice and successes.
- The industry lacks advocates amongst those who lead public opinion - the media, academics and community leaders.
- The industry is a soft target, easily portrayed as a sectional, self pleading, profit driven protaganist for mining at any cost, by the 'disinterested' protectors of the environment and the rights of unborn generations.
- The industry has been unable to establish in the public mind a direct link between individual and the industry's prosperity.
- The industry has argued issues on a case-by-case basis without placing them in the context of the wider community interest.

The process of political decision making depends on the success of sectional interests and pressure groups in establishing their support on particular issues. Activity takes place on three levels - public opinion, as measured by polls, letters to the editor, letters to politicians and so on; the views of opinion leaders as presented by the media; and the political process itself involving the public service and parliamentarians.

As a national organisation, the Australian Mining Industry Council has largely concentrated its efforts at the political level, with a fair degree of success. But increasingly, by the time an issue reaches this level, decision making has been severely constrained by activity at the other two levels. Logical arguments about economic contributions and past environmental performance are irrelevant to a politician who fears the loss of votes.

The mining industry has therefore been gradually excalating a program directed at shifting public opinion on a long-term basis. The paper describes the various facets of this program, discusses successes (and failures) and suggests some future directions.

This page intentionally left blank.

# Mining, Reclamation and the Media

## SURFACE MINING, RECLAMATION AND THE MEDIA
### Panel Discussion

Participants:

Norman Kilpatrick, Surface Mining Research Library
Paul Nyden, Investigative Reporter, Charleston (WV) Gazette
Art Sanda, Eastern Field Editor, Coal Magazine
Ellen Smith, Editor, Mining Regulation Reporter

Moderator:

Bill Case, Acting Editor, College of Agriculture and Forestry, West Virginia University

The participants in this panel represent a range of media perspectives on mining issues. Brief statements of those perspectives are presented below:

**Norman Kilpatrick** **(304) 768-5669**
**Surface Mining Research Library**
**Box 5024**
**Charleston, WV 25361**

"...I am both a news-maker and a news writer for a number of publications... I feel my role is to try to make public the real issues and politics going on in the coal and energy area, as it affects the Mountain State. From 1973-77 I served as the coal consultant to the West Virginia Legislature and attempted the same role for that body.

"It has been my experience, as a news-maker, that the West Virginia news media generally ignores on-going coal issues, unless they are strikes with violence. So the efforts to abandon the rail system in West Virginia really go unreported. The rise of foreign coal imports is rarely commented on. And the development of protectionist laws in other coal states goes unreported.

"At the same time, one state paper The Charleston Gazette, does take a continuing interest in coal, and that view is totally negative. Gazette editor Don Marsh puts it nicely, 'I have come to look upon coal as West Virginia's curse.' And as a news-maker I know what it is to be misquoted by Marsh's top anti-coal writer, Dr. Paul Nyden. (I did get a front page 'correction' from the Gazette to one of his misquotes.)

"My position is likely unique, since I do write for publications....and so I generally can try to bring out issues that elements of the coal industry want unknown and the West Virginia media ignores."

**Paul Nyden** **(304) 348-5100**
**Charleston Gazette**
**1001 Virginia St., E.**
**Charleston, WV 25301**

Paul Nyden is an investigative reporter for The Charleston Gazette, West Virginia's largest-circulation daily newspaper. He is the winner of a number of national awards for his stories on the West Virginia mining industry.

He has written extensively about state regulatory agencies, mining companies, environmental issues and community organizations in mining communities.

**Ellen Smith, Editor** **(703) 528-1244**
**Mine Regulation Reporter**
**Pasha Publications, Inc.**
**1401 Wilson Blvd, Suite 900**
**Arlington, VA 22209**

"MINE REGULATION REPORTER is a regulatory information service for mine operators, contractors, attorneys and government officials. Every two weeks this service offers in-depth reporting on legal issues that affect the mining community. This report briefs the mining industry on federal/state regulations and legislation of health, safety and environmental requirements for surface and underground mines and quarries...

"Every day I go through the Federal Register gathering information that would interest the mining community. I keep in touch with states regarding their state programs. I've subscribed to a service that sends me federal and district court cases taht affect the mining community. Lastly, I cover agencies entrusted with mining regulation such as OSM, IBLA, OHA, MSHA and FMSHRC.

"All this vital information gets transcribed from 'legalese' to plain English that the lay person can understand...There is a document section in the newsletter with reprints of all the documents collected for that issue.

"My goal is to provide companies with information that can help them avoid costly and damaging violations of mining regulations and to help companies and groups prepare compliance or defense strategies."

**Art Sanda** **(304) 255-5653 or 252-6801**
**Eastern Field Editor**
**Coal Magazine**
**101 Pine Villa**
**Beckley, WV 25801**

"Broadly stated, my interest is the coal industry; from the product in the ground, its mining, beneficiation, sale and use, and all the processes and people involved.

"More closely defined, those interests are in new technology and mining methods being developed for underground and surface mining, as well as for the preparation plant and the handling of waste materials and inactive facilities. Due to the nature of our publication, COAL, the technology takes preference over the individual. It is the how, not the who, in which lies our primary interest.

"...Our readers (are) the decision makers within the coal industry and those industries which support it. Our goal is to remain the 'bible' of the coal industry, informing more that entertaining; always factual, never farcical.

"Within the journalistic family, ours is 'friendly press,' meaning our goal is dissemination of information which will be of benefit to the industry. Errors in judgment of the moment, interesting asides which would embarrass the individual while entertaining the reader, and sensationalism are not the grist for our mill. Accuracy is one of our cornerstones, and maintaining it often means working with the principals after the initial interviews...Our success is based on and dependent on a mutual respect and trust between the editors, the subjects, and the readers."

# Overburden Analysis

This page intentionally left blank.

# EVALUATION OF ACID BASE ACCOUNTING DATA USING COMPUTER SPREADSHEETS[1]

Michael W. Smith and Keith B.C. Brady[2]

**Abstract.** Overburden analysis data in the form of acid-base accounting (ABA) can be efficiently and effectively evaluated using spreadsheet programs for personal computers. The Pennsylvania Department of Environmental Resources (PaDER), Bureau of Mining and Reclamation has developed a spreadsheet which calculates several parameters from ABA data including mass-weighted maximum potential acidity (MPA), neutralization potential (NP) and net neutralization potential (NNP). The spreadsheet also summarizes the overburden analysis in terms of the ratio of NP to MPA and the percent sandstone. With the spreadsheet, aggregate overburden characteristics can be summarized for an entire mine site.

Computer spreadsheet software is ideal for performing the numerous repetitive calculations necessary in integrating large quantities of ABA data. The ABA spreadsheet integrates sulfur content and neutralization potential data, as well as sample interval thicknesses, the percentage of each unit spoiled, and overburden unit weights. The area of influence of each drill hole is determined and the actual mass of strata are calculated by taking into account the geometry of the mine site and overburden unit weights. The ABA summary data can be compared using a variety of significance thresholds for NP and percent sulfur, and other factors can readily be changed to review their impact. The spreadsheet approach permits more complex and detailed analysis of overburden data and facilitates comparison between calculation methods.

Additional Key Words: overburden analysis, acid base accounting, computer spreadsheet.

## Introduction

Acid base accounting (Sobek et al. 1978) is the most commonly used overburden analysis technique for predicting the water quality likely to result from a coal mining operation. In Pennsylvania, acid base accounting (ABA) has been used since about 1979 and now accompanies approximately fifty percent of current surface mining permit applications. ABA evaluates the maximum potential acidity (MPA) and the neutralization potential (NP) from individual strata and expresses them as tons per thousand tons of calcium carbonate equivalent. As such, it has been used to identify potentially acid-forming or alkaline-forming strata. Although not originally intended for the purpose, ABA is also used to predict postmining water quality. This is done by individually sampling all of the strata within a proposed mine site. The weighted aggregate of all samples are considered together to evaluate the potential of the site to produce acidic or alkaline water following mining.

Using ABA data for prediction of postmining water quality requires the integration of the chemical characteristics of each individual stratum in order to characterize the entire mine site. The large volume of ABA data which accompanies a typical permit application makes it all but impossible to accomplish this intuitively. However, the widespread availability of personal computers and spreadsheet software greatly enhance one's ability to quantitatively and objectively evaluate ABA data and to evaluate it using a variety of procedures. This paper discusses a method developed by the Pennsylvania Department of Environmental Resources (PaDER) that offers a reliable and convenient means of summarizing large volumes of overburden data.

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]Michael W. Smith and Keith B.C. Brady are Hydrogeologists, Pennsylvania Department of Environmental Resources, Bureau of Mining and Reclamation, Harrisburg, PA 17120.

Using ABA data for prediction of postmining water quality requires the integration of the chemical characteristics of each individual stratum in order to characterize the entire mine site. The large volume of ABA data which accompanies a typical permit application makes it all but impossible to accomplish this intuitively. However, the widespread availability of personal computers and spreadsheet software greatly enhance one's ability to quantitatively and objectively evaluate ABA data and to evaluate it using a variety of procedures. This paper discusses a method developed by the Pennsylvania Department of Environmental Resources (PaDER) that offers a reliable and convenient means of summarizing large volumes of overburden data.

## History of Quantitative ABA Evaluation

Early users of ABA data as a predictor of postmining water quality tended to balance the MPA against the NP for an overburden column giving equal weight to the strata at the top and bottom. It was presumed that the postmining water quality would be determined by whichever factor predominated. For example, where NP exceeded MPA, alkaline drainage would result. Issuance of surface mining permits based on this assumption frequently resulted in severe acid mine drainage pollution (Brady and Hornberger 1989) leading to the realization that modifications to ABA review procedures were required for reliable predictions of postmining water quality.

The need for a simple method of computing and summarizing ABA data became essential in the early 1980's when coal companies began to propose the addition of supplemental alkaline material as a means of offsetting an NP deficiency (MPA > NP) and the potential for acid production at a mine site. Because of the large number of arithmetic operations required, the concept of using computer spreadsheets for calculating the deficiencies soon followed. Originally, calculations were weighted only according to thickness. However, in most cases the topography is hilly with flat-lying strata so that the uppermost strata are not as aerially extensive as lower strata, making volume or mass-weighted calculations essential. Later developments in the utility of computer spreadsheet software enabled the PaDER to design more sophisticated spreadsheets which considered mass or volumetric weighting of each overburden sampling interval.

As the use of ABA for postmining water quality prediction became more widespread, the spreadsheet was also used to summarize the data numerically for comparison with postmining water quality. Again, this required a quantitative summary of the data which took into account the actual volume or mass of each overburden sampling interval. Two recent studies by diPretoro and Rauch (1988) and Erickson and Hedin (1988) have compared numerical summations of ABA data with postmining water quality. Because of the large number of mine sites required for studies of this type and the effort required to measure the areal extent of each stratigraphic interval, precise volumetric calculations were not performed. diPretoro and Rauch calculated volume-weighted summary values by assuming an idealized right-triangle shaped area to be mined. Erickson and Hedin used essentially the same method. This assumption was easily applied to single core log data where the coal was not steeply dipping. However, it can only be considered as an approximate technique for volumetric adjustment of ABA data. In these studies, aggregate MPA and aggregate NP were calculated. These volumes were then used to determine the net neutralization potential (NNP = NP - MPA) of the aggregate overburden volume and the NP/MPA ratio. Since the strata are represented more realistically, they found that volume-weighted calculations yielded better results than a columnar (equal unit-volume for all strata) approach.

## Methodology

The current spreadsheet program used by the PaDER includes several important parameters useful in summarizing aggregate overburden characteristics. These include mass weighting using selected unit weights according to rock type, the percentage of a unit that is spoiled, and threshold significance levels for NP and sulfur content. The current spreadsheet represents a culmination of efforts over the last several years. The concepts presented here could be adopted with almost any commercial spreadsheet software.

Figure 1 is a printout of the overburden calculation spreadsheet used by PaDER, showing the layout of the input data and calculated results. Data input is designed to mimic ABA reports currently in use. Manual data input from the keyboard is kept to a minimum. The only required input columns are bottom depth, rock type, % sulfur, NP, and fizz. All other values are either calculated or can use predetermined defaults.

### Area of Influence

The spreadsheet is designed to analyze ABA data for individual drill holes. Where only one drill hole is available to characterize the overburden at a given mine site, quantitative analysis is simplified, although a single hole may not be representative of the entire mine site. In most cases, the information available from surface mining permit applications in Pennsylvania is complete enough such that a more rigorous approach to quantitative evaluation of overburden analysis data can be undertaken. Virtually every site in the state has two (and usually more) overburden holes. Using the outcrop boundary and the limits of mining as delineated in the permit application, the area of influence of each drill hole can be approximated using the Thiessen polygon method (Davis 1973 and Brassington 1988).

An example Thiessen polygon construction at a hypothetical mine site is shown in Figure 2. The areas within each polygon are closer to the data point (drill hole) in the center of the polygon than they are to any other data point. In brief, the polygons are constructed by drawing lines between each drill hole. Each line is then bisected with a perpendicular and the perpendiculars are extended to form polygons. The area of each polygon can be calculated using a planimeter and this value can be used as a factor in determining actual volume or mass or for applying relative weights to each drill hole. Although there are other more elaborate methods which could be employed, they usually require many data points and lengthy, complex calculations. The Thiessen method is more representative than simple arithmetic

OVERBURDEN ANALYSIS SPREADSHEET

OPERATOR: MINE SITE 1
PERMIT NO: DR HOLE: DH EXAMPLE
COUNTY: LYCOMING TOWNSHIP: PINE

THRESHOLD VALUES: SULFUR 0, NP 0.00, FIZZ 0
NUMBER OF INTERVALS 21 (Alt A; Alt B to execute)

| | | | |
|---|---|---|---|
| CLAY | CL | 3450 | |
| SHALE | SH | 3700 | |
| SILTSTONE | ST | 3750 | ALK ADD(t/a CaCO3): |
| SANDSTONE | SS | 3670 | COAL SEAMS: |
| LIMESTONE | LS | 3670 | STATE PLANE ZONE: |
| COAL | CO | 1800 | FEET (NORTH/SOUTH): |
| CARBONOLITH | CB | 2580 | FEET (EAST/WEST): |
| OTHER | OT | 2000 | SURFACE ELEV. (FT): |

| BOTTOM DEPTH (FT) | THICKNESS FEET | ROCK TYPE | FIZZ RATING | SULFUR % | NP | DEFICIENCY /EXCESS | ACREAGE | UNIT WT TONS/AC-FT | FRACTION SPOILED | TONS MPA | TONS NP | NET NP (TONS) | TONS OF OVERBURDEN |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 7.00 | 7.00 | OT | 0 | 0.05 | 1.90 | 0.34 | 10.50 | 2000 | 1.00 | 229.69 | 279.30 | 49.61 | 147000 |
| 14.00 | 7.00 | SS | 0 | 0.04 | 1.65 | 0.40 | 13.98 | 3670 | 1.00 | 448.89 | 592.54 | 143.65 | 359115 |
| 21.00 | 7.00 | SS | 0 | 0.05 | 1.90 | 0.34 | 15.72 | 3670 | 1.00 | 630.94 | 767.22 | 136.28 | 403800 |
| 28.00 | 7.00 | SH | 0 | 0.07 | 28.99 | 26.80 | 17.46 | 3700 | 1.00 | 989.08 | 13107.86 | 12118.78 | 452151 |
| 33.00 | 5.00 | CO | 0 | 0.50 | 2.41 | -13.22 | 18.70 | 1800 | 0.10 | 262.97 | 40.56 | -222.41 | 16830 |
| 37.00 | 4.00 | CL | 0 | 0.08 | 3.42 | 0.92 | 19.69 | 3450 | 1.00 | 679.44 | 929.48 | 250.03 | 271776 |
| 46.00 | 9.00 | SS | 0 | 0.07 | 2.41 | 0.22 | 21.93 | 3670 | 1.00 | 1584.53 | 1745.70 | 161.17 | 724358 |
| 60.00 | 14.00 | SS | 0 | 0.09 | 5.19 | 2.38 | 25.41 | 3670 | 1.00 | 3671.77 | 6775.64 | 3103.87 | 1305519 |
| 71.00 | 11.00 | SS | 0 | 0.12 | 15.82 | 12.07 | 28.14 | 3670 | 1.00 | 4260.41 | 17973.25 | 13712.84 | 1136110 |
| 81.00 | 10.00 | SH | 0 | 0.28 | 30.25 | 21.50 | 30.63 | 3700 | 1.00 | 9915.58 | 34279.58 | 24364.00 | 1133209 |
| 82.00 | 1.00 | SH | 0 | 1.29 | 2.41 | -37.90 | 30.88 | 3700 | 1.00 | 4605.31 | 275.32 | -4329.99 | 114240 |
| 86.00 | 4.00 | CL | 0 | 0.49 | 9.75 | -5.56 | 31.87 | 3450 | 1.00 | 6734.47 | 4288.07 | -2446.40 | 439802 |
| 90.00 | 4.00 | CL | 0 | 0.59 | 7.09 | -11.35 | 32.86 | 3450 | 1.00 | 8361.74 | 3215.44 | -5146.30 | 453518 |
| 92.50 | 2.50 | CO | 0 | 0.95 | 2.66 | -27.03 | 33.48 | 1800 | 0.10 | 447.34 | 40.08 | -407.26 | 15068 |
| 98.00 | 5.50 | CL | 0 | 0.54 | 3.57 | -13.31 | 34.85 | 3450 | 1.00 | 11159.56 | 2360.87 | -8798.70 | 661308 |
| 100.00 | 2.00 | CO | 0 | 0.74 | 1.65 | -21.48 | 35.35 | 1800 | 0.10 | 294.28 | 21.00 | -273.28 | 12725 |
| 104.00 | 4.00 | CL | 0 | 0.19 | 5.45 | -0.49 | 36.34 | 3450 | 1.00 | 2977.81 | 2733.31 | -244.49 | 501525 |
| 121.00 | 17.00 | SH | 0 | 0.12 | 33.04 | 29.29 | 40.57 | 3700 | 1.00 | 9568.66 | 84306.30 | 74737.63 | 2551643 |
| 127.00 | 6.00 | SH | 0 | 0.10 | 3.93 | 0.81 | 42.06 | 3700 | 1.00 | 2917.74 | 3669.36 | 751.61 | 933678 |
| 131.00 | 4.00 | CO | 0 | 0.68 | 3.17 | -18.08 | 43.05 | 1800 | 0.10 | 658.69 | 98.26 | -560.43 | 30997 |
| 132.00 | 1.00 | CL | 0 | 0.10 | 2.91 | -0.22 | 43.30 | 3450 | 1.00 | 466.83 | 434.71 | -32.12 | 149385 |
| | | | | | | | | | | 70865.73 | 177933.85 | 107068.12 | 11813759 |

| | | | | | |
|---|---|---|---|---|---|
| TOTAL OVERBURDEN VOL.(ACRE-FT): | 3305 | TOTAL (TONS): | 70865.73 | 177933.85 | 107068.12 |
| PERCENT SANDSTONE: | 32% | TOTAL (TONS/THOUSAND): | 6.00 | 15.06 | 9.06 |
| DER RATIO: | 2.51 | diPRETORO METHOD (T/TT): | 6.71 | 15.77 | 9.06 |
| diPRETORO RATIO: | 2.35 | | | | |
| TONS/ACRE REQUIRED (1:1): | 2472.70 EXCESS | | | | |

ABA SUMMARY VALUES USING % SULFUR * 62.5 = MPA:

| | | | | | |
|---|---|---|---|---|---|
| DER RATIO: | 1.26 | TOTAL (TONS): | 141731.46 | 177933.85 | 36202.38 |
| diPRETORO RATIO: | 1.17 | TOTAL (TONS/THOUSAND): | 12.00 | 15.06 | 3.06 |
| TONS/ACRE REQUIRED (1:1): | 836.08 EXCESS | diPRETORO METHOD (T/TT): | 13.42 | 15.77 | 2.34 |

Figure 1. Example of acid base accounting spreadsheet printout.

averaging of drill hole data since each drill hole represents a volume of overburden proportional to its actual location within the mine site.

## Volumetric Calculations

In contrast to thickness weighting and volume weighting using the triangle approximation of diPretoro, the overburden analysis spreadsheet uses actual measured acreages of the area to be mined. For most uses it is impractical to measure the area for each sample (stratigraphic) interval, which would be necessary for precise volumetric calculations. As an alternate and much simpler method, the acreage for the uppermost unit and lowermost unit can be determined, with the spreadsheet interpolating the areas for each intervening sample interval. In most instances, this method will provide acceptable volumetric calculations under a wide range of topographic conditions. The area covered by each sample interval is estimated by determining its depth in the drill hole at the middle of the interval. This technique is illustrated in Figure 3. The spreadsheet calculates this value using the equation:

$$A_I = A_T + [((D_{TOP}+D_{BOT})/2)/D_{TOT}] \times (A_B-A_T) \quad (1)$$

Where: $A_I$ = Acreage at sample (stratigraphic) interval

$A_T$ = Acreage represented by the middle of the uppermost sample interval

$A_B$ = Acreage represented by the middle of the lowermost sample interval

$D_{TOT}$ = Drill hole total depth

$D_{TOP}$ = Depth at top of sample interval

$D_{BOT}$ = Depth at bottom of sample interval

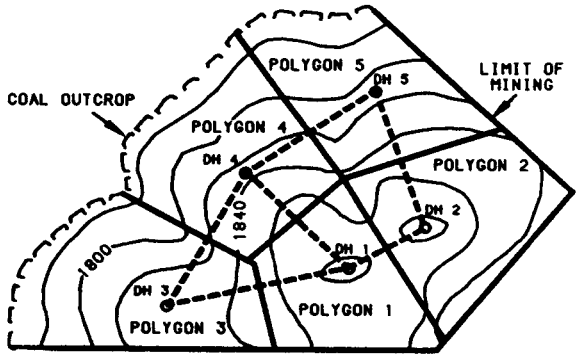



Figure 2. Typical mine plan map showing Theissen polygon method of defining area of influence for each drill hole. Dashed lines are construction lines drawn between drill holes. Polygons are constructed by bisecting the construction lines with perpendiculars. Outer limits of polygons are defined by mining limits and coal outcrop boundary. Interior polygons (1 and 2) are calculated as columns (top acreage = bottom acreage). Remaining polygons calculated using different top and bottom acreages.

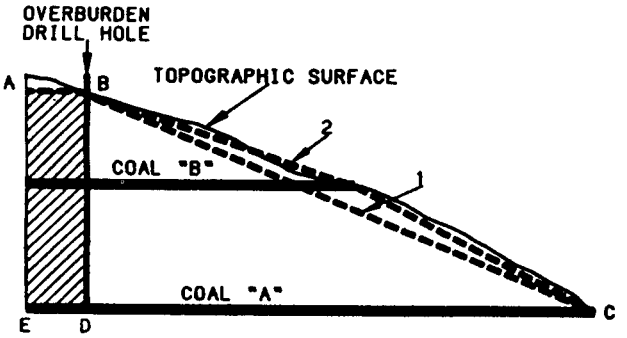



Figure 3. Schematic cross-sectional view of area to be mined with overburden drill hole slightly below point of maximum overburden thickness. diPretoro volume-weighting approximation shown by line 1 includes the area of triangle BCD. Spreadsheet method calculates area of triangle BCD plus the cross-hatched rectangle ABDE. Where necessary due to irregular topography, spreadsheet-calculated volumes can be improved by dividing drill hole into two or more intervals as shown by line 2.

This equation provides a reasonable approximation of acreages covered by each sample interval provided that the strata are not steeply dipping and that the topography, if viewed in cross section, is not markedly convex, concave or irregular. It can be appropriately applied to area mines, contour mines, and mountaintop removal mines and works equally well in steep or gentle topography. The data input requirements are minimal. The acre-acreage represented by the lowermost unit (usually measured as the coal outcrop limit) and the uppermost unit must be provided. Where no acreages are provided, the default value of 1.0 is used for all intervals and the spreadsheet performs column (thickness-weighted) calculations. If the drill hole is located to represent maximum cover, then the top acreage is very small and the default value of 1.0 acre is probably appropriate.

If the drill hole is located where overburden thickness is at a maximum, then the modeled geometry resembles a cone and the calculated results should approximate those using diPretoro's method. However, for most applications where the uppermost unit covers a significant acreage but less than the lowermost unit, the site geometry is modeled as a truncated cone. Area mines are represented with nearly equal top and bottom acreages. For multiple drill hole sites, a combination of geometries may be most appropriate. For example, where interior polygon sections which are bounded by other polygons rather than the coal outcrop limits exist, they are best represented as area mine sections with the bottom acreage equal to the top acreage.

Where unusual topography occurs which renders the linear interpolation method inaccurate, the overburden analysis can be divided into two or more sections. Measured acreages are used for the upper and lowermost units of each section and the values are interpolated between measured units.

Actual volumes are calculated by multiplying the unit thickness by its area. But ABA data are expressed as tons per thousand tons of overburden. Therefore, it is more appropriate to evaluate the data in the same units. The importance of using weight rather than volume units is intensified where the unit weights of the rock types varies significantly. ABA values are expressed in terms of absolute quantities by multiplying their volumes by unit weights. The spreadsheet includes default unit weight values for each rock type. Unit weights typical of Allegheny group rocks in western Pennsylvania were obtained from Geyer and Wilshusen (1982). Coal and carbonolith (carbonaceous sedimentary rock) unit weights were determined from PaDER file data. Through a series of nested "IF" operators, the appropriate unit weight is selected according to the rock type specified. The unit weight for any particular rock type can be changed by entering the desired value in the unit weight table.

Most ABA reports include the "pavement" or underlying strata, which can vary in thickness. Although this material will not be mined, it is presumed that at least the uppermost portion of it will be disturbed by mining and that it will have some impact on mine drainage chemistry. For most applications, it is appropriate to include the upper foot of pavement material as the lowermost sample interval in the spreadsheet.

Fraction Spoiled

Previous quantitative evaluations of ABA data invariably discounted the coal, presuming that its recovery in the mining operation was 100% complete. Typically this is not the case, and pit losses in the order of 5 to 20% can be expected. The spreadsheet includes a column to indicate the fraction of

material to be returned to the backfill. For overburden, this will be 1.0 (i.e., 100%) since all of the overburden will be returned. Thus, the default value is set at 1.0. For coals which will be removed, a fraction representing pit losses can be entered (such as 0.10). This permits the calculations to reflect the retention of potentially acid forming pit cleanings and coal rejects. It may also be possible that various alternate mining schemes are proposed, such as removal of some overburden strata either for commercial purposes or to minimize acid formation. The fraction spoiled column can be used to reflect this.

## NP and MPA

For each stratigraphic (or sample) interval the maximum potential acidity ($MPA_I$) and neutralization potential ($NP_I$) are calculated and expressed as total tons $CaCO_3$ equivalent using the following equations:

$$MPA_I = \%Sulfur \times 31.25 \times T_I \times A_I \times Unit\ Wt. \times Fraction\ Spoiled \quad (2)$$

$$NP_I = NP \times T_I \times A_I \times Unit\ Wt. \times Fraction\ Spoiled \quad (3)$$

Where: $T_I$ = Thickness of sample (stratigraphic) interval (feet)

$A_I$ = Acreage covered by sample interval

These two values represent the total amount of potential acidity and neutralization potential for a single interval. In this sense, the spreadsheet goes one step beyond volumetric adjustment of the acid base accounting data by using actual densities to calculate total tonnages. Then, for each interval, the net NP ($NNP_I$) is determined by subtracting $MPA_I$ from $NP_I$. The total tons of overburden for a sample interval is calculated by multiplying thickness times unit weight times area. This number will be used for subsequent summary parameters.

## Alkaline Addition Rate

For mine sites where off-site alkaline materials will be imported, the impact of the additional alkaline material, in terms of ABA summary parameters, can be examined by including it in the ABA calculations. A spreadsheet entry (labelled ALK ADD (tons/ac CaCO3) in Figure 1) is used. The "alkaline addition" rate is entered in units of tons $CaCO_3$/acre. If a different material or impure limestone will be used, it must be converted to $CaCO_3$ equivalent. Since the acreage of the bottom sample interval equals the total surface acreage represented by this drill hole, the alkaline addition rate per acre is multiplied by this number to obtain the total quantity of imported alkaline material as $CaCO_3$. This value is added to the column of NP values and is reflected in the total NP. It is also added to the total overburden weight for calculation of NP and MPA in tons/1000 tons.

## Summary Parameters

The principal purpose of the overburden spreadsheet is to provide useful parameters which summarize the aggregate overburden characteristics. The summary parameters calculated by the spreadsheet, along with their formulas, are listed in Table 1. The most obvious parameters are calculated total tonnages of MPA, NP, NNP, and total tons of overburden. They represent absolute amounts expressed in tons and are calculated by summing the value for each individual stratigraphic interval. Total tons MPA, NP and NNP are also expressed as tons per thousand tons of overburden representing the entire overburden volume as if it were a single homogeneous sample.

Table 1. ABA spreadsheet summary parameters

| Parameter | Description | Formula |
|---|---|---|
| tons overburden | total weight of overburden | Σ tons overburden for each sample interval |
| MPA (tons) | total maximum potential acidity | Σ MPA for each sample interval |
| MPA[1] (tons/thousand) | " | (MPA (tons)/tons overburden)x 1000 |
| NP (tons) | total neutralization potential | Σ NP for each sample interval[2] |
| NP[1] (tons/thousand) | " | (NP (tons) / tons overburden) x 1000 |
| NNP (tons) | total net neutralization potential | NP (tons) - MPA (tons) |
| NNP[1] (tons/thousand) | " | (NNP (tons)/tons overburden) x 1000 |
| overburden volume (acre-ft.) | total volume of overburden | Σ (thickness x acreage x fraction spoiled) for each interval |
| percent sandstone | total volume - weighted percent sandstone | (Σ (thickness x acreage x fraction spoiled) for each sandstone interval/overburden volume) x 100 |
| DER ratio | NP to MPA ratio | NP (tons)/ MPA (tons) |
| diPretoro ratio | " | NP[1] (tons)/MPA[1] (tons) |
| tons/acre required | $CaCO_3$ equivalent required for DER Ratio = 1.0 | NNP (tons) / acreage at bottom sample interval (> 0 = excess; < 0 = deficiency) |

[1] for diPretoro method, calculated as described by diPretoro 1988.
[2] includes NP from alkaline addition

For comparison, the same parameters are calculated using diPretoro's (1988) triangle-volume weighting method. As part of a study on the effectiveness of alkaline addition in ameliorating acid mine drainage (Brady et al. 1990), the spreadsheet was used to characterize overburden conditions for ten different mine sites. The results using the acreage interpolation method and diPretoro's method are expressed as aggregate NNP and as the NP/MPA ratio and are compared in Table 2. Although they always differ slightly, the results from both methods are usually fairly close, especially the NP/MPA ratio. In most of these cases, the diPretoro method calculated a somewhat lower NNP. This apparently results from the diPretoro triangle-approximation method which tends to underestimate upper, generally higher NP strata where the mine site configuration resembles a truncated cone or where overburden holes were not drilled at maximum cover.

## Tons per acre NP required

Especially where alkaline addition requirements are to be determined, it is useful to know the limestone application rate which would be required to fulfill any net neutralization potential deficiency. In practical terms, this amount is expressed as the required tons per acre. It is the total tons NNP divided by the total acreage represented by the drill hole (i.e., the acreage of the bottom sample interval). Positive NNPs are indicated as an excess. Negative NNPs are indicated as deficiencies. Of course, this does not presume that merely bringing the NNP to zero indicates a suitable alkaline addition rate. It is presented only as a summary parameter to be used in

Table 2. Comparison[1] of Net NP Values and NP/MPA Ratios as determined by Pa DER spreadsheet and diPretoro methods[2]

| SITE NO. | Net NP (tons/thousand) | | | NP/MPA Ratio | | |
|---|---|---|---|---|---|---|
| | Spreadsheet | diPretoro | difference | Spreadsheet | diPretoro | difference |
| 1 | 6.10 | 6.94 | +0.84 | 2.33 | 2.35 | +0.02 |
| 2 | -3.92 | -5.93 | -2.01 | 0 | 0 | 0 |
| 3 | 2.85 | 2.54 | -0.31 | 1.36 | 1.31 | -0.05 |
| 4 | 2.77 | 1.60 | -1.17 | 1.20 | 1.11 | -0.09 |
| 5 | 15.69 | 15.29 | -0.40 | 2.33 | 2.26 | -0.07 |
| 6 | 4.10 | 3.05 | -1.05 | 1.52 | 1.34 | -0.18 |
| 7 | -3.38 | -4.52 | -1.14 | 0.59 | 0.55 | -0.04 |
| 8 | 5.55 | 4.37 | -1.18 | 2.52 | 2.12 | -0.40 |
| 9A | -3.40 | -4.89 | -1.49 | 0.25 | 0.20 | -0.05 |
| 9B | -6.16 | -5.21 | +0.95 | 0.06 | 0.09 | +0.03 |
| 9C | -9.68 | -11.98 | -2.30 | 0.06 | 0.05 | -0.01 |
| 10 | 170.47 | 144.14 | -26.33 | 6.74 | 4.64 | -2.10 |

[1]Mine site data from Brady et al. 1990. [2] Method used in diPretoro 1988.

the review of ABA data. Also, where this parameter is used in combination with the significance threshold values discussed below, the limestone requirement may be very different than for a straightforward balance of total NP versus total MPA.

If an alkaline addition rate has already been specified in the spreadsheet input data, it will be reflected in all of the applicable summary parameters. Therefore, the amount required is in addition to the specified value. By comparing the same spreadsheet with and without alkaline addition or at various alkaline addition rates, the impact of the alkaline addition on the ABA summary parameters can easily be observed. In most cases, only very large (500 tons/acre or more) alkaline addition rates have a noticeable impact on these values.

## NP/MPA Ratio

diPretoro (1988) used the ratio of aggregate neutralization potential to maximum potential acidity as an ABA summary parameter. When this ratio equals 1, the NP and MPA are theoretically equal. This parameter is calculated by dividing total tons NP by total tons MPA. diPretoro found that mine sites with a ratio less than 2.4 generally resulted in acid mine drainage, whereas mine sites with ratios above 2.4 usually produced alkaline drainage. For comparison, both the numbers derived from the acreage interpolations (labelled as the DER ratio on Figure 1) and the diPretoro triangle method (labelled diPretoro ratio) are shown.

## Percent Sandstone

Several studies have found (diPretoro and Rauch 1987, Williams et al. 1982, Brady et al. 1988, Kanai et al. 1989) that where overburden is mostly composed of sandstone, acid drainage predominates. Consequently, a summary value showing the percentage of sandstone overburden was incorporated into the spreadsheet. This is a volume percentage calculated by summing the volume of each sample interval identified as sandstone and dividing this by the total overburden volume.

## Significance Thresholds

Brady and Hornberger (1989) suggested threshold values of NP = 30 tons/1000 tons $CaCO_3$ with "fizz" and percent sulfur = 0.5 as reasonable guidelines to define potentially alkaline or acid-producing strata, respectively. Overburden calculations for alkaline addition were made using this scheme, such that $CaCO_3$ requirements were based only on strata with sulfur contents and NPs exceeding these thresholds. The spreadsheet was designed to perform these or similar calculations. Threshold values for sulfur content, NP, and fizz rating are defined near the top of the spreadsheet. Through a series of "IF" operators, MPA and NP values are only calculated for sample intervals which equal or exceed the threshold value. Using threshold values of 0.5% sulfur, NP = 30, and fizz = 1, for example, MPA will be determined only if the sulfur content equals or exceeds 0.5%. Where sulfur < 0.5%, MPA = 0. NP will only be calculated where the sample NP equals or exceeds 30 and fizz is greater than or equal to 1. Therefore, where NP < 30 or fizz = 0, NP = 0.

Presumably, this method can be used to eliminate the NP or MPA contribution from strata which are insignificant in terms of the production of acidity or alkalinity. Of course, if no threshold calculation is desired, then zero is entered for the threshold values. Any combination of thresholds can be used. In this manner, the summary parameters can be readily calculated for a variety of threshold values.

## Sulfur Content - Carbonate Equivalence

Cravotta and others (1990) have suggested that by using a stoichiometric equivalence factor of 31.25 to compare percent sulfur (MPA) to NP, the actual neutralization requirements may be understated by a factor of two. The alternate equivalence factor to convert percent sulfur to MPA is 62.5. Accordingly, the spreadsheet also reflects this alternate method. All of the summary values are calculated using both the 31.25 and 62.5 equivalence factors.

## Summary and Discussion

Computer spreadsheets are an effective means of summarizing and evaluating ABA overburden analysis data. Volumetric and mass-weighted calculations based on actual mine site geometry and overburden unit weights can easily be performed. Underlying assumptions can be readily changed and the spreadsheet automatically performs any recalculations. Through the use of summary parameters, ABA data from an entire mine site can be integrated to form a conceptual picture of the site's aggregate overburden conditions. Moreover, the spreadsheet can be used to perform various quantitative calculations such as alkaline addition requirements.

A study by Brady et al. (1990) used the spreadsheet method of overburden analysis computation to evaluate the effectiveness of alkaline additives to surface mines in preventing pollution from acid mine drainage. Because it considers rock mass, the fraction spoiled, and a closer approximation of actual site geometry, the spreadsheet offers a better representation of actual field conditions than previous summary methods.

The recent availability of extensive database capabilities and geographic information systems (GIS) provides new uses for summarized ABA data. In many cases, overburden analysis data is functionally irretrievable and can be used only by initiating a very laborious file search and data compilation effort. Computer storage of summarized ABA data through such a database and retrieval system could greatly facilitate its use for cumulative hydrologic impact analysis (CHIA) and comparison of overburden quality with postmining water quality. The computer spreadsheet discussed herein or its modifications could readily support such a system.

## Acknowledgments

The authors wish to thank Charles Cravotta III and Richard S. diPretoro for their review of the manuscript and helpful comments. We also are indebted to Vincent Meehan and the numerous PaDER Bureau of Mining Reclamation staff members for their extensive trial-and-error debugging and critical review of the ABA spreadsheet.

## Literature Cited

Brady, K.B.C. and R.J. Hornberger. 1989. Mine drainage prediction and overburden analysis in Pennsylvania. *In* Proceedings: West Virginia Surface Mine Drainage Task Force Symposium. (Morgantown, WV, April 25-26, 1989).

Brady, K.B.C., J.R. Shaulis, and V.W. Skema. 1988. A study of mine drainage quality and prediction using overburden analysis an paleoenvironmental reconstructions, Fayette County, Pennsylvania. p. 33-43. *In* Proceedings: Mine Drainage and Surface Mine Reclamation, Vol. 1: Mine Water and Mine Waste. (Pittsburgh, PA, April 19-21, 1988). U.S. Bur. of Mines Inform. Circ. 9183.

Brady, K.B.C., M.W. Smith, R.L. Beam, and C.A. Cravotta III. 1990. Effectiveness of the addition of alkaline materials on surface coal mines in preventing or abating acid mine drainage: Part 2. Mine site case studies. *In* Proceedings: The 1990 Mining and Reclamation Conference. (Charleston, WV, April 23-26, 1990).

Brassington, R. 1988. *Field Hydrogeology*. The Geological Society of London Handbook Series, John Wiley & Sons, New York, p. 41-43.

Cravotta, C.A. III, K.B.C. Brady, M.W. Smith, and R.L. Beam. 1990. Effectiveness of the addition of alkaline materials at surface coal mines in preventing or abating acid mine drainage: Part 1. Geochemical considerations. *In* Proceedings: The 1990 Mining and Reclamation Conference. (Charleston, WV, April 23-26, 1990).

Davis, J.C. 1973. *Statistics and Data Analysis in Geology*. John Wiley & Sons, New York, p. 356-361.

diPretoro, R. and H. Rauch. 1987. Premining prediction of acid potential for surface mines in Northern West Virginia. p. 395-404. Symposium on Surf. Mining, Hydrology, Sedimentology, and Reclamation. (Lexington, KY, Dec., 1987).

diPretoro, R. and H. Rauch. 1988. Use of acid-base accounts in premining prediction of acid drainage potential. p. 2-10. *In* Proceedings: Mine Drainage and Surface Mine Reclamation, Vol. 1: Mine Water and Mine Waste. (Pittsburgh, PA, April 19-21, 1988). U.S. Bur. of Mines Inform. Circ. 9183.

Erickson, P.M., and R. Hedin. 1988. Evaluation of overburden analytical methods as means to predict post-mining coal mine drainage quality. p. 11-19. *In* Proceedings: Mine Drainage and Surface Mine Reclamation, Vol. 1: Mine Water and Mine Waste. (Pittsburgh, PA, April 19-21, 1988). U.S. Bur. of Mines Inform. Circ. 9183.

Geyer, A.R., and J.P. Wilshusen. 1982. Engineering characteristics of the rocks of Pennsylvania. Pennsylvania Geologic Survey, Environmental Geology Report EG-1, 2nd ed., 300 p.

Kania, T.C., R.W. Wood, R.L. Beam, T.P. Pongrac, and R. Naylor. 1989. The occurrence of acid mine drainage in the Johnstown area and the implications for the future exploitation of remaining coal reserves. p. 11-19. 54th Annual Field Conference of Pennsylvania Geologists Guidebook, Pennsylvania Geologic Survey.

Sobek, A.A., W.A. Schuller, J.R.Freeman, and R.M. Smith. 1978. Field and Laboratory Methods Applicable to Overburden and Minesoils. U.S. E.P.A. Report EPA-600/2-78-054.

Surface Mine Drainage Task Force. 1979. Suggested Guidelines for Method of Operation in Surface Mining of Areas With Potentially Acid Producing Materials. Green Lands 9:21-40.

Williams, E.G., A.W. Rose, R.R. Parizek, and S.A. Waters. 1982. Factors Controlling the Generation of Acid Mine Drainage. Final Report to the U.S. Bur of Mines, Research Grant No. G5105086, 256p.

This page intentionally left blank.

# EFFECTIVENESS OF THE ADDITION OF ALKALINE MATERIALS AT SURFACE COAL MINES IN PREVENTING OR ABATING ACID MINE DRAINAGE: PART 1. GEOCHEMICAL CONSIDERATIONS[1]

Charles A. Cravotta III, Keith B. C. Brady, Michael W. Smith, and Richard L. Beam[2]

Abstract. The addition of alkaline materials to supplement deficient "neutralization potential" (NP) of mine spoil, and thus to prevent or abate acid mine drainage, has not been successful at most surface coal mines in Pennsylvania. A basic problem may have been improper accounting for acid-production potential and thus inadequate addition rates of calcium carbonate ($CaCO_3$), calcium oxide (CaO), or calcium hydroxide [$Ca(OH)_2$] at many mines. The commonly used acid-base accounting method is based on the following overall reaction:

$$FeS_2 + 2\ CaCO_3 + 3.75\ O_2 + 1.5\ H_2O \Longrightarrow Fe(OH)_3 + 2\ SO_4^{-2} + 2\ Ca^{+2} + 2\ CO_2(g),$$

where the acidity from 1 mole of pyrite ($FeS_2$) is neutralized by 2 moles of $CaCO_3$. This method presumes that gaseous carbon dioxide ($CO_2$) will exsolve, and therefore may underestimate by up to a factor of 2 the quantity of $CaCO_3$ required to neutralize the "maximum potential acidity" (MPA) in the mine spoil. This paper reviews some geochemical reactions involving $FeS_2$ and various alkaline additives that support the argument that the acid-base accounting method for computing MPA from overburden analyses should be revised. Considering the stoichiometry of the following overall reaction:

$$FeS_2 + 4\ CaCO_3 + 3.75\ O_2 + 3.5\ H_2O \Longrightarrow Fe(OH)_3 + 2\ SO_4^{-2} + 4\ Ca^{+2} + 4\ HCO_3^{-},$$

4 moles of $CaCO_3$ are required to neutralize the maximum potential acidity produced by the oxidation of 1 mole of $FeS_2$. Therefore, the multiplication factor for computing MPA from the overburden sulfur concentration, in weight percent, should be increased from 31.25 to 62.5.

## Introduction

Acid mine drainage (AMD), in which total mineral acidity exceeds alkalinity, is a persistent problem associated with many surface coal mines. AMD typically contains large concentrations of sulfate, iron, and other metals, and results mainly from the exposure and accelerated oxidation of pyrite ($FeS_2$) and additional iron-sulfide or -sulfate minerals in the coal and overburden. However, where substantial calcium- or magnesium-carbonate materials, such as limestone strata, overlie the coal, mine drainage is commonly alkaline. By corollary, where mined strata contain pyrite but lack naturally occurring calcareous material, the importation and addition of alkaline material to the mine spoil should offset the deficiency and prevent or abate AMD. However, where alkaline additives have been incorporated with mine spoil at surface coal mines in Pennsylvania, few sites have shown improvement in water quality or abatement of AMD (Brady et al. 1990).

This paper describes the chemical reactions that are the basis for computing the maximum potential acidity and net neutralization potential of mine spoil. Emphasis is placed on evaluating reactions with calcite (calcium carbonate, $CaCO_3$), "quick lime" (calcium oxide, CaO), and "hydrated lime" [calcium hydroxide, $Ca(OH)_2$], which are used as alkaline additives intended to produce near-neutral ($6 < pH < 8$) discharge water from surface coal mines.

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]Charles A. Cravotta III is a hydrologist at U.S. Geological Survey, Harrisburg, PA 17108; Keith B. C. Brady, Michael W. Smith, and Richard L. Beam are hydrogeologists at Pennsylvania Department of Environmental Resources, Harrisburg, PA 17120.

## Acid-Base Accounting

Acid-base accounting (ABA), which simplifies the complex hydrogeochemical system through use of a limited number of variables, has been used extensively in the past to estimate the quantity of alkaline material required to neutralize the potential acidity of mine spoil (diPretoro 1986; diPretoro and Rauch 1988; Skousen et al. 1987; Erickson and Hedin 1988; Ferguson and Erickson 1988; Brady and Hornberger 1989; Smith and Brady 1990; Brady et al. 1990). ABA was developed on the assumption that the stoichiometry of the following overall reaction of $FeS_2$ and $CaCO_3$ applies (Sobek et al. 1978; Williams et al. 1982):

$$FeS_2 + 2\ CaCO_3 + 3.75\ O_2 + 1.5\ H_2O \Longrightarrow$$

$$Fe(OH)_3 + 2\ SO_4^{-2} + 2\ Ca^{+2} + 2\ CO_2(g). \quad (1)$$

The implication of reaction 1 is that acidity produced from 1 mole (mol) of $FeS_2$ [64 grams (g) of sulfur (S)] is neutralized by 2 mol of $CaCO_3$ (200 g), or 1 g S to 3.125 g $CaCO_3$. On this basis, 31.25 tons of calcium carbonate ($CaCO_3$) will neutralize the acidity from 1,000 tons of rock that contains 1.0 weight percent (%) pyritic sulfur. In accordance with accepted ABA methods (Sobek et al. 1978), the total sulfur concentration in weight percent is multiplied by 31.25 to obtain a "maximum potential acidity" (MPA), which has units of tons of $CaCO_3$ per 1,000 tons of overburden (tons $CaCO_3$/1,000 tons) and which assumes that the sulfur is pyritic and acid producing. The 31.25 multiplication factor was intended to provide equivalent units for direct comparison with "neutralization potential" (NP), which has units of tons $CaCO_3$/1,000 tons. Subsequent workers computed the "net neutralization potential" (NNP) for coal-bearing strata by subtracting volume- or weight-weighted MPA from NP (NNP = NP - MPA) (Erickson and Hedlin 1988; diPretoro and Rauch 1988; Brady et al. 1990). A negative, or deficient, NNP has been interpreted as the quantity of $CaCO_3$ that must be added to abate or prevent AMD. For example, if weight-weighted NP is 30 tons $CaCO_3$/1,000 tons and total sulfur concentration is 1.0%, then MPA = 31.25 tons $CaCO_3$/1,000 tons and NNP = -1.25 tons $CaCO_3$/1,000 tons. To create a net neutral mine spoil, 1.25 tons of $CaCO_3$ would need to be added to every 1,000 tons of overburden. However, the ABA method based on the stoichiometry of reaction 1 may underestimate MPA because of the presumption that $CO_2$ will exsolve, and thus may underestimate the $CaCO_3$ required to supplement deficient NNP.

## Previous Work

Although not originally intended for the purpose, ABA following the method of Sobek et al. (1978) has been used in attempts to predict post-mining water quality. However, several researchers have arrived independently at the conclusion that equal quantities of NP and MPA (computed by multiplying the total sulfur in percent by a factor of 31.25) do not prevent AMD. Brady and Hornberger (1989) identified a given stratum as potentially acid or alkaline producing by using threshold concentrations for total sulfur or NP, respectively, of 0.5 % or 30 tons $CaCO_3$/1,000 tons (and which reacted with dilute hydrochloric acid). These threshold concentrations were corrorabated by laboratory experiments by Williams et al. (1982) and Morrison (1988) and also by Pennsylvania Department of Environmental Resources data on overburden and water quality at numerous surface coal mines. Note that 0.5 percent total sulfur, when multiplied by 31.25 to compute MPA, equals 15.6 tons $CaCO_3$/1,000 tons, roughly half the guideline number of 30 tons $CaCO_3$/1,000 tons for NP. Skousen et al. (1987, p.4) suggested that a stratum which contains values greater than 5 tons $CaCO_3$/1,000 tons as "Max Needed" (negative NNP) produced acid; conversely, values greater than 20 tons $CaCO_3$/1,000 tons as "Excess" (positive NNP) produced alkaline drainage.

diPretoro (1986) and diPretoro and Rauch (1988) showed that NP and MPA were not equivalent in using the ratio derived by dividing the cumulative volume-weighted NP by MPA for composite strata. diPretoro and Rauch (1988) found that sites having a NP/MPA ratio of less than about 2.4 produced acidic drainage, whereas most sites having a ratio greater than 2.4 produced alkaline drainage. Ferguson and Erickson (1988) showed that mine sites with a multiple-strata average NNP of 30 tons $CaCO_3$/1,000 tons or greater always produced alkaline drainage. They also found that 59 percent of mine sites with NNP of 7 to 30 tons $CaCO_3$/1,000 tons produced alkaline drainage, and only 11 percent of the sites with NNP less than 7 tons $CaCO_3$/1,000 tons produced alkaline drainage. Weighted NP in "equivalent" amounts as MPA was not sufficient to prevent AMD.

## Geochemistry of Acid Mine Drainage and Alkaline Additives

The following discussion reviews some overall acid-forming and neutralizing reactions that are relevant to AMD, ABA, and the addition of alkaline materials at surface coal mines. No effort is made to account for hydrogeochemical variables such as surface- and ground-water flow paths, proximity and distribution of reacting minerals, solubilities and reaction rates of minerals, or the wide range of hydrochemical conditions in mine spoil.

## Production of Acidity

AMD results from the interactions of oxygen, water, bacteria, and sulfide minerals (Singer and Stumm 1970a, 1970b; Nordstrom et al. 1979; Kleinmann et al. 1980; Cathles 1982). Pyrite ($FeS_2$), and less commonly, marcasite ($FeS_2$) are the principal sulfur-bearing minerals in bituminous coal (Davis 1981; Hawkins 1984), and because of its wide distribution, pyrite is recognized as the major source of AMD in the eastern United States (Stumm and Morgan 1981, p. 469-471). The following overall stoichiometric reactions may characterize the oxidation of pyrite and other $FeS_2$ minerals:

$$FeS_2 + 3.5\ O_2 + H_2O \Longrightarrow Fe^{+2} + 2\ SO_4^{-2} + 2\ H^+, \quad (2)$$

$$Fe^{+2} + 0.25\ O_2 + H^+ \Longrightarrow Fe^{+3} + 0.5\ H_2O, \quad (3)$$

$$Fe^{+3} + 3\ H_2O \Longrightarrow Fe(OH)_3 + 3\ H^+. \quad (4)$$

The oxidation of sulfide in pyrite to sulfate (reaction 2) releases dissolved ferrous iron ($Fe^{+2}$) and "acidity" ($H^+$) into the water. Subsequently, ferrous iron is oxidized to ferric iron ($Fe^{+3}$) (reaction 3), which if separated from the pyrite surface, hydrolyzes and forms insoluble ferrihydrite [$Fe(OH)_3$)] (reaction 4), and releases more acidity. The overall combination of reactions 2 through 4 may be written as follows:

$$FeS_2 + 3.75\ O_2 + 3.5\ H_2O \Longrightarrow$$

$$Fe(OH)_3 + 2\ SO_4^{-2} + 4\ H^+. \qquad (5)$$

In reaction 5, 3.75 mol of oxygen are consumed to oxidize 1 mol of pyrite, and 2 mol of sulfate, 4 mol of acidity, and 1 mol of ferrihydrite are produced.

**Neutralization of Acidity**

Acidity produced by the aqueous oxidation of pyrite may react with carbonate, silicate, and hydroxide minerals composing the sedimentary rocks in the coal-bearing sequence. Dissolution of these acid-soluble minerals neutralizes acidity and produces the other major ions in AMD in addition to sulfate and iron, such as manganese, aluminum, calcium, magnesium, sodium, potassium, and silica.

The most acid-reactive minerals are the carbonates: calcite ($CaCO_3$), dolomite [$CaMg(CO_3)_2$], and siderite ($FeCO_3$). Carbonates are present in variable quantities as individual mineral grains and as cementing agents in limestone, dolostone, sandstone, and shale. Limestone and dolostone are composed predominantly of calcite and dolomite, respectively; shale and sandstone are composed predominantly of silicate minerals, but may contain some carbonate as cement or matrix. Dissolution of calcite, dolomite, and other calcium- or magnesium-bearing carbonate minerals tends to reduce acidity, increase alkalinity, and raise pH; however, dissolution of siderite and the subsequent hydrolysis of iron may increase acidity, and reduce pH. The dissolution of calcite by the following sequence of reactions serves as an example:

$$CaCO_3 + 2\ H^+ \Longrightarrow Ca^{2+} + H_2CO_3^*, \qquad (6)$$

$$CaCO_3 + H_2CO_3^* \Longrightarrow Ca^{2+} + 2\ HCO3^-. \qquad (7)$$

Reaction 6 represents acidic conditions (pH < 6.4) where calcite is not abundant and is totally dissolved by reacting 1 mol of calcite and 2 mol of free acidity and producing 1 mol of dissolved calcium and 1 mol of dissolved carbon dioxide {$[H_2CO_3^*] = [CO_2(aq) + [H_2CO_3^o]$}, which is a weak acid (Krauskopf 1979, p. 40-42; Stumm and Morgan 1981, p. 171-214; Hem 1985, p. 92, 105-111). Note that gaseous $CO_2$ is not indicated as the product in reaction 6. In practice, during laboratory determination of overburden NP, $CO_2$ may be exsolved when calcite and other carbonate minerals are reacted with dilute acid. However, $CO_2$ may concentrate in both the gaseous and aqueous phases in mine spoil. Elevated partial pressure of $CO_2$ in the unsaturated zone of mine spoil is common (Lusardi and Erickson 1985), especially during the growing season, and will cause a concomitant increase in the concentration of $H_2CO_3^*$ and other aqueous carbon-dioxide species (Langmuir 1971; Shuster and White 1972; Harmon et al. 1975).

If calcite is abundant, the dissolved carbon dioxide will continue to react with calcite (reaction 7) producing bicarbonate ions and raising pH. The overall combination of reactions 6 and 7 represents the condition where dissolution of calcite produces "alkalinity" in excess of "acidity" and raises the pH above 6.4 where bicarbonate ($HCO_3^-$) is the dominant dissolved carbon-dioxide species:

$$CaCO_3 + H^+ \Longrightarrow Ca^{2+} + HCO3^-. \qquad (8)$$

Reaction 8 shows that 1 mol of calcite will neutralize 1 mol of free acidity and produce 1 mol each of dissolved calcium and bicarbonate. Reactions 5 and 8 may be combined to indicate a "maximum" neutralization potential of calcite, where no $CO_2$ is exsolved:

$$FeS_2 + 4\ CaCO_3 + 3.75\ O_2 + 3.5\ H_2O \Longrightarrow$$

$$Fe(OH)_3 + 2\ SO_4^{-2} + 4\ Ca^{+2} + 4\ HCO3^-. \qquad (9)$$

Reaction 9 shows that the acidity produced from the oxidation of 1 mol of $FeS_2$ (64 g S) may be neutralized by 4 mol of $CaCO_3$ (400 g), which is a mass ratio of 6.25 g of calcite to 1 g of pyritic sulfur. The calcite-to-sulfur mass ratio of 6.25 is twice the ratio of 3.12 which would be derived considering the unlimited exsolution of $CO_2$ (Williams et al. 1982).

On the basis of the calcite-to-sulfur mass ratio of 6.25, a multiplication factor of 62.5 to compute MPA from total sulfur is appropriate for ABA if all sulfur is from pyrite. Therefore, considering the earlier example for overburden, where NP = 30 tons $CaCO_3$/1,000 tons and total sulfur = 1 % (only now using the 62.5 factor), then MPA = 62.5 tons $CaCO_3$/1,000 tons and NNP = -32.5 tons $CaCO_3$/1,000 tons. Instead of the previously computed 1.25 tons, now 32.5 tons of $CaCO_3$ per 1,000 tons overburden would be required to supplement the deficient NNP.

"Quick lime" (calcium oxide, CaO) and "hydrated lime" [calcium hydroxide, $Ca(OH)_2$] (Rochow 1977, p. 129), which compose lime-kiln flue dust, have twice the neutralization potential as calcite. Because the lime compounds have lower unit mass than $CaCO_3$, they are required in equivalent ratios less than 3.12 according to the following reactions:

$$Ca(OH)_2 + 2\ H^+ \Longrightarrow Ca^{+2} + 2\ H_2O, \qquad (10)$$

$$CaO + 2\ H^+ \Longrightarrow Ca^{+2} + H_2O. \qquad (11)$$

Reactions 10 and 11 show that 1 mol of hydrated lime (74 g) or 1 mol of quick lime (56 g) may neutralize 2 mol of free acidity. Combining reactions 10 and 5:

$$FeS_2 + 2\ Ca(OH)_2 + 3.75\ O_2 \Longrightarrow$$

$$Fe(OH)_3 + 2\ SO_4^{-2} + 2\ Ca^{+2} + 0.5\ H_2O. \qquad (12)$$

Reaction 12 shows that the acidity produced from the oxidation of 1 mol of pyrite (64 g S) may be neutralized by 2 mol of calcium hydroxide (148 g), which is a mass ratio of 2.31 g of calcium hydroxide to 1 g of pyritic sulfur. Analogously, from combining reactions 5 and 11, a mass ratio of 1.75 g of calcium oxide to 1 g of pyritic sulfur is required to attain neutralization. Thus on a weight basis, 1 ton of $Ca(OH)_2$ has the neutralization equivalent of 2.7 tons of $CaCO_3$.

Siderite ($FeCO_3$) is common in coal-bearing strata and is frequently cited as having no net effect on acid-production or neutralization where $CO_2$ gas is exsolved (Stumm and Morgan, 1981; Williams et al. 1982). However, considering the argument for conditions with elevated partial pressure of $CO_2$, oxidation of siderite may produce net acidity in the form of dissolved carbon dioxide:

$$FeCO_3 + 0.25\ O_2 + 2.5\ H_2O \Longrightarrow Fe(OH)_3 + H_2CO_3^*. \qquad (13)$$

The $H_2CO_3^*$ generated in reaction 13 may react with additional carbonate, silicate, or hydroxide minerals. The effect of siderite as a potential acid-forming mineral is apparent by combining reactions 7 and 13 as follows:

$$FeCO_3 + CaCO_3 + 0.25\ O_2 + 2.5\ H_2O \Longrightarrow$$

$$Fe(OH)_3 + Ca^{+2} + 2\ HCO3^-. \qquad (14)$$

In reaction 14, the acidity produced from 1 mol of siderite is neutralized by 1 mol of calcite. Thus, if siderite is present, additional alkaline material beyond that required to neutralize the acidity from pyrite may be necessary. Impurities such as Mn, Mg, and to a lesser extent Ca, may substitute for Fe in siderite (Morrison et al. 1990). The Fe and Mn may hydrolyze and produce acid; however, the Mg and Ca may have neutralizing ability similar to dolomite and calcite.

## Discussion

The presentation of acid-forming and acid-neutralizing reactions was simplified by writing and then combining independent equations as "neutral overall" reactions that eliminated $H^+$ as a reactant or product. Thus, the overall stoichiometries in reactions 1 and 9 equate quantities of acid-producing and -neutralizing materials and are useful for acid-base accounting application. However, reactions 1 and 9 are "end-member" reactions; the hydrogeochemical relations in mine-spoil ground water or discharge probably lie soemwhere between the two end members because some $CO_2$ will exsolve and some will dissolve.

No attempt has been made in the above review to discuss the combined effects of variable purities, degrees of crystallinity, and particle sizes of minerals; microbiological catalysis of reactions; or relative reaction rates. For example, the presumption that 4 mol of $CaCO_3$ are required to neutralize the acidity from 1 mol of $FeS_2$ (reaction 9) implies that the production of acidity is rate limiting, or slow relative to neutralization, and that neutralization is instantaneous. Furthermore, the computation of maximum potential acidity (MPA) as 62.5 times the total sulfur concentration, in weight percent, should yield a conservative estimate, because not all $CO_2$ will dissolve nor will all sulfur be pyritic and acid producing. To determine quantities of alkaline additives required at surface coal mines, site-specific characteristics such as mining method, pre- and post-mining overburden composition, post-mining reclamation and hydrogeology, and alkaline additives used and placement technique also must be evaluated. The companion paper by Brady et al. (1990) reviews some of the site-specific factors and compares post-mining water quality and ABA computations of MPA using the conventional and newly proposed multiplication factors of 31.25 and 62.5, respectively, for selected surface coal mines in Pennsylvania.

## Summary and Conclusions

In summary, the ABA method currently in use, which presumes 2 mol of $H^+$ may be neutralized by 1 mol of $CaCO_3$, may underestimate by up to a factor of 2 the $CaCO_3$ required to neutralize the maximum potential acidity from the oxidation of pyrite and the hydrolysis and precipitation of iron, because of the presumption that all $CO_2$ will exsolve. However, some $CO_2$ will dissolve forming a weak acid that reacts with carbonate minerals. Assuming no exsolution of $CO_2$, 1 mol of $FeS_2$ will produce 4 mol of $H^+$, which may be neutralized by 4 mol of $CaCO_3$ (reaction 9), 2 mol of $Ca(OH)_2$ (reaction 12), or 2 mol of CaO. On a weight basis then, 1 g of pyritic sulfur may be neutralized by 6.25 g of $CaCO_3$, 2.31 g of $Ca(OH)_2$, or 1.75 g of CaO. Considering these equivalent weights, MPA as tons of $CaCO_3$ deficiency per 1,000 tons of overburden should be computed by multiplying total sulfur, in weight percent, by 62.5. The above discussion is based only on the stoichiometry of the overall reactions (9 and 12) and assumes that the rate of acid production will not exceed the rate of acid neutralization. The actual acidity may be less than the computed MPA because not all $CO_2$ dissolves and not all sulfur generates acidity. Finally, dissolution of siderite will produce net acidity when the partial pressure of $CO_2$ becomes elevated and the iron is hydrolyzed and precipitated.

In conclusion, for conservative estimates of overburden net neutralization potential (NNP), a revised multiplication factor of 62.5 should be used to compute maximum potential acidity (MPA) from the total sulfur concentration, in weight percent.

## Acknowledgments

The authors wish to thank Arthur W. Rose, Jay W. Hawkins, Patricia M. Erickson, Dale W. Blevins, and Andrew C. Ziegler for helpful comments during preparation of this manuscript.

## Literature Cited

Brady, K.B.C., and R.J. Hornberger. 1989. Mine drainage prediction and overburden analysis in Pennsylvania. In Proceedings: West Virginia Surface Mine Drainage Task Force Symposium. (Morgantown, WV, April 25-26, 1989).

Brady, K.B.C., M.W. Smith, R.L. Beam, and C.A. Cravotta III. 1990. Effectiveness of the addition of alkaline materials at surface coal mines in preventing or abating acid mine drainage—Part II. Mine site case studies. In Proceedings of the 1990 Mining and Reclamation Conference and Exhibition. (Charleston, WV, April 23-26, 1990).

Cathles, L.M. 1982. Acid mine drainage. The Pennsylvania State University Earth and Mineral Sciences 51(4):37-41.

Davis, Alan. 1981. Sulfur in coal. The Pennsylvania State University Earth and Mineral Sciences 51(2):13-21.

diPretoro, R.S. 1986. Premining prediction of acid drainage potential for surface coal mines in northern West Virginia. M.S. thesis. West Virginia University. 217 p.

diPretoro, R.S., and H.W. Rauch. 1988. Use of acid-base accounts in premining prediction of acid drainage potential. p. 2-10. In Proceedings: Mine Drainage and Surface Mine Reclamation, Vol. 1. Mine Water and Mine Waste (Pittsburgh, PA, April 19-21, 1988) U.S. Bureau of Mines Information Circular 9183.

Erickson, P.M., and R. Hedin. 1988. Evaluation of overburden analytical methods as means to predict post-mining coal mine drainage quality. p. 11-19. In Proceedings: Mine Drainage and Surface Mine Reclamation, Vol. 1. Mine Water and Mine Waste (Pittsburgh, PA, April 19-21, 1988) U.S. Bureau of Mines Information Circular 9183.

Ferguson, Keith, and P.M. Erickson. 1988. Approaching the AMD problem--from prediction and early detection. International Conference on Control of Environmental Problems from Metals Mines (Roros, Norway, June 20-24, 1988).

Harmon, R.S., W.B. White, J.J. Drake, and J.W. Hess. 1975. Regional hydrochemistry of North American carbonate terrains: Water Resources Research 11(6):963-967.

Hawkins, J.W. 1984. Iron disulfide characteristics of the Waynesburg, Redstone, and Pittsburgh coals in West Virginia and Pennsylvania. M.S. thesis. West Virginia University. 195 p.

Hem, J.D. 1985. Study and interpretation of the chemical characterisics of natural water (3d). U.S. Geological Survey Water-Supply Paper 2254, 263 p.

Kleinmann, R.L.P., D.A. Crerar, and R.R. Pacelli. 1980. Biogeochemistry of acid mine drainage and a method to control acid formation. Mining Engineering 33:300-306.

Krauskopf, K.B. 1979. Introduction to geochemistry. New York, McGraw-Hill, 617 p.

Langmuir, Donald. 1971. The geochemistry of some carbonate groundwaters in central Pennsylvania. Geochimica et Cosmochimica Acta 35:1023-1045.

Lusardi, P.J., and P.M. Erickson. 1985. Assessment and reclamation of an abandoned acid-producing strip mine in northern Clarion County, Pennsylvania. p. 313-321. In Proceedings: 1985 Symposium on Surface Mining Hydrology, Sedimentology, and Reclamation. (Lexington, KY, University of Kentucky).

Morrison, J.L. 1988. A study of factors controlling the severity of acid mine drainage in the Allegheny Group of western Pennsylvania. M.S. thesis. Pennsylvania State University. 145 p.

Morrison, J.L., S.D. Atkinson, Alan Davis, and B.E. Scheetz. 1990. The use of $CO_2$ coulometry in differentiating and quantifying the carbonate phases in the coal-bearing strata of western Pennsylvania--its applicability in interpreting and modifying neutralization potential (NP) measurements. In Proceedings of the 1990 Mining and Reclamation Conference and Exhibition (Charleston, WV, April 23-26, 1990).

Nordstrom, D.K., E.A. Jenne, and J.W. Ball. 1979. Redox equilibria of iron in acid mine waters. p. 51-79. In Jenne, E.A., ed., Chemical modeling in aqueous systems--Speciation, sorption, solubility, and kinetics. American Chemical Society Symposium Series 93.

Rochow, E.G. 1977. Modern descriptive chemistry. Philadelphia, W. B. Saunders Co., 253 p.

Shuster, E.T., and W.B. White. 1972. Source areas and climatic effects in carbonate groundwaters determined by saturation indices and carbon dioxide pressures. Water Resources Research 8(4):1067-1073.

Singer, P.C., and Werner Stumm. 1970a. Acidic mine drainage--the rate-determining step. Science 167:1121-1123.

Singer, P.C., and Werner Stumm. 1970b. Oxygenation of ferrous iron. U.S. Department of the Interior, Federal Water Quality Administration (U.S. Environmental Protection Agency) Water Pollution Control Research Report 14010--067/69.

Skousen, J.G., J.C. Sencindiver, and R.M. Smith. 1987. A review of proceedures for surface mining and reclamationin areas with acid-producing materials. West Virginia Surface Mine Drainage Task Force, 39 p.

Smith, M. W., and K.B.C. Brady. 1990. Review and summary of acid base accounting data using computer spreadsheets. In Proceedings of the 1990 Mining and Reclamation Conference and Exhibition (Charleston, WV, April 23-26, 1990).

Sobek, A.A., W.A. Schuller, J.R. Freeman, and R.M. Smith. 1978. Field and laboratory methods applicable to overburdens and minesoils. Cincinnatti, Ohio, U.S. Environmental Protection Agency Environmental Protection Technology EPA-600/2-78-054, 203 p.

Stumm, Werner, and J.J. Morgan. 1981. Aquatic chemistry--an introduction emphasizing chemical equilibria in natural waters (2nd). New York, John Wiley and Sons, 780 p.

Williams, E.G., A.W. Rose, R.R. Parizek, and S.A. Waters. 1982. Factors controlling the generation of acid mine drainage. Final report to the U. S. Bureau of Mines, Research Grant No. G5105086, 256 p.

This page intentionally left blank.

# EFFECTIVENESS OF THE ADDITION OF ALKALINE MATERIALS AT SURFACE COAL MINES IN PREVENTING OR ABATING ACID MINE DRAINAGE: PART 2. MINE SITE CASE STUDIES[1]

Keith B. C. Brady, Michael W. Smith, Richard L. Beam, and Charles A. Cravotta III[2]

Abstract. The effectiveness of preventing or ameliorating acid mine drainage (AMD) through the application of alkaline additives is evaluated for eight surface coal mines in Pennsylvania. Many of the mine sites had overburden characteristics that made prediction of post-mining water quality uncertain. Alkaline materials were applied at rates ranging from 42 to greater than 1,000 tons as calcium carbonate per acre. In addition, two sites that were mined and reclaimed without alkaline additives are included for comparative purposes.

Overburden sulfur concentration and "neutralization potential" (NP) data for multiple strata at each mine site were used to compute the cumulative, mass-weighted "maximum potential acidity" (MPA) and "net neutralization potential" (NNP = NP - MPA) by using three different calculation methods. Post-reclamation water-quality data were used to compute the net alkalinity (= alkalinity - acidity). The most conservative determination of NNP, whereby MPA is calculated by multiplying the total sulfur concentration, in weight percent, by 62.5 instead of 31.25, yielded the best agreement with net alkalinity (matching signs on NNP and net alkalinity). The error in prediction using each method was that the reclaimed overburden was computed to be alkaline overall (NNP > 0), but the post-reclamation water was acid (net alkalinity < 0).

In general, alkaline addition rates were probably insufficient to neutralize, or too late to prevent, acid production in the mine spoil. At six of the seven mine sites that had overburden with insufficient NP relative to MPA (NNP < 0), the addition of alkaline materials failed to create alkaline mine drainage; AMD was formed or persisted. A control site which also had insufficient alkaline material, but did not incorporate alkaline additives, generated severe AMD. Two sites that had substantial, natural alkaline overburden produced alkaline drainage. Although the addition rates appear to be inadequate, other factors, such as unequal distribution and exposure of the acid-forming or neutralizing materials and hydrogeological variability, complicate the evaluation of relative effectiveness of using different alkaline materials and placement of the acid- or alkaline-producing materials.

## Introduction

Acid mine drainage (AMD) from surface coal mines is a severe problem in Pennsylvania as well as other Appalachian coal mining states. Discharges of water from reclaimed mine sites must meet acceptable effluent limits; treatment of the discharges can be a major financial burden to a coal mine operator.

In Pennsylvania, the strata at some mine sites cannot be mined without causing AMD pollution because insufficient quantities of naturally alkaline material are present to neutralize the AMD. At coal mines with abundant naturally alkaline strata, mine drainage is commonly alkaline. Therefore, the importation and addition of alkaline

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]Keith B. C. Brady, Michael W. Smith, and Richard L. Beam are hydrogeologists at Pennsylvania Department of Environmental Resources, Harrisburg, PA 17120; Charles A. Cravotta III is a hydrologist at U.S. Geological Survey, Harrisburg, PA 17108.

material to alkaline-deficient mine spoil may produce sufficient neutralization potential to prevent or abate AMD. However, the majority of mine sites that have been reclaimed using imported alkaline materials produce AMD. One problem is knowing the quantity of alkaline material that must be added to produce the desired effect. Another problem is knowing the best place to add alkaline material within a mine site.

This paper summarizes the history of alkaline addition as a reclamation technique, and presents the results of a study of overburden and water-quality data at 10 reclaimed surface coal mines in the bituminous coal fields of Pennsylvania (fig. 1). Emphasis is placed on evaluating the recommended alkaline-addition rates as compared to the calcium-carbonate deficiency calculated by acid-base accounting (Sobek et al. 1978; Smith and Brady 1990; Cravotta et al. 1990). Factors such as the mine hydrogeology, operational history, mining method, placement and type of imported alkaline material, and selective handling of strata are evaluated. The study sites include mines where alkaline addition was used in an attempt to abate existing AMD problems, as well as mines where alkaline addition was used in an attempt to prevent AMD in areas not previously mined.

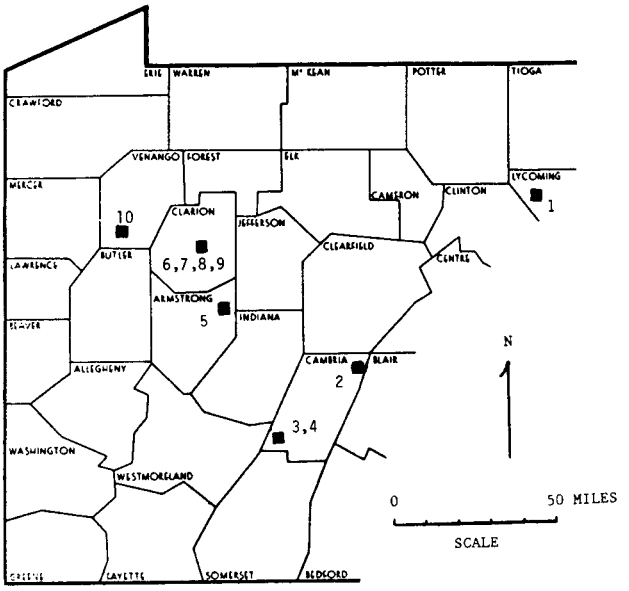



Figure 1. Map showing the locations of mine sites 1 through 10 in western Pennsylvania.

## Historical Background on Alkaline Addition in Pennsylvania

The Pennsylvania Department of Environmental Resources (PaDER) is responsible for the review and approval of permits for surface coal mines in Pennsylvania. The PaDER developed procedures for review of plans for alkaline addition on the basis of two reports. The first report is, "Suggested Guidelines for Method of Operation in Surface Mining of Areas with Potentially Acid-Producing Materials," by the West Virginia Surface Mine Drainage Task Force (1979). In "Appendix C" entitled "Immediate Lime Requirement" was the suggestion that, "A realistic lime requirement figure is probably a third of the maximum potential acidity from total sulfur." The second report is, "The Application of Limestone and Lime Dust in the Abatement of Acidic Drainage in Centre County, Pennsylvania" by Waddell et al. (1980). Rock, stratigraphically lower than the coal-bearing units, was excavated and redeposited during construction of Interstate 80. Acidic discharges flowed from the toe of this spoil. A mixture of limestone and lime-kiln flue dust was added to part of the spoil at the rate of 267 tons per acre (tons/acre). Although improvements in water quality were observed within the treated and untreated areas, the improvements in the treated areas were thought to be the most significant.

These early suggestions of success in correcting AMD encouraged the PaDER to permit alkaline addition at surface coal mines where the strata, according to acid-base accounting (ABA), showed only a slight deficiency of calcareous material. ABA considers two variables--neutralization potential (NP), in tons of calcium carbonate per 1,000 tons of overburden (tons $CaCO_3$/1,000 tons), and total sulfur, in weight percent (%), which is converted to "maximum potential acidity" (MPA) reported as tons $CaCO_3$/1,000 tons. A detailed discussion of the chemical stoichiometric relations that are assumed in ABA is given in Cravotta et al. (1990). The net neutralization potential (NNP) of mine spoil is computed by subtracting mass-weighted MPA from NP. A negative, or deficient, NNP has been interpreted as the quantity of $CaCO_3$ that must be added to prevent or abate AMD. Alkaline addition rates were generally calculated at one-third the NNP as suggested by the West Virginia Surface Mine Drainage Task Force (1979), although sometimes the addition rate was increased slightly as a "safety factor." As this study clearly shows, most reclamation using alkaline addition at surface coal mines in Pennsylvania has failed to prevent or abate AMD.

Previous studies focused on alkaline addition as a means of abating existing AMD problems (Geidel 1982; Lusardi and Erickson 1985; Caruccio and Geidel 1984, 1986; Waddell et al. 1980) and not as part of the ongoing mining operation. In general, two alkaline-treatment schemes were advanced: (1) Waddell et al. (1980) hypothesized that it was unnecessary to neutralize all the potential acidity in pyritic spoil because the addition of alkaline materials to create pH greater than 4.5 would inhibit the bacterial catalysis of pyrite oxidation. (2) Lusardi and Erickson (1985) assumed that most acid is produced near the surface and that it was only necessary to add sufficient limestone to balance the net deficiency in the upper spoil zone. However, these attempts to abate AMD by adding alkaline material to pyritic spoil resulted in limited, if any, success because effluent limits were seldom met or maintained.

The advent of alkaline addition and selective handling, as well as the poor success of literal interpretation of ABA, necessitated the development of guidelines for understanding which strata were potentially alkaline or acid producing. On the basis of PaDER's experience it was concluded that a NP of 30 tons $CaCO_3$/1,000 tons (and "fizz", effervescence during reaction with dilute hydrochloric acid) and a total sulfur content of 0.5% were

reasonable guidelines to be used in defining strata that were potentially alkaline or acid producing (Brady and Hornberger 1989). Addition rates were generally calculated on the basis of total sulfur values greater than 0.5%; the permittee was given "credit" for strata with NP's greater than 30 tons $CaCO_3$/1,000 tons.

## Methods

The method of selection of mine sites for this study was one of gathering data and then eliminating sites for which necessary information was lacking. The data required for site selection included ABA overburden data and post-mining discharge or ground-water-quality data (pH and concentrations of alkalinity, acidity, iron, manganese, and sulfate). Overburden samples must have been collected from drill holes no farther than a few hundred feet from the area mined. Furthermore, the mining methods and plans must have been documented and the alkaline addition must have been performed as specified in the permit. All sites that met the selection criteria were included.

Overburden analysis data for total sulfur and NP at sites 1 through 8 and 10 (fig. 1) were obtained from PaDER permit files. Data for mine site 9 were obtained from U.S. Geological Survey (USGS) project files. Cumulative mass-weighted NNP was calculated by using a computer-spreadsheet program (Smith and Brady 1990) that included multiplication factors of 31.25 and 62.5 to compute MPA from the total sulfur data (Cravotta et al. 1990).

Water-quality data were obtained from coal-company files and PaDER permit or USGS project files; PaDER and USGS samples were analyzed by the PaDER laboratory. Water samples were collected from toe-of-spoil seeps, monitor wells, and (or) deep mine discharges that were downflow from the mine site. Net alkalinity [in milligrams per liter as calcium carbonate (mg/L $CaCO_3$)] was calculated by subtracting acidity (base-neutralizing capacity) from alkalinity (acid-neutralizing capacity) (Stumm and Morgan 1981, p. 163-166). Net alkalinity was used in previous AMD studies (diPretoro 1986; diPretoro and Rauch 1988; Erickson and Hedin 1988) because it reflects the regulatory requirement that alkalinity exceed acidity, and it allows comparison of a single water-quality parameter with overburden NNP.

Water-quality data were evaluated using "notched" boxplots (Velleman and Hoaglin 1981; Helsel 1987) (fig. 2). The box is defined by the interquartile range (IQR = 25th to 75th percentiles). The median (50th percentile) is shown as a "+" within the box. Notches "()" identify the 95-percent confidence interval around the median (Hettmansperger and Sheather 1986) and are useful in testing the difference between median values for data subsets. For example, a pair of boxplots in figure 2 shows that the medians for two subsets of data are significantly different at the 95-percent confidence level. The notches enclosing the median do not overlap--the right-side notch of the upper box is not greater than the left-side notch of the lower box. However, if the notches for a pair of boxplots overlap, then the medians for the two subsets are not significantly different at the 95-percent confidence level.

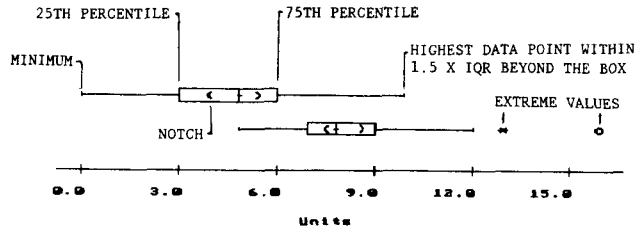



Figure 2. Explanation of boxplots showing median "+" (50th percentile), interquartile range (IQR = 25th to 75th percentile), notches "( )", and extreme values "*" and "o". Horizontal lines are drawn to the data points farthest from the box, yet within a distance 1.5 times the IQR beyond the box. Extreme values beyond this distance are plotted individually.

The mining history for individual case studies was determined from permit-file inspection reports, discussions with coal-company personnel and PaDER mine inspectors, and field investigations by the authors. The historical information gathered includes the area mined, mining method and equipment used, compliance record, hydrologic conditions during mining, speed of the operation, and size of the active mine.

## Case Studies

Eight surface-coal mines where alkaline addition was permitted by the PaDER met the selection criteria for inclusion in this study. Two additional mines, where alkaline materials were not added, are also included: Mine site 3 is a control for comparison with site 4, and site 10 is an example with abundant naturally alkaline strata. For comparison of similarities and differences of the ten mine sites selected for study, table 1 lists site characteristics, and table 2 lists the NNP of the overburden and net alkalinity of post-mining water samples. Each mine site is unique in terms of mining methods, size of the mine, hydrogeology, stratigraphic interval, overburden quality, pre- and post-mining water quality, monitoring programs, and so forth. With the exception of mine sites 3 and 9, the mines were developed after 1980, and modern mining and reclamation practices were used. Overburden NP and total sulfur data were available for all sites except site 2, for which only total sulfur data were available. Pre- and post-mining water-quality information was available for all sites except mine site 10. References to MPA and NNP are based on the revised chemical stoichiometry of Cravotta et al. (1990). The following discussions are provided to address the mine and overburden characteristics, methods and materials used for alkaline additives, and related water-quality data.

**Mine Site 1: Lycoming County.** Mine site 1 is situated on a relatively flat, isolated hilltop. Approximately 210 acres of the lower Kittanning (Bloss) coal were mined, along with lesser acreages of the overlying middle Kittanning, upper Kittanning, and lower Freeport coals. Maximum highwall height was about 135 ft. The mining area includes an abandoned deep-mine complex of approximately 70 acres in the lower Kittanning coal. It also

Table 1: Mine Site Characteristics

| MINE SITE | COUNTY | AREA (ACRES) | NO. OF OVERBURDEN HOLES | -------Tons as CaCO3 per acre------- ALKALINE ADDITION TOTAL | PIT FLOOR | SPOIL | SPOIL SURFACE | QUALITATIVE OVERBURDEN COMPOSITION** | SELECTIVE OVERBURDEN HANDLING |
|---|---|---|---|---|---|---|---|---|---|
| 1 | Lycoming | 45 | 3 | 500 | 50 | 350 | 100 | Low NP/Low S | NO |
| 2 | Cambria | 26 | 1 | 42 | 22 | 10 | 10 | Low NP/Low S | YES |
| 3 | Cambria | 47 | 2 | 0 | 0 | 0 | 0 | Low NP/Med S | NO |
| *4 | Cambria | 75 | 2 | 648 | 54 | 0 | 594 | Low NP/Med S | YES |
| 5 | Armstrong | 19 | 1 | 140 | 0 | 0 | 140 | Med NP/Med S | NO |
| 6A | Clarion | 54 | 2 | 100 | 20 | 0 | 80 | Low NP/Low S | NO |
| 7 | Clarion | 60 | 2 | 300 | 20 | 0 | 280 | Low NP/Low S | NO |
| *8 | Clarion | 29 | 2 | 54 | 0 | 0 | 54 | Low S/Low NP | YES |
| *9A | Clarion | 2.5 | 2 | 1.120 | 0 | 0 | 1,120 | Low NP/High S | NO |
| *9B | Clarion | 2.5 | 2 | 724 | 0 | 0 | 724 | Low NP/High S | NO |
| *9C | Clarion | 2.5 | 2 | 6 | 0 | 0 | 6 | Low NP/High S | NO |
| 10 | Venango | 41.5 | 2 | 0 | 0 | 0 | 0 | High NP/Med S | YES |

*Alkaline material includes hydrated lime, $Ca(OH)_2$, which is reported as equivalent tons $CaCO_3$ (see Cravotta et al., 1990).

**NP = neutralization potential, in tons $CaCO_3$ /1000 tons; S = total sulfur concentration, in weight percent.

**Table 2:** Comparison of net neutralization potential of overburden [1], [2] and post-treatment net alkalinity[3] of coal-mine drainage

| Mine Site[5] | NNP Before Alkaline Addition (no thresholds) | | NNP After Alkaline Addition MPA = 31.25 x % S | | NNP After Alkaline Addition MPA = 62.5 x % S | | Post-treatment Median Net Alkalinity of Coal-Mine Drainage |
|---|---|---|---|---|---|---|---|
| | MPA = 31.25 x %S | MPA = 62.5 x %S | (no threshold) | (thresholds)[4] | (no threshold) | (thresholds)[4] | |
| 1 | +6.03 | +1.51 | +6.90 | -0.31 | +2.31 | -1.43 | +67 |
| 2 | -3.92 | -7.84 | -3.06 | -1.73 | -6.98 | -4.32 | -62 |
| 3 | +2.85 | -4.97 | +2.85 | -3.76 | -4.97 | -7.71 | -468 |
| 4 | +2.77 | -10.81 | +4.91 | -6.07 | -7.97 | -13.03 | -74 |
| 5 | +15.69 | +3.91 | +16.68 | +14.65 | +4.91 | +7.06 | +11 |
| 6 A | +4.10 | -3.82 | +4.36 | +2.88 | -3.56 | -1.71 | -55 |
| 7 | -3.88 | -11.67 | -1.14 | -1.03 | -9.43 | -4.30 | -685 |
| 8 | +5.55 | +1.90 | +5.76 | +3.17 | +2.32 | +1.88 | -184 |
| 9 A | -3.40 | -7.92 | -0.81 | +0.61 | -2.91 | +1.04 | -2190 |
| B | -6.16 | -13.23 | -3.65 | -2.20 | -9.99 | -6.19 | -860 |
| C | -9.68 | -20.74 | -9.66 | -6.83 | -20.70 | -13.65 | -3332 |
| 10 | +170.47 | +156.81 | +170.47 | +168.77 | +156.81 | +156.6 | +118 |

1 "Net neutralization potential" (NNP = NP-MPA) reported in tons $CaCO_3$/1,000 tons (Smith and Brady 1990); "maximum potential acidity" (MPA) computed by multiplying total sulfur (S), in weight percent by 31.25 and 62.5, respectively (Cravotta et al. 1990).
2 An appendix with the actual overburden analysis data is available from the authors.
3 Net alkalinity (= alkalinity - acidity) in mg/L as $CaCO_3$.
4 Threshold where total S<0.5%, calculated MPA=O; where NP<30 or no effervesence reaction with dilute HCl, calculated NP=0.
5 Mine sites 3 and 10 did not have alkaline additives incorporated with the mine spoil. These sites are included for comparative purposes.

includes approximately 34 acres where the lower Freeport coal was previously surface mined; drainage was diverted from the surface mined lower Freeport to the deep mine via an underdrain system. Mining has taken place continuously since the early 1970s and continues to date. Underground mine workings were first encountered in late 1984. Approximately 45 acres have been daylighted using a dragline.

Most ground water from the site discharges from the abandoned underground mine via an air shaft. Water quality was monitored at this discharge point since 1979, and flow rates were monitored continuously since 1982. Prior to the 1985 water year (October 1, 1984 through September 30, 1985), pH and net alkalinity were relatively constant, while iron and manganese concentrations varied by a factor of 10 (fig. 3).

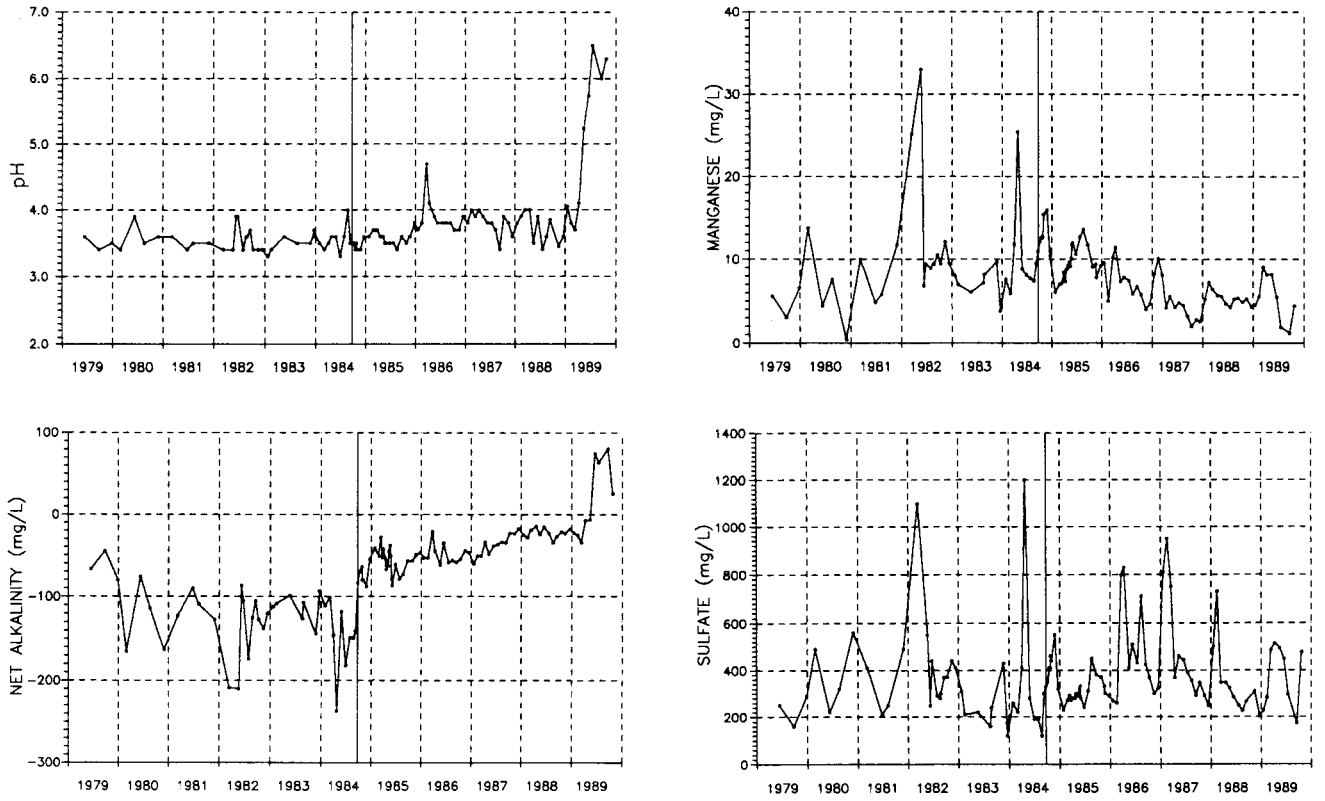



Figure 3. Changes in selected water-quality characteristics over time at mine site 1 in Lycoming County. The vertical line in 1984 indicates the time when deep mine daylighting and increased alkaline addition began.

Overburden from the three drill holes sampled is characterized by low total sulfur and low NP contents. However, many samples are composites of strata from large intervals (up to 28 ft thick). The highest total sulfur content measured was 1.29%. Only one sample, other than the coals, contained sulfur exceeding 0.5%. The highest NP was 33 tons $CaCO_3$/1,000 tons.

Pit cleanings were segregated from the overburden using a loader and were placed high in the backfill. The operator initially applied nearly-pure $CaCO_3$ (a limestone crusher screening waste product) to the pit floor and upper part of the backfill at a total rate of about 50 tons/acre. In late 1984, roughly at the same time the deep mine complex was encountered, this rate was increased to approximately 500 tons/acre, a rate that exceeded permit requirements. Approximately 50 tons/acre were spread on the pit floor and 100 tons/acre were applied to the upper surface of the rough backfill. The remaining 350 tons/acre were distributed through the backfill using limestone crusher screenings as blast hole stemming. From January 1985 through September 1989, a total of 19,000 tons of calcareous material were applied over 48 acres. Table 2 indicates that the overburden NNP increases, and is positive, after the addition of limestone on the basis of MPA computed by multiplying total sulfur by 31.25 or 62.5; however, NNP is negative if thresholds are used. For site 1 the NNP computed using the thresholds is unreliable because large intervals of strata were sampled and composited causing the possible dilution of high-sulfur or high-NP concentrations.

Figure 3 shows that a dramatic increase in net alkalinity, accompanied by a decrease in iron concentrations, occurred during the 1985 water year and continued through the 1989 water year. Sulfate and manganese concentrations, however, show little change (fig. 3). Manganese has a tendency to remain dissolved in acidic to neutral solutions (pH < 8) whereas iron will readily precipitate at pH > 4.5 (Hem, 1985). This increase in alkalinity is concurrent with the deep-mine daylighting and the application of alkaline material at a rate of 500 tons/acre. The increasing alkalinity of the water in combination with the persistently elevated sulfate concentrations indicates that acid production continues in the spoil but is neutralized.

Since 1988, water quality at the deep-mine discharge has attained "conventional mine-drainage effluent standards" (alkalinity > acidity; 6 < pH < 8; Fe < 6 mg/L; Mn < 4 mg/L).

The apparent neutralization of acidity indicates that the addition of alkaline materials played the dominant role in effecting the water-quality improvement. The removal of unmined, pyritic coal pillars by daylighting may have reduced that contribution of sulfate; however, surface mining may have produced additional sulfate which offsets the daylighting reduction. Furthermore, if neutralizers were liberated from the overburden, net alkalinity would have increased prior to the alkaline addition.

**Mine Site 2: Cambria County.** Site 2 encompassed 26 acres of Mercer coal removal by surface mining. No part of the permit area had been previously mined, although the adjacent property had been extensively mined, with AMD resulting. A period of 43 months elapsed from initiation of mining on mine site 2 to final backfilling (0.6 acres/month). Maximum highwall height was 45 ft.

Site 2 occupies the crest of a gently sloping hill and recharge to the site is predominantly from precipitation. Because AMD occurred at an adjacent mine, the permit approval for mining at site 2 required overburden analysis and alkaline addition. The strata were assumed to lack substantial carbonate minerals, so only total sulfur was determined in the overburden samples.

The sulfur concentrations and lithologic descriptions of samples from one overburden hole drilled within the area mined were used in ABA calculations (table 2); however, additional lithologic data from other drill holes were used to define the stratigraphy. The overburden composition and thickness differed across the site, because of removal of strata by erosion and replacement by channel sandstones, and because of bifurcation (splitting) of the coal seam. The number of splits in, and thickness of the coal (2 to 5 ft), differed from hole to hole. The majority of the strata consists of sandstone, with "coal spars" at the base. In most drill holes a 0.5- to 3-ft-thick, black, carbonaceous shale was encountered immediately above the coal. The overburden hole encountered 1.5 ft of this black shale, which contains 1.12% total sulfur, and 2 ft of coal, which contains 2.64% total sulfur. Because of the numerous binders in the coal, approximately 30 percent of the coal horizon was spoiled as "pit cleanings." The pit cleanings were segregated and placed in pods a minimum of 10 ft above the pit floor to keep them above the water table after the mine was reclaimed. Because of the large amount of pit cleanings and the thin cover, some of this material was placed within a few ft of the final graded surface.

The alkaline material consisted of baghouse limestone (captured air-borne particulate material), which was applied at a rate of 42 tons/acre. About 22 tons/acre were added to the pit floor; about 10 tons/acre were added to the tops of the pit-cleaning pods and to the backfilled surface prior to replacement of topsoil.

About 200 ft downslope and downdip from mine site 2, water discharges from a seep on the adjacent, previously mined area. Samples collected over 3.5 years define conditions prior to mining at site 2 (fig. 4). Since mining at site 2, the pH of the seep has decreased, and acidity, sulfate, iron, and manganese concentrations have increased. Following backfilling, the seep water quality has maintained significantly lower pH and net alkalinity and greater iron, manganese, and sulfate concentrations than are present in the premining samples (fig. 4). Water from a monitor well screened through the backfill was similar in quality to that from the seep. Alkaline addition has not prevented acid formation at this mine site. AMD may have been predicted on the basis of the negative NNP (table 2).

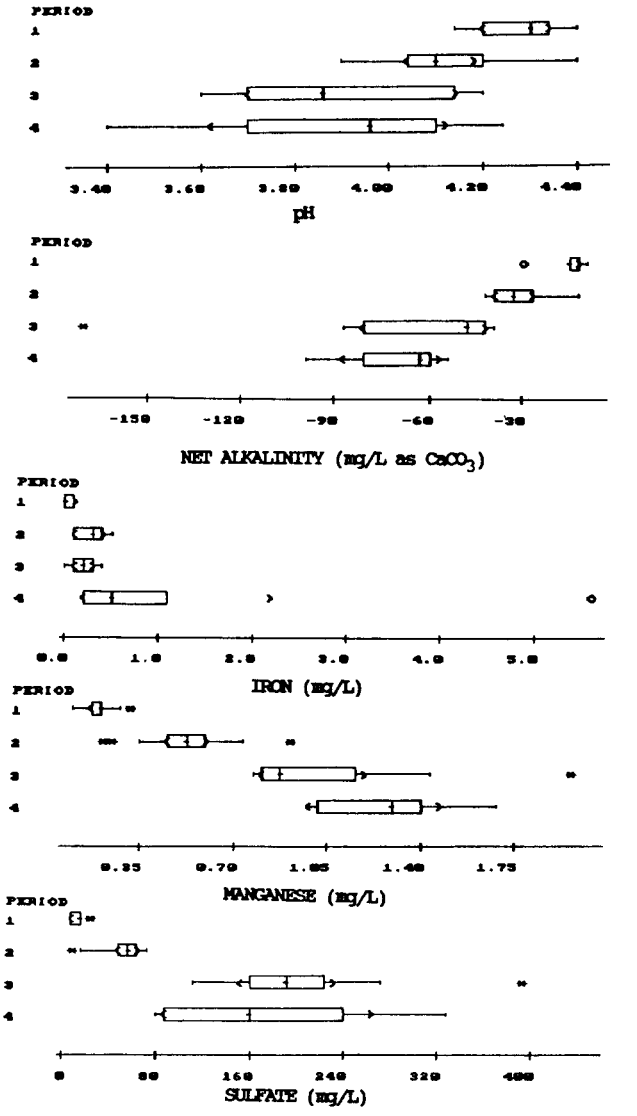



Figure 4. Boxplots showing changes in water quality at mine site 2 in Cambria County. Period 1 represents data collected over a 3-year period prior to activation of mining (N=14). Period 2 is the first 20 months of mining (N=14), and period 3 is the second 20 months of mining (N=11). Period 4 represents post-mining water quality (N=6).

**Mine Sites 3 and 4. Cambria County.** Site 3 is an unreclaimed mine similar in size and configuration to the adjacent mine site 4. No alkaline addition or special handling was conducted at mine site 3, and the backfilling and reclamation are incomplete. In contrast, mine site 4 is a recently reclaimed mine where alkaline materials have been added to a 75-acre area where the lower Kittanning, lower Kittanning rider, and upper Kittanning coals were mined. The site was active for 60 months (1.25 acres/month). Site 4 can be categorized as a contour block-cut operation with a maximum highwall height of 125 ft. Mining and backfilling were on a continuous basis; the open pit area did not exceeded 100 ft by 300 ft.

Surface and ground waters affected by nearby deep mining or surface mining of the lower Kittanning are severely degraded, whereas waters from unmined areas have near-neutral pH, low buffering capacity, and low concentrations of metals. Although mine site 4 is adjacent to previously affected areas (including mine site 3), it is neither upflow nor downflow from preexisting AMD.

Overburden-analysis data for mine sites 3 and 4 indicate the presence of some potentially acid-producing strata associated with and overlying the lower Kittanning and lower Kittanning rider seams and the absence of strata having NP greater than 30 tons $CaCO_3$/1,000 tons. Overburden quality is summarized in tables 1 and 2. Selective handling of pyritic materials, removal of pit cleanings, and alkaline addition were performed at mine site 4 to avoid post-mining water-quality problems associated with mine site 3. Hydrated lime was added at a total rate of 240 tons/acre [648 tons $CaCO_3$/acre, assuming $Ca(OH)_2$ has 2.7 times the neutralization capacity of $CaCO_3$ (Cravotta et al. 1990)]--a rate that exceeded permit requirements. The alkaline material was distributed on the pit floor at a rate of 54 tons $CaCO_3$/acre; the remaining 594 tons $CaCO_3$/acre were spread over the surface of the rough backfill prior to topsoil replacement (table 1). In accordance with the $CaCO_3$ deficiency, the above alkaline addition rate represented 44% of the total calculated deficiency. The plans for special handling and alkaline addition were diligently implemented, and most of the pyritic materials associated with the lower Kittanning coal were exported from the mine site.

Figure 5 compares the post-mining water quality for two toe-of-spoil discharges at the downdip boundaries of mines 3 and 4. Both sites have AMD; except for iron, the water quality at the alkaline addition site 4 is significantly better than that at site 3. The traditional method of computing the overburden MPA yields positive NNP (table 2); however, use of thresholds or a multiplication factor of 62.5 in ABA computations yields negative NNP values (table 2), which are consistent with the negative value for post-treatment net alkalinity at both sites. The water-quality difference between sites 3 and 4 is attributable to the addition of alkaline materials and the different mining and reclamation methods employed at the two sites.

**Mine Site 5: Armstrong County.** Mine site 5 encompasses 19 acres. The lower and middle Kittanning coal were removed by surface mining of cover that was 30 to 80 ft thick. The area had been mined previously to a depth of 30 to 40 ft. Eight months elapsed from initiation of mining to completion of backfilling (2.4 acres/month). Reclamation was concurrent with mining.

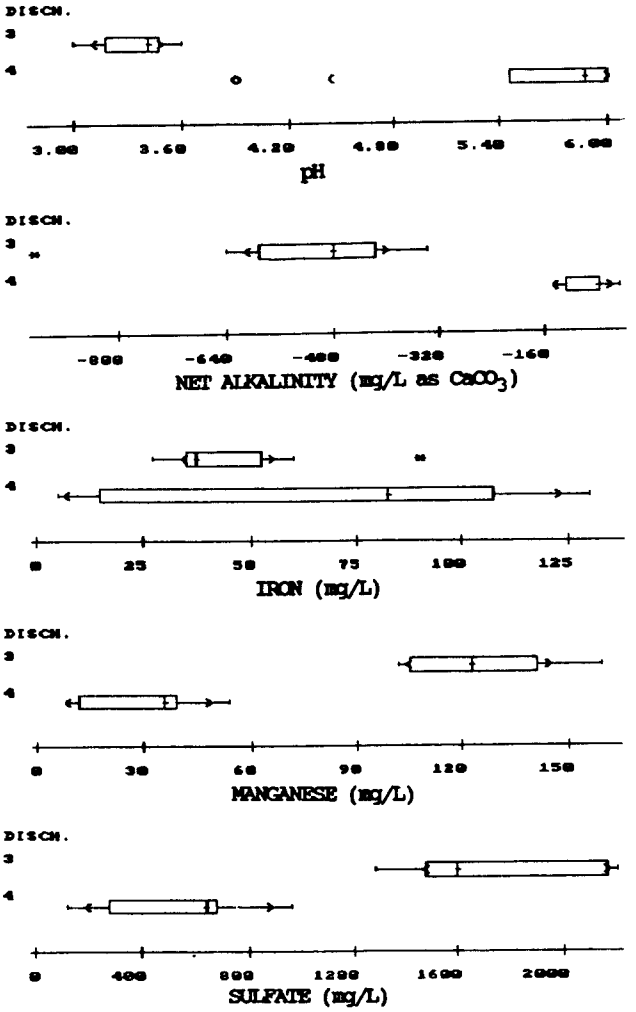



Figure 5. Boxplots showing comparison of water quality data for mine sites 3 (N=8) and 4 (N=5) in Cambria County. Mine site 4 had alkaline addition; mine site 3 did not and served as a control.

Several toe-of-spoil discharges emanated from the previously mined area. The water quality was generally alkaline, and the pH was 5 to 6. Site 5 is located on a hillside with a substantial area upslope from the mine. Ground-water recharge to the site is from direct precipitation and from the drainage of upslope areas.

The overburden hole closest to the area mined was drilled above the highwall through strata that were not mined. The unmined, upper cover was not included in the ABA calculation, and the top part of the hole, to the weathered depth of 20 ft, was assumed to be inert (NP and sulfur are 0). Nine ft of shale above the lower Kittanning coal had total sulfur content ranging from 0.69 to 1.36%, and 28 ft of overlying strata had NP ranging from 30 to 72 tons $CaCO_3$/1,000 tons. The composite strata from this hole contained an equivalent NP as 3,875 tons $CaCO_3$/acre extrapolated over the area mined. Although the overburden analysis for the area mined shows an overall excess of neutralizers (table 2),

overburden holes outside the area mined lacked alkaline strata, and AMD discharges from nearby mines. Therefore, alkaline addition was performed at mine site 5 because of the uncertain effect of mining on water quality and the variable overburden composition. The alkaline material consisted of imported limestone dust that was added at a rate of 140 tons/acre to the backfill surface prior to topsoil replacement.

Boxplots in figure 6 show the quality water from a toe-of-spoil seep below mine site 5. The initial water quality is representative of the effects of the preexisting shallow-cover mining on water quality. The subsequent samples illustrate water quality during and after the thick-cover mining. There is no significant difference between the pre- and post-mining water quality (pH, net alkalinity, iron, manganese, and sulfate) at the 95-percent confidence level. Even the lowest pH and highest concentrations of sulfate and metals measured during the post-mining period meet conventional effluent limits. The positive net

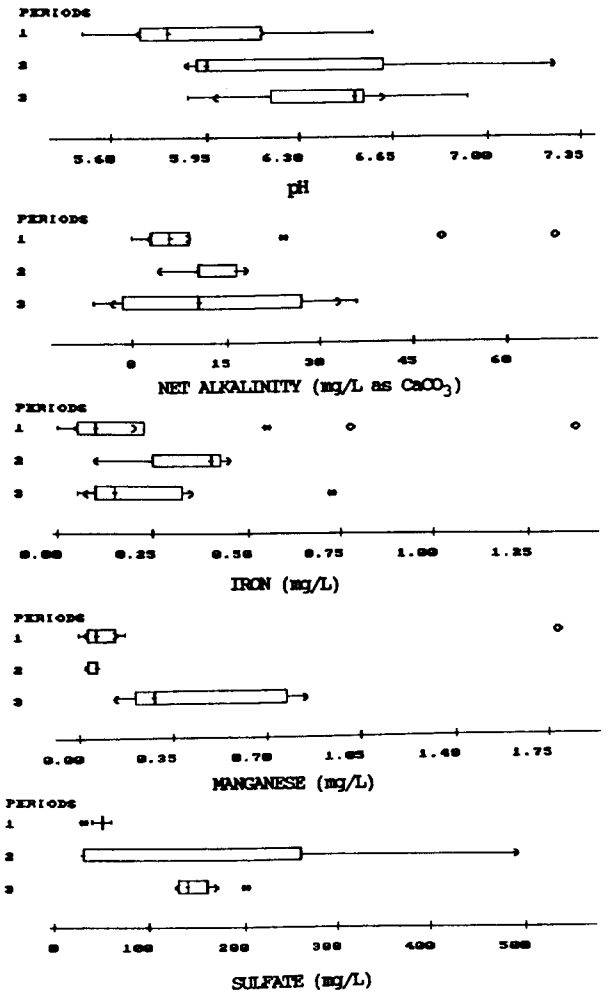



Figure 6. Boxplots showing changes in water quality through time for mine site 5 in Armstrong County. Time 1 represents "premining" data (N=13) over a 2-year period. Time 2 is data collected during mining (N=3), and time 3 is 1 year of post-mining water quality (N=7).

alkalinity of the post-treatment drainage is consistent with positive NNP computed by any method (table 2).

The mining and reclamation at site 5 did not cause degradation of the seep; however, the remedial effects of alkaline addition cannot be isolated from those of naturally occurring calcareous strata. The quantity of naturally alkaline material in the mine spoil was much greater than that of imported alkaline material. Additionally, ground water seepage to the open pit was probably alkaline, and the mining operation proceeded very quickly, thus limiting the aeration of the spoil.

Mine Site 6: Clarion County. Mine site 6 consists of two adjacent, similarly-sized mines, 6A and 6B. Approximately 54 acres of upper and lower Clarion coal were mined at site 6A where alkaline materials were applied during mining and reclamation. The same seams of coal were removed at mine site 6B, but alkaline addition was not implemented; mine site 6B serves as a control. Both mines are located on the side of a hill, and both mines receive some recharge from areas above the area mined. Prior to mining at site 6A, water did not discharge from the coal outcrop, but during mining, seepage(?) water was observed in the open pit.

Rock samples from the two overburden holes indicate considerable differences in the stratigraphic positions of the high-sulfur and alkaline strata. The strata with the highest sulfur content were encountered between the lower and upper Clarion coal seams. Other than the coal samples, no strata contain sulfur greater than 1.0%. Drill hole 1 encountered a 12-ft-thick shale between the Clarion coals that had NP of 36 to 43 tons $CaCO_3$/1,000 tons. Drill hole 2 encountered strata between the coals that had a maximum NP of 21 tons $CaCO_3$/1,000 tons and strata immediately above the upper Clarion coal that had a NP of 32 tons $CaCO_3$/1,000 tons. The amount of naturally alkaline material (composite NP for all strata) was equivalent to 1,925 tons $CaCO_3$/acre for the mined area. A summation of the overburden quality is given in table 2.

Approximately 21 months elapsed from initiation of mining to completion of backfilling at mine site 6A (2.6 acres/ month). Maximum highwall height was about 85 ft. Pit cleanings were selectively handled, and placed at least 10 ft above the pit floor. Alkaline materials were applied as a "safety factor." Baghouse limestone with a $CaCO_3$ equivalent of nearly 100 percent was applied to the pit floor at a rate of 20 tons $CaCO_3$/acre and near the top of the backfilled spoil, but beneath the topsoil, at a rate of 80 tons $CaCO_3$/acre. Application rates on the pit floor on at least one occasion were greater than 20 tons $CaCO_3$/acre. Nevertheless, the quantity of alkaline material added to the spoil may not have been adequate to offset the pre-treatment, negative NNP (table 2). If MPA is computed by multiplying total sulfur by 31.25, then NNP is positive after alkaline addition; however, if MPA is computed by multiplying total sulfur by 62.5, then NNP is negative. Addition of limestone increased the overburden NNP from -3.82 to only -3.56 tons $CaCO_3$/1,000 tons on the basis of the 62.5 factor (table 2).

Discharges from mine sites 6A and 6B were both alkaline and acidic. Special handling of overburden or addition of alkaline material was not performed at mine site 6B, which serves as a control. Post-treatment water-quality data for several discharges are grouped by mine site and are shown as boxplots in figure 7. There is no significant difference between the post-treatment water quality of sites 6A and 6B at the 95-percent confidence level. The negative values of NNP after alkaline addition, on the basis of MPA computed using the factor of 62.5 (table 2), suggest that acidic discharge from mine site 6A may have been predicted. Thus, the alkaline-addition rate at site 6A may not have been adequate to cause substantial improvement in the water quality.

**Mine Site 7: Clarion County.** Minesite 7 encompassed a mined area of approximately 60 acres of lower Clarion coal and 31 acres of upper Clarion coal. A period of 20 months elapsed from initiation of mining to completion of backfilling (3 acres/month). The maximum highwall height was 60 ft. Although mine site 7 is situated on a hilltop, in a groundwater recharge area, considerable water was encountered during mining. The downflow monitor well consistently had a static water level of 0.5 to 1 ft above the bottom of the coal.

A 2- or 3-acre portion of site 7 was previously mined and abandoned. It had developed a mildly acidic discharge, which could possibly be eliminated by additional mining and reclamation.

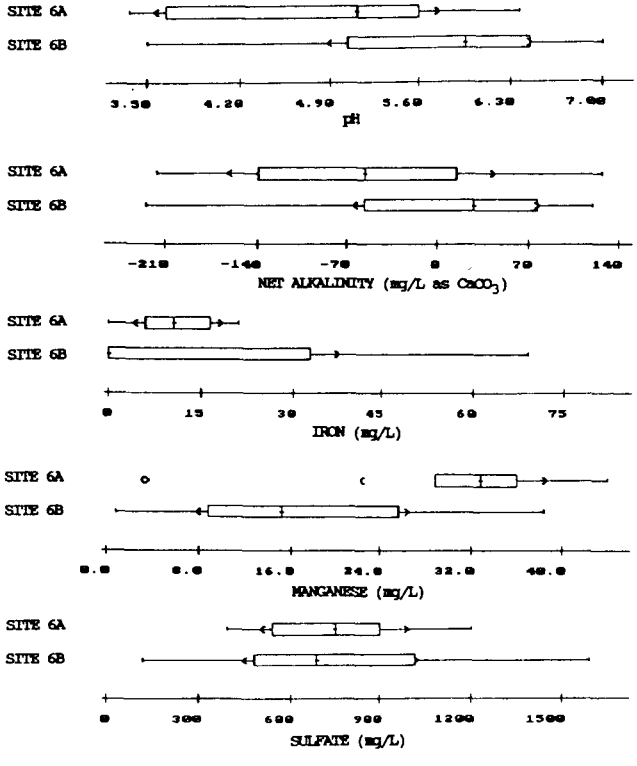



Figure 7. Boxplots comparing water quality of multiple discharges from mine site 6A (N=6) with those from mine site 6B (N=11) in Clarion County. Mine site 6A had alkaline addition; mine site 6B did not and served as a control.

Because the overburden analysis characterized this site as low sulfur and low NP (tables 1 and 2), mining of the additional 60 acres was planned with an alkaline addition rate of 300 tons $CaCO_3$/acre (about 19 percent of the calculated NNP, where MPA is computed using a factor of 62.5). A 2-ft-thick sandstone stratum contained total sulfur of 0.78% and NP of 28 tons $CaCO_3$/1,000 tons, the highest measured in the overburden, respectively. A 5-ft-thick shale stratum had 0.53% total sulfur. Other than the coal, these are the only strata that contained greater than 0.5% total sulfur.

There was no selective handling plan, however the rock between the Clarion coal seams was routinely placed high in the backfilled mine spoil above the mine floor. Baghouse limestone with a $CaCO_3$ equivalent of nearly 100 percent was applied at a total rate of 300 tons $CaCO_3$/acre. Approximately 280 tons $CaCO_3$/acre were placed on the backfilled surface, below the topsoil, and an additional 20 tons $CaCO_3$/acre were spread on the pit floor.

Figure 8 shows temporal variations in concentrations of acidity and iron in water sampled from a downflow monitor well from 1982 through 1986. Water quality of seepage prior to mining in 1982 can be characterized as mildly acidic with moderate metals and sulfate concentrations, which indicate some effects from previous mining. Figure 8 shows that the concentrations of acidity and iron increased through the period of mining, and then decreased following backfilling. The addition of

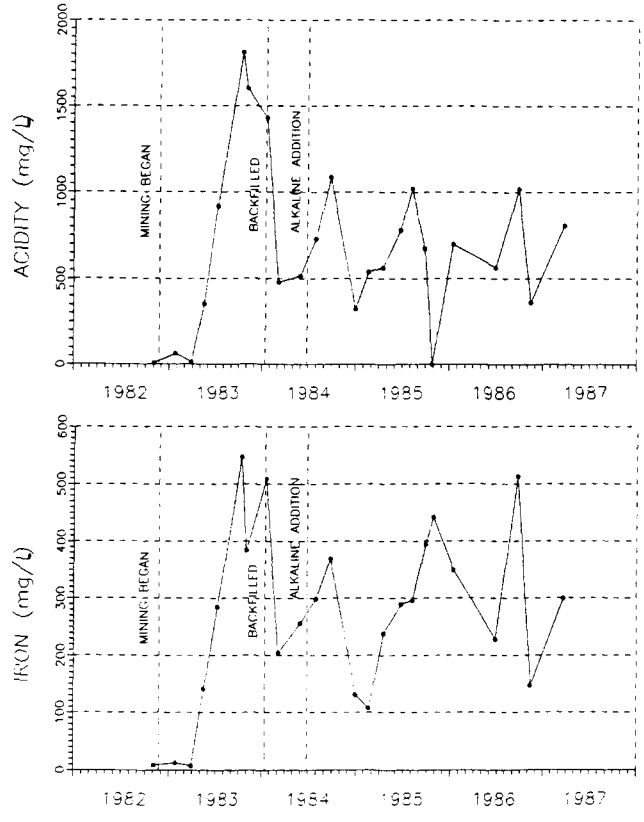



Figure 8. Acidity and iron concentrations over time for a downflow monitor well at mine site 7 in Clarion County.

alkaline materials had no apparent effect on the water quality, which remained considerably worse than that of the pre-mining period. Although the preexisting discharge disappeared, alkaline addition failed to prevent or abate acid ground water. The negative NNP after alkaline addition (table 2) suggests that insufficient quantities of alkaline materials were added and that acidic, post-mining water quality may have been predicted.

**Mine Site 8. Clarion County.** Mine site8 encompassed a 29-acre area mined for the middle Kittanning coal, 2 acres of which were mined for the upper Kittanning coal. The mine site can be characterized as a hilltop removal/block-cut operation. The maximum overburden thickness was 85 ft. Mining occurred on a continuous basis over a 22 month period (1.3 acres/month), and the open pit area did not exceed 300 by 500 ft. The premining water quality at site 8 can be characterized by near-neutral pH, positive net alkalinity, and low concentrations of iron, manganese, and sulfate (fig. 9).

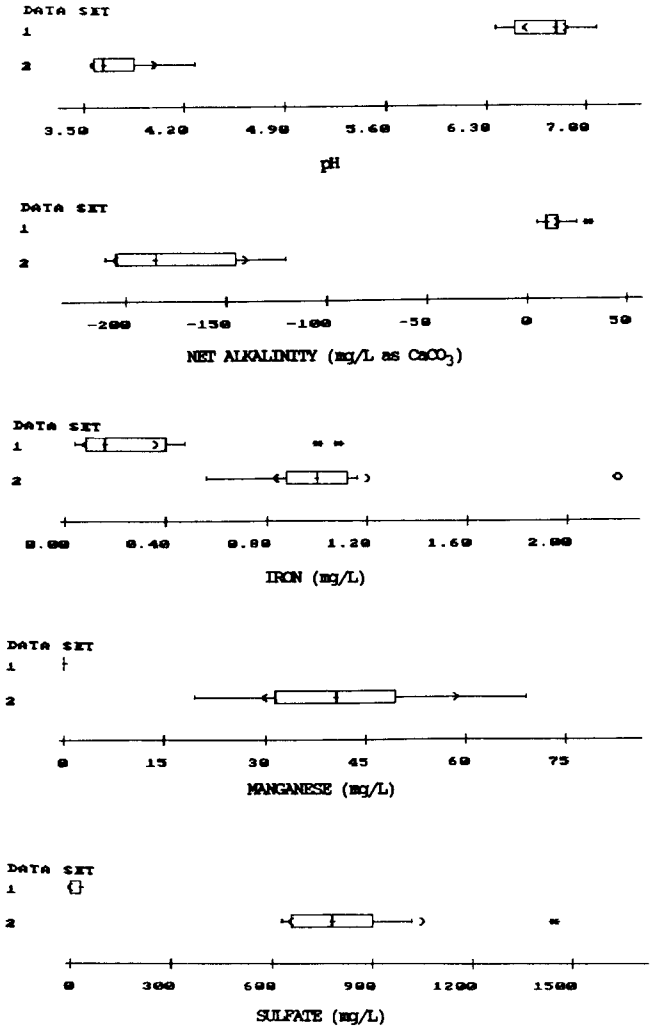



Figure 9. Comparison of pre- (data set 1) and post-mining (data set 2) water quality at mine site 8 in Clarion County. Data set 1 is for a monitor well (subsequently mined through) (N=12). Data set 2 is for a toe of spoil discharge (N=6).

Two holes were drilled in overburden at or near the maximum anticipated final highwall. One of the holes penetrated some strata with NP exceeding 30 tons $CaCO_3$/1,000 tons, but overall the strata had only a slight positive NNP; however, the overburden penetrated by the other hole exhibited a negative NNP (alkaline deficiency). Table 2 summarizes the ABA results of the overburden analysis. The overburden is predominantly sandstone, which contains carbonaceous inclusions and which extends down to the middle Kittanning coal. The sandstone interval from the top of the coal to 2.5 ft above the coal had total sulfur contents ranging from 0.56 to 2.63%.

Most mining took place in the alkaline-deficient area; only a small part of the mine uncovered alkaline strata. Table 2 shows a positive NNP, which is probably higher than was actually encountered by mining. The alkaline-deficient strata encountered by one overburden hole are probably representative of the majority of the strata mined. However, overburden data for samples from both holes were used in ABA calculations.

The 2-ft thick sandstone stratum overlying the middle Kittanning coal seam was selectively handled and segregated. For the alkaline-deficient part of the permit area, alkaline materials were applied a rate of 52 tons $CaCO_3$/acre over the surface of the backfill prior to topsoil replacement.

Following backfilling and reclamation, acidic water began to discharge from an adjacent area downflow from site 8. Figure 9 compares the pre-mining quality of water from a well within the area mined with that from the post-mining discharge. The pH is lower, and concentrations of net alkalinity, iron, manganese, and sulfate are higher in the post-mining discharge than in pre-mining water at the 95-percent confidence level. No mitigative effects are apparent following the alkaline addition. Furthermore, alkaline, not acidic, discharge would be predicted on the basis of the positive NNP data in table 2; however, the calculated NNP may not be accurate because few alkaline strata were mined.

**Mine Site 9. Clarion County.** Approximately 48 acres of upper and (or) lower Clarion coals were mined intermittently from 1961 through 1975 at mine site 9. About half the area was mined for the lower Clarion coal leaving a 70-ft highwall separating an upper and lower bench. At the completion of mining, the mine was backfilled with overburden and coal waste, including tipple refuse from other mines, and was regraded to the original hillside configuration. A toe-of-spoil seep produced severe AMD that required treatment. In 1984, calcareous waste materials were applied to the surface in quantities that were not sufficient to neutralize MPA but sufficient to produce alkaline water in the unsaturated zone (pH > 4.5) that would presumably inhibit bacteria thought to catalyze pyrite oxidation (Waddell et al. 1980).

The mine receives recharge from the reclaimed hillside and undisturbed hilltop. The reclaimed mine spoil has a total thickness ranging from about 10 to 70 ft, in which ground-water saturation varies from less than 5 to nearly 20 ft. The lower Clarion coal crops out along an intermittent stream at the southern boundary of the mine. The coal was not mined near the stream channel in an attempt to restrict the discharge of ground water from the

mine; this ground water is acidic (pH < 3) and contains substantial concentrations of dissolved sulfate, iron, and manganese (Williams et al. in press).

The overburden and coal are characterized as having high sulfur and negligible calcareous contents (tables 1 and 2). Eleven samples collected during drilling of a 90-ft-deep drill hole through the highwall represent a relatively complete stratigraphic column; typical concentrations of total sulfur range from 0.05 to 4.77%. The highest sulfur concentrations were from the lower Clarion coal and from strata about 1 ft above and below the coal. None of the sampled strata had a NP greater than 3.5 tons $CaCO_3$/1,000 tons. Samples of the tipple refuse had a sulfur content of 1.5 to 2.18%.

In the spring of 1984, limestone fragments (nearly pure $CaCO_3$) and lime-kiln flue dust [$Ca(OH)_2$] were applied on the land surface within two 2.5-acre treatment plots and a control plot. ABA computations were performed using overburden analyses of samples from two or more drill holes in each 2.5-acre plot. Treatment plot 9A had a net deficiency of -1,769 tons/acre and received 40 tons/acre limestone plus 400 tons/acre hydrated lime. Treatment plot 9B had a net deficiency of -2,955 tons/acre and received 400 tons/acre limestone plus 120 tons/acre hydrated lime. However, the control plot 9C had a net deficiency of -3,990 tons/acre and received only 2.4 tons/acre hydrated lime (to allow grass planting). Thus, after alkaline addition, assuming hydrated lime has 2.7 times the neutralization capacity as $CaCO_3$, the net neutralization potentials for plots 9A, 9B, and 9C, respectively, were about -649, -2,231, and -3,984 tons $CaCO_3$/acre.

Ground-water-quality data were collected from two or three monitor wells in each of plots 9A, 9B, and 9C that were screened through the spoil to the mine floor (bedrock). Data for plots 9A and 9B were collected monthly during 1 year before and 3 years after surficial treatment with calcareous materials; however, monthly data for plot 9C were collected only 1 month before but 3 years after treatment. Figure 10 shows that the ground water in the mine spoil, before and after treatment, contained high concentrations of acidity (pH < 4.5; net alkalinity < 0) and of dissolved sulfate, iron, and manganese and that the water-quality generally differed between 9A, 9B, and 9C, reflecting variations in overburden composition and possibly other factors. A comparison of data before and after alkaline addition at plot 9B, shows that none of the parameters (pH, net alkalinity, Fe, Mn, or $SO_4$) is significantly different at the 95-percent confidence level. Data collected at plots 9A and 9C before and after treatment indicate that pH increased, and Mn concentrations decreased significantly after treatment. However, because the water quality at the control plot 9C may have improved, the improvement in water quality at treatment plot 9A cannot be attributed solely to the addition of alkaline material. Because pyrite and leachable minerals will be depleted as the spoil weathers, a gradual improvement in water quality is expected in both untreated and treated spoil. Nevertheless, if alkaline additives were to prevent or slow the oxidation of pyrite, a difference in the rate of improvement would be expected for the treatment plots 9A and 9B compared with the control plot 9C.

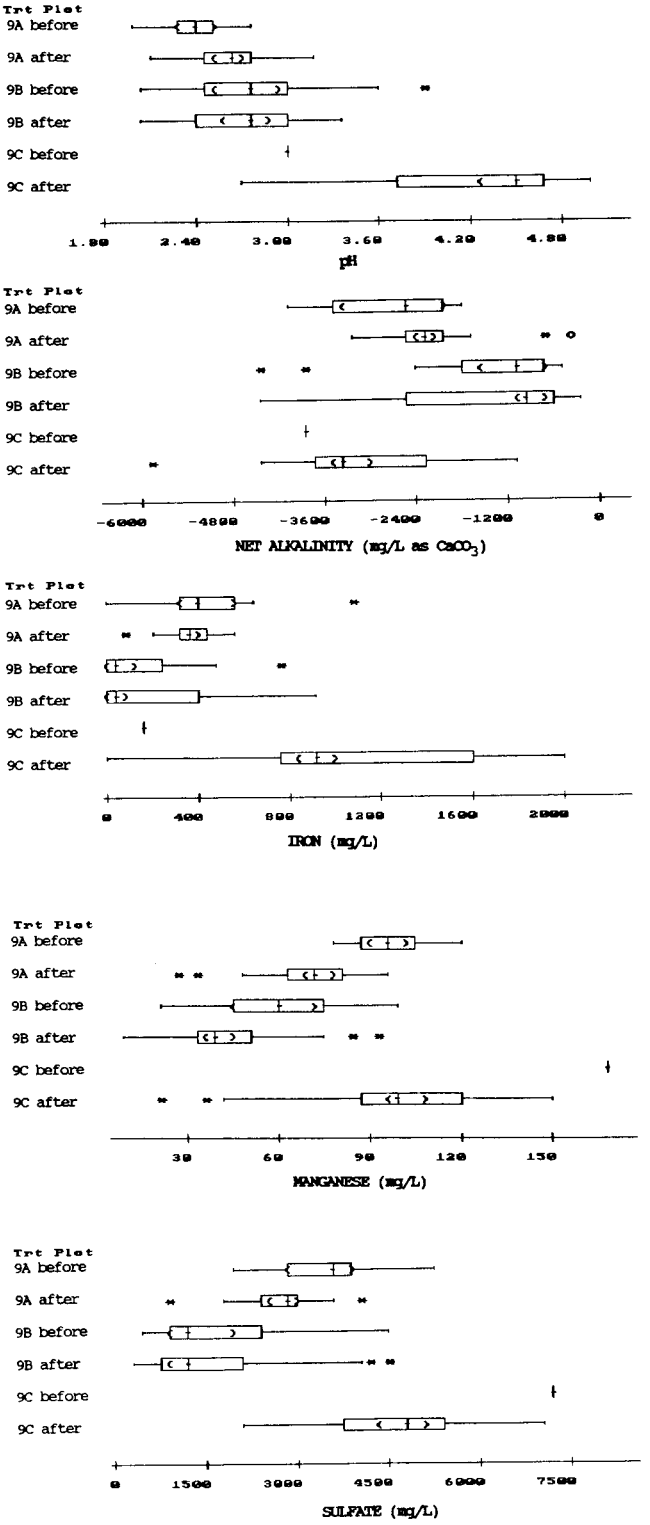



Figure 10. Boxplots of water quality from monitor wells for three 2.5-acre plots at mine site 9 in Clarion County. Plots 9A and 9B had alkaline addition, plot 9C served as a control. The boxplots compare water quality before and after treatment with alkaline additives. Number of samples for each boxplot are as follows: 9A before (N=12), after (N=39); 9B before (N=23), after (N=70); 9C before (N=1), after (N=61).

However, nonparametric trend tests of the post-treatment data indicate that the rates of concentration changes are not significantly different between treated and untreated plots (Cravotta, U.S. Geological Survey, written commun., 1989).

Acid-mine drainage was not abated nor groundwater quality noticeably improved by the surficial application of alkaline materials at mine site 9. The NNP data in table 2 suggest that acidic ground water at each plot likely could have been predicted considering the MPA computed using the factor of 62.5 following the method of Cravotta et al. (1990). One possible explanation for the persistent acidic ground water in the reclaimed mine spoil is that prior to treatment, pyrite in the stockpiled overburden or mine spoil had been oxidized to ferric-sulfate minerals, which then provided a source of long-term, leachable acid, sulfate, and iron; the alkaline materials may have been applied too late to inhibit the oxidation of pyrite. Another explanation is that the rate of pyrite oxidation exceeds the rate of dissolution of calcareous materials.

**Mine Site 10: Venango County.** Mine site 10 was included in the study because limestone and other calcareous rock were present and abundant relative to potentially acid-producing strata. Forty-one acres of Brookville coal were mined, using the box cut method, with trucks and loaders. Maximum highwall height was about 50 ft.

About 8 to 10 ft of marine limestone strata is present about 25 ft above the coal at site 10. The limestone has NP ranging from 536 to 932 tons $CaCO_3$/1,000 tons. Another 7 to 13 ft of strata has NP over 100 tons $CaCO_3$/1,000 tons. However, a 1-to-1.5-ft-thick stratum immediately above the coal contains 1.09 to 5.5% sulfur, and underclay below the coal contains 1.53 to 2.9% sulfur. No alkaline addition was proposed because the natural strata contained the equivalent of about 18,000 tons $CaCO_3$/acre. The sulfur-bearing strata immediately overlying the coal were segregated and placed high above the mine floor in the backfilled mine spoil.

The NNP and net alkalinity data in table 2 indicate that the discharge would be alkaline on average. Post-mining discharge quality, however, has varied from highly alkaline to moderately acid (fig. 11). The most acidic water was collected after heavy rains. The acidity probably result from recharge that dissolves ferric-sulfate minerals that had accumulated in the unsaturated zone during drier periods and from unequal rates of pyrite oxidation and dissolution of limestone. The water samples contain iron and manganese concentrations that exceed conventional effluent limits (6 and 4 mg/L, respectively). These metals and the elevated sulfate indicate that the abundant alkaline material has not prevented the oxidation of pyrite in the backfill, but has neutralized the acidity produced by the oxidation reaction.

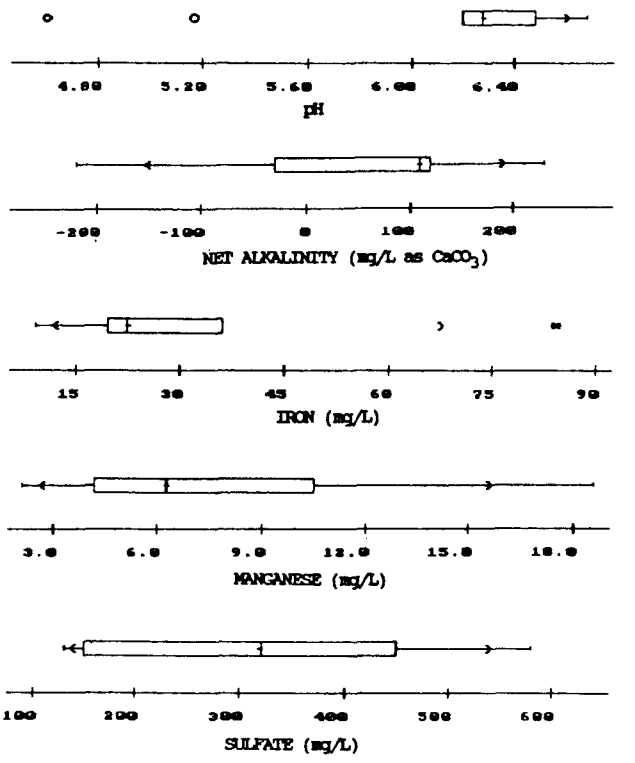



Figure 11. Post-mining water quality of discharge from mine site 10 in Venango County (N=5). This mine had abundant naturally occurring alkaline strata.

## Results and Discussion

Every mine site is unique. Overburden composition, mining methods employed, volume of rock disturbed, pre- and post-mining ground-water chemistry, hydrogeology, weathering, and many other factors differ among the sites. Each of these factors affects the associated post-mining water quality. Consequently the water quality at each mine site is monitored according to a unique program. Table 1 illustrates some similarities and differences among mine sites in this study. Alkaline-addition rates ranged from zero at sites 3 and 10, which were included for comparative purposes, to greater than 1,000 tons $CaCO_3$/acre for a part of site 9.

The uniqueness of the mines and number of variables to be considered makes a comparison of mine sites difficult; however, some relations between post-treatment, cumulative NNP of overburden and median net alkalinity of mine drainage from the 10 mine sites are apparent (table 2). Six of the eight alkaline-addition plans (mine sites 2, 4, 6, 7, 8, and 9) failed to prevent AMD. Of the two sites where post-treatment discharge water was alkaline (sites 1 and 5), site 5 has a substantial thickness of naturally alkaline strata, which were likely to produce alkaline drainage without the supplemental addition of limestone. After alkaline addition, the overburden NNP at mine site 1 was positive (+6.90 tons $CaCO_3$/1,000 tons) if calculated by the traditional methods of ABA; however, NNP was slightly negative (-0.31 tons $CaCO_3$/1,000 tons) if calculated using thresholds (table 2). The water quality at site 1 improved after the addition of alkaline material (fig. 3). The formerly acidic discharge from the deep mine that underlies much of the site now meets conventional mine-drainage effluent standards. Although discharges from alkaline-addition site 4 do not meet conventional effluent limits, the water quality is substantially better than that of nearby discharges from mine site 3, an untreated control (fig. 5).

Mine sites 1 and 4, the two sites where water quality improved after the addition of imported alkaline materials, had several things in common: (1) They were treated with the largest total quantities and the second and third largest alkaline-addition rates of the sites studied (table 1). (2) The mining and reclamation techniques used at both sites included selective handling of pyritic materials, timely backfilling, and alkaline-addition rates that exceeded permit requirements. (3) Some potentially acid-forming strata were removed from the mine sites. The coal seam on mine site 1 that was causing acid water to emanate from the deep-mine discharge was removed from the mine site by daylighting. Mine site 4 included removal of most of the pit cleanings.

Two adjacent mine sites were also examined in Clarion County. Mine site 6A received alkaline additives, and mine site 6B did not. No observable improvement in water quality is apparent from the alkaline additives, which may not have been applied in adequate amounts.

Mine site 9 is the only site in this study where alkaline material was applied solely as a remedial measure 9 years after completion of mining and reclamation. Three 2.5-acre plots were studied, two with alkaline addition and one without. All three continued to produce severe AMD through 1987 fig. 10). The AMD production results from the limited amount of alkalinity generated from the limestone relative to the amount of acidity produced by pyrite oxidation and by leaching of previously formed ferric-sulfate minerals in the mine spoil.

Alkaline materials were not added at mine site 10, but the site was included in this study to illustrate the potential effect of large amounts of naturally alkaline strata on water quality at a site where there is also some potentially acid-producing strata. The strata at site 10 contained the equivalent of 18,000 tons $CaCO_3$/acre. On average, the post-mining water quality was alkaline, but the concentrations of metals and sulfate were elevated (fig. 11). Apparently, the presence of large amounts of naturally alkaline material do not always preclude pyrite oxidation and the dissolution of metals and sulfate from the mine spoil.

The variable alkaline and acidic water quality of several discharges from site 6 show that alkaline and acid conditions can be created within the same spoil. Mine site 10, which had an abundance of limestone strata, had a single discharge that varied from alkaline to acid following recharge events. Alkaline-addition sites also have produced alkaline and acidic discharges, but with median net alkaline discharge. Clearly, the processes of pyrite oxidation and carbonate neutralization are complicated by the unequal distribution of acid-forming and neutralizing materials. Under some conditions, alkaline waters need to contact the locally pyritic zones of the mine spoil to prevent or neutralize AMD. Hydrogeochemical factors such as the microbial activity; mineral-surface areas; $O_2$, $CO_2$, nutrient, and moisture contents; temperature; and pH of mine spoil also must be considered.

Table 1 generally indicates mine sites where imported alkaline materials were placed, and if special overburden handling was implemented. Each of the eight alkaline-addition sites, except site 1, had alkaline materials placed entirely, or mostly, at the spoil surface--the easiest method of application. Alkaline materials were incorporated on the pit floor and within the spoil at five and two of the sites, respectively. Although alkaline-materials were spread on the pit floors at mine sites 1 and 4, where water quality improved after alkaline addition, alkaline materials were also incorporated with the backfill at site 1 and pyritic materials were selectively handled at site 4. Furthermore, sites 2 and 8 also employed selective handling of pyritic material and small amounts of variably placed alkaline additives, but both produced AMD (table 2). Therefore, effects of placement of alkaline materials or selective handling of acidic materials cannot be readily evaluated in this study.

Table 2 shows the overall mass-weighted NNP for each study site before and after alkaline addition. Although total quantities, or rates of application, of alkaline materials appear to be large, they generally are insignificant relative to the NP or NNP of the entire overburden volume before alkaline addition. Table 2 also shows the median post-mining net alkalinity of associated mine drainage or ground water for the 10 mine sites. Mine site 9 was subdivided into three plots 9A, 9B, and 9C, which in combination with mine sites 1-8 and 10, allows comparison of overburden NNP and water-quality net alkalinity for 12 sets of data. By comparing the sign on NNP and net alkalinity values in table 2, it is apparent that the traditional ABA computation of MPA, by multiplying total sulfur, in weight percent, by 31.25, results in mismatched signs--a wrong prediction of water quality--for 4 of the 12 sites. The errors in prediction are that the mine spoil is alkaline (NNP > 0), whereas the associated water is acidic (net alkalinity < 0). Prediction was not improved by using thresholds, where only values of NP greater than 30 tons $CaCO_3$/1,000 tons (with fizz) and of total sulfur greater than 0.5% are used to compute NNP, although substantially different values of NNP resulted. However, if MPA is calculated by multiplying total sulfur, in weight percent, by 62.5 following the method of Cravotta et al. (1990), then the sign of NNP matched the sign of the overall net alkalinity of waters at 11 of 12 sites (table 2). The acidic discharge from mine site 8 was not predicted by any of the acid-base accounting computation methods, possibly because of an inaccurate estimate of NNP.

The method of ABA calculation that used "thresholds" eliminated some of the problems associated with low-NP overburden, which typically produces AMD even though the sulfur content may be low. For example, mine sites 3 and 4 produced AMD which was correctly predicted by the NNP "with thresholds" (table 2). Using only NP values greater than 30 tons $CaCO_3$/1,000 tons, with a "fizz," eliminated much of the presumed influence of siderite on NP determinations (Morrison et al. 1990). Unless the influence of siderite in NP determinations can be eliminated, thresholds remain a useful concept. However, a disadvantage of assuming strata are inert if total sulfur and NP content do not exceed the threshold values is when large intervals of strata are sampled and composited, causing dilution of high-sulfur or high-NP concentrations. For example, the NNP calculated using thresholds are unreliable for mine sites 1 and 9, for which large thicknesses (up to 28 ft) of composite overburden samples were analyzed.

The results of this study are consistent with previous reports that concluded that NP and "traditional" MPA values, despite being reported in the same units (tons $CaCO_3$/1,000 tons), are not equivalent. In previous practice, it appeared that overburden NP must be at least twice MPA to produce alkaline mine drainage (diPretoro 1986; Skousen et al. 1987; Brady and Hornberger 1989; Ferguson and Erickson 1988). The observed inequality in NP and MPA is partly attributable to the incorrect assumption that $CO_2$ is completely exsolved during neutralization. Cravotta et al. (1990) argue that some $CO_2$ will dissolve in the ground water and form $H_2CO_3^*$, a weak acid. Therefore, MPA should be computed by multiplying total sulfur, in weight percent, by a factor of 62.5 instead of the traditional factor of 31.25. Use of the 62.5 factor assumes that the total sulfur is pyritic and that 4 moles of $CaCO_3$ are required to neutralize the acidity from 1 mole of pyrite.

Mining practices such as selective handling of coal pit cleanings, removal of acidic material from the mine site, and concurrent reclamation, appear to have enhanced the success of the alkaline addition.

## Summary and Conclusions

In summary, empirical as well as theoretical considerations suggest that alkaline-addition rates are typically inadequate to neutralize AMD. Application rates for alkaline additives are best computed by considering a conservative estimate of MPA, which may be computed by multiplying total sulfur, in weight percent, by a factor of 62.5 (Cravotta et al. 1990; Smith and Brady 1990). Furthermore, the alkaline material may be most effective if incorporated concurrent with mining and backfilling, when and where the acid-production reactions occur. The alkalinities that can be generated from alkaline additives may be insufficient to abate the acidity of severe AMD and the localized production of acidity from reactive pyrite and ferric-sulfate minerals. Adequate alkaline-addition rates that create positive NNP at mine sites containing acidic strata may neutralize acidity and produce alkaline effluent water, which still contains unacceptable concentrations of sulfate, iron, and other metals.

On the basis of this study of the addition of alkaline materials to selected surface coal mines in western Pennsylvania, the following conclusions were made:

1) Previous methods for determining alkaline-addition rates, especially the concept that only one-third the calculated deficiency was necessary, have failed to prevent or abate AMD. Most alkaline addition rates are negligible relative to calculated deficiencies (NNP < 0) and insufficient to prevent or neutralize AMD.

2) The addition of alkaline materials to prevent AMD from surface coal mines may be effective providing that the alkaline-addition rates are sufficient (to offset negative NNP) and the overburden has relatively low-sulfur content. Alkaline materials added to high-sulfur mine spoil, even if sufficient to neutralize acid water, may not reduce concentrations of dissolved iron, manganese, and sulfate.

3) Certain mining practices, such as addition of more alkaline material than required by permit conditions, selective handling of pit cleanings, removal of pyritic material from the mine site, and concurrent reclamation appeared to enhance the effect of alkaline addition on reducing acidity.

4) There is good agreement of signs on post-treatment overburden NNP and median net alkalinity of associated mine-drainage water when overburden MPA is computed by multiplying total sulfur, in weight percent, by 62.5.

5) Additional studies are needed to determine the most beneficial rates of application and placement of the alkaline materials. Calculations of deficiencies (NNP) and application rates should be conservative and consider the theoretical arguments given by Cravotta et al. (1990) and the empirical results of this study.

## Acknowledgments

The authors wish to thank Arthur W. Rose, Jay W. Hawkins, Patricia M. Erickson, Dale W. Blevins, and Andrew C. Ziegler for helpful comments during preparation of this manuscript.

## Literature Cited

Brady, K.B.C., and R.J. Hornberger. 1989. Mine drainage prediction and overburden analysis in Pennsylvania. In Proceedings: West Virginia Surface Mine Drainage Task Force Symposium. Morgantown, WV, 25-26, April, 1989.

Caruccio, F.T., and G. Geidel. 1984. Induced alkaline recharge zones to mitigate acid seeps. p. 43-47. Symposium on Surf. Mining Hydrology, Sedimentology, and Reclamation. Lexington, KY, December, 1984.

Caruccio, F.T., and G. Geidel. 1986. Reclamation strategies as applied at the DLM properties. In Proceedings: 7th Annual West Virginia Surface Mine Drainage Task Force Symposium, Morgantown, WV, April, 1986.

Cravotta, C.A. III, K.B.C. Brady, M.W. Smith, and R.L. Beam. 1990. Effectiveness of alkaline addition at surface coal mines in preventing and abating acid mine drainage: Part 1. Geochemical considerations. In Proceedings: The 1990 Mining and Reclamation Conference and Exhibition. Charleston, WV, 23-26, April, 1990.

diPretoro, R.S. 1986. Premining Prediction of Acid Drainage Potential for Surface Coal Mines in Northern West Virginia. M.S. Thesis, West Virginia University, Morgantown, WV.

diPretoro, R.S. and H.W. Rauch. 1988. Use of acid-base accounts in premining prediction of acid drainage potential. p. 2-10. In Proceedings: Mine Drainage and Surface Mine Reclamation, Vol. 1: Mine Water and Mine Waste. Pittsburgh, PA, 19-21 April, 1988. U.S. Bur. of Mines Inform. Circ. 9183.

Erickson, P.M., and R. Hedin. 1988. Evaluation of overburden analytical methods as means to predict post-mining coal mine drainage quality. p. 11-19. In Proceedings: Mine Drainage and Surface Mine Reclamation, Vol. 1: Mine Water and Mine Waste. Pittsburgh, PA, 19-21 April, 1988. U.S. Bur of Mines Inform. Circ. 9183.

Ferguson, Keith, and P.M. Erickson. 1988. Approaching the AMD problem - from prediction and early detection. In Proceedings: International Conference on Control of Environmental Problems from Metals Mines. Roros, Norway, June 20-24, 1988.

Geidel, G. 1982. An Evaluation of a Surface Application of Limestone for Controlling Acid Mine Discharges from Abandoned Strip Mines, Sewellsville, Ohio. Ph.D. Dissertation, Univ. of South Carolina, Columbia, S.C.

Helsel, D.R. 1987. Advantages of nonparametric procedures for analysis of water quality data. Hydrological Sciences Journal 32(2):179-190.

Hem, J.D. 1985. Study and interpretation of the chemical characteristics of natural water (3d). U.S. Geological Survey Water Supply Paper 2254, 263 p.

Hettmansperger, T.P., and S.J. Sheather. 1986. Confidence intervals based on interpolated order statistics. Statistics and Probability Letters, 4(2):75-79.

Lusardi, P.J., and P.M. Erickson. 1985. Assessment and reclamation of an abandoned acid-producing strip mine in northern Clarion County, Pennsylvania. p. 313-321. In Proceedings: 1985 Symposium Surface Mining Hydrology, Sedimentology, and Reclamation, Univ. of Kentucky, Lexington, KY.

Morrison, J.L., S.D. Atkinson, A. Davis, and B.E. Sheetz. 1990. The use of $CO_2$ coulometry in differentiating and quantifying the carbonate phases in the coal-bearing strata of western Pennsylvania: its applicability in interpreting and modifying neutralization potential. In Proceedings: The 1990 Mining and Reclamation Conference and exhibition. Charleston, WV, 23-26, April, 1990.

Skousen, J.G., J.C. Sencindiver, and R.M. Smith. 1987. A Review of Procedures for Surface Mining and Reclamation in Areas with Acid-Producing Materials. W. Va. Surf. Mine Drainage Task Force, 39 p.

Smith, M.W., and K.B.C. Brady. 1990. Review and summary of acid base accounting data using computer spreadsheets. In Proceedings: The 1990 Mining and Reclamation Conference and Exhibibition. Charleston, WV, 23-26, April, 1990.

Sobek, A.A., W.A. Schuller, J.R. Freeman, and R.M. Smith. 1978. Field and Laboratory Methods Applicable to Overburden and Minesoils. U.S. E.P.A. Report EPA-600/2-78-054.

Stumm, Werner, and J.J. Morgan. 1981. Aquatic chemistry--an introduction emphasizing chemical equilibria in natural waters (2nd): New York, John Wiley and Sons, 780 p.

West Virginia Surface Mine Drainage Task Force. 1979. Suggested Guidelines for Method of Operation in Surface Mining of Areas With Potentially Acid-Producing Materials. Green Lands, 9:21-40.

Velleman, P.F., and D. Hoaglin. 1981. Applications, Basics, and Computing of Exploratory Data Analysis. Duxbury Press, Boston, MA, 354 p.

Waddell, R.K., R.R. Parizek, and D.R. Buss. 1980. The Application of Limestone and Lime Dust in the Abatement of Acidic Drainage in Centre County, Pennsylvania. Final Report Research Project 73-9. Commonwealth of Pennsylvania, Department of Transportation, Office of Research and Special Studies, 245 p.

Williams, J.H., J.H. Henke, K.L. Pattison, R.R. Parizek, R.J. Hornberger, and C.A. Cravotta III. (in press). Hydrogeology and water quality at a surface coal mine in Clarion County, Pennsylvania: Pennsylvania State University Coal Research Report.

This page intentionally left blank.

# THE USE OF $CO_2$ COULOMETRY IN DIFFERENTIATING AND QUANTIFYING THE CARBONATE PHASES IN THE COAL-BEARING STRATA OF WESTERN PENNSYLVANIA: ITS APPLICABILITY IN INTERPRETING AND MODIFYING NEUTRALIZATION POTENTIAL (NP) MEASUREMENTS[1]

by

J.L. Morrison, S.D. Atkinson, Alan Davis, and B.E. Scheetz[2]

Abstract. $CO_2$ coulometry is a rapid, highly accurate technique which can be used to determine the carbonate carbon contents of coals and associated overburdens. The rate at which the $CO_2$ is evolved during the acid digestion process is a function of which carbonate phases are present. Calcite and dolomite reaction times are approximately six to eight minutes under the conditions described herein, with the maximum change in reaction rates occurring between two and four minutes. Siderite requires a lengthy 10 to 15 minutes to completely react with the maximum change in reaction rate occurring between six and eight minutes. The determination of neutralization potential (NP) in the coal-bearing overburdens of the eastern United States is inaccurate when siderite is present because of the slow conversion of $Fe^{2+}$ to $Fe^{3+}$. To alleviate the effects of siderite on NP, a 30 wt.% $H_2O_2$ solution can be used to accelerate the conversion of $Fe^{2+}$ to $Fe^{3+}$ during the titration process. The addition of $H_2O_2$ increases both the accuracy and precision of the NP determination. Despite the addition of $H_2O_2$, many siderite-bearing overburdens still possess NP. This is interpreted as being the result of: 1) the incomplete oxidation of $Fe^{2+}$ to $Fe^{3+}$ and the incomplete precipitation of iron hydroxide, and/or 2) the presence of Ca and/or Mg occurring in solid solution in the siderite.

Additional key words: fizz rating, hydrogen peroxide, ferrous and ferric iron, carbonate carbon, $CO_2$ coulometry, x-ray diffraction, microprobe analysis.

## Introduction

Prediction of acidic mine drainage is based in part on the ability to accurately characterize overburdens with respect to sulfur and carbonate. The acid/base accounting method is the most frequently used type of overburden characterization. The acid/base accounting method is composed of two determinations: 1) total sulfur, and 2) neutralization potential. Total sulfur contents are then multiplied by 31.25 to calculate potential acidity. Total sulfur determinations are relatively rapid and accurate. Neutralization potential is determined by digesting the overburden in HCl, back titrating with NaOH to an endpoint of pH = 7 which is held for 30 seconds (Noll, Bergstresser, and Woodcock 1988). This determination was originally used to find the neutralization equivalence of limestone (Jackson 1964). When applied to determining the neutralization potential of overburdens, this method is less accurate when siderite is present. Siderite, a common constituent in overburdens, is believed not to have the

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]J.L. Morrison, and S.D. Atkinson are research assistants and Dr. B.E. Scheetz an Associate Professor of Solid State Science at the Materials Research Laboratory. Dr. Alan Davis is the Director of the Energy and Fuels Research Center. All of the authors hold faculty positions at The Pennsylvania State University.

same neutralization capabilities as other carbonates (Williams et al. 1982; Palmer 1973). Siderite ($FeCO_3$) is at least partially digested in the HCl digestion step of the NP determination and will ultimately affect the stability of the titration endpoint because of the slow and incomplete conversion of $Fe^{2+}$ to $Fe^{3+}$. This instability affects both the accuracy and precision of the NP determination.

Because the differentiation of the various carbonate phases is essential in the prediction of acidic mine drainages, several studies have attempted to differentiate and quantify the various carbonate phases present in coal-bearing strata. The most conventional method used to ascertain variations in carbonate mineralogy is quantitative x-ray diffraction (QXRD) (Despard 1988; Morrison 1988). The primary difficulty with this technique is the lower detection limit of one to two percent. Less conventional methods of carbonate characterization include evolved gas analysis (Hammack 1987) and automated pressure cells (Evangelou, Roberts, and Szekeres 1985).

This paper will present an alternative method of carbonate characterization known as $CO_2$ coulometry (Huffman 1977). $CO_2$ coulometry is an analytical technique which is routinely used to determine carbon (both total and carbonate). It is a two-step process in which total carbon is determined by measuring the amount of $CO_2$ evolved when the sample is heated to 1000°C. Carbonate carbon is determined by digesting the sample in hot perchloric acid and measuring the amount of $CO_2$ evolved during the digestion. Organic carbon is then determined by difference.

Because calcite, dolomite, and siderite exhibit different rates of dissolution in acids (Evangelou, Roberts, and Szekeres 1985) the $CO_2$ coulometer was used to measure the evolution of $CO_2$ with respect to time. To augment the $CO_2$ coulometry data, x-ray diffraction was used for carbonate phase identification.

## Materials and Methods

$CO_2$ coulometry is a relatively rapid analytical technique which can accurately determine the carbonate carbon content of a given sample. The schematic of the $CO_2$ coulometer apparatus is shown in Figure 1.

The overburden sample was accurately weighed (20 to 25 mg) in a ceramic weighing boat and then placed in the sample tube housing shown in Figure 1. The side of the sample housing was rinsed with one mL of deionized water to ensure that any particles adhering to the side of the sample housing would be rinsed down to the zone in which the acid will be introduced. This rinse was kept constant at one mL to prevent creating subtle variations in the normality of the perchloric acid which might have an impact on the rate of digestion. The sample housing was then closed and the system degassed for approximately two minutes to remove the atmospheric $CO_2$ introduced into the system when the sample was placed in the sample housing. After degassing, the analysis is commenced by the addition of 2 mL of 2 N perchloric acid. The $CO_2$ evolved during the hot perchloric acid digestion is carried to the $CO_2$ coulometer cell where it is electrochemically titrated. Results are expressed as wt% carbonate carbon and reported to the nearest 0.01%.

Calcite, dolomite, and siderite standards were analyzed on the $CO_2$ coulometer. Calcite and dolomite standards were provided by Bellefonte Lime of Bellefonte, Pennsylvania and are reported as being 97.73% and 98.85% in purity, respectively. Sample (195) is a high quality limestone (Valentine member of the Linden Hall Fm. collected from Bellefonte, PA) and sample (196) is a high quality dolomite (Ledger dolomite member collected in southeast PA near Devault, PA) in which

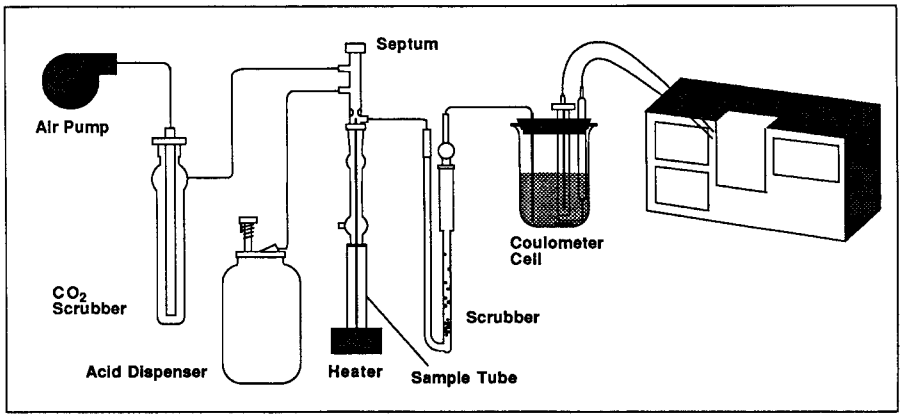



Figure 1: Schematic of $CO_2$ coulometer.

the atomic ratio of Ca/Mg is nearly one. Siderite standards are more difficult to obtain. The siderite standard (197) is an in-house standard.

## Results and Discussions

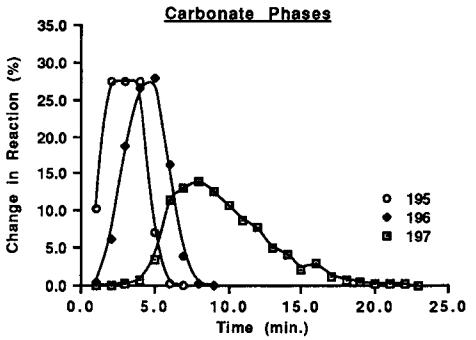
Figure 2 summarizes the $CO_2$ coulometry profiles and the measured NP values for the various standards described above. When viewing the $CO_2$ coulometry profiles in Figure 2, the following observations should be noted: 1) the time for 100% reaction, 2) the time at which the maximum change in reaction occurs, and 3) the overall geometry of the profile. The calcite (195) and dolomite (196) profiles are characterized by relatively short reaction times over which the rate of the reaction is relatively rapid. In contrast to this, the siderite (197) profile is characterized by a much longer reaction time and a more sluggish reaction rate as evidenced by a much broader profile. It is important to note that siderite did not generate any $CO_2$ until the two to three minute reading.

Figure 3 illustrates the $CO_2$ coulometry profiles for three overburden samples with wide ranging NP's, fizz ratings, carbonate carbon contents, and lithologies.

Figure 3A is a $CO_2$ coulometry profile for a dark gray shale and appears to be very similar to the siderite standard. This profile is characterized as being relatively broad with the maximum rate of reaction occurring at the six minute mark. In addition, $CO_2$ is undetectable until the two to three minute mark. X-ray diffraction analysis found siderite to be the only detectable carbonate phase. Multiplying the carbonate carbon content (1.30) by $FeCO_3/C$ ratio determines there is 12.54% siderite present.

In Figure 3B, the $CO_2$ coulometry profile for a siderite concretion (sample 71) is illustrated. Again, the $CO_2$ profile is relatively broad and the rate of reaction is slow. Unlike the previous $CO_2$ profile, however, there is the indication of a slight reaction having occurred at the one to two time interval. X-ray diffraction analysis found siderite to be the only detectable carbonate phase present. Microprobe analysis of this sample determined 3.52 +/- 0.93 wt% Ca present in the siderite (Morrison, Atkinson, and Scheetz 1990). This presence of Ca in the siderite may explain the slight reaction encountered during the one to two minute reaction time. Assuming that all the carbonate present is in the siderite phase, the carbonate content of 7.11 translates into 68.58% siderite.

In Figure 3C, the $CO_2$ coulometry profile for an Upper Freeport limestone is presented. This profile is very different from the previous two profiles (173 and 71). This profile is characterized by a rapid reaction rate but requires a relatively long time (13 mins.) to react completely. Although the two minute mark




| Sample | Description | Carbonate Carbon | Theoretical Carbonate Carbon for Pure Sample | Time (min) for 100% Reaction | Neutralization Potential |
|---|---|---|---|---|---|
| 195 | High Calcium Ls | 11.885 | 12.000 | 7 | 971.58 |
| 196 | Dolomite | 12.952 | 13.026 | 9 | 1054.66 |
| 197 | Siderite | 10.573 | 10.367 | 23 | 37.10 |

Figure 2: $CO_2$ coulometry profiles for calcite, dolomite, and siderite standards.

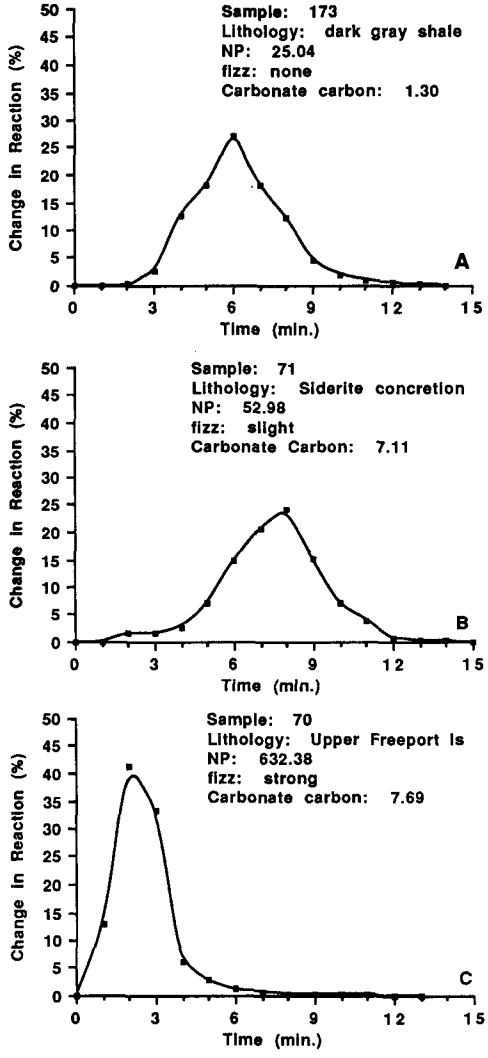



Figure 3: $CO_2$ coulometry profiles for selected overburden samples.

for reaction time is similar to that of calcite, the lengthy time for 100% reaction indicates that other carbonate phases are present. X-ray diffraction identified abundant calcite and ankerite. The presence of ankerite is interpreted as being responsible for the lengthy reaction time. Despite this, multiplying the carbonate carbon (7.69) by the $CaCO_3/C$ ratio indicates there is 64.08% calcite present which translates to a NP of 640.08 which is in very good agreement with the actual measured NP of 632.38.

The primary difficulty in determining the neutralization potential of overburdens is that the presence of siderite affects both the accuracy and the precision of the method. Because siderite is very common in the coal-bearing strata of the eastern United States, NP values are often questioned. This is particularly true if the NP is accompanied by an assigned fizz rating of zero (none). NP's which have fizz ratings of zero are often discounted because they are interpreted as being attributable to the presence of siderite. If the overburden sample exhibits a fizz of one or more (e.g. 71), there is no criterion by which one can interpret the NP as being valid. In the case of overburden sample (71), the dominant carbonate phase is siderite, yet it produced an NP of 52.98. $CO_2$ coulometry can very accurately differentiate calcite from siderite. This method also can determine if the carbonate is present as a mixture by a bimodal $CO_2$ coulometry profile; however, the method cannot yet accurately quantify individual components of a multiple phase system.

Because $CO_2$ coulometry can accurately differentiate between calcite-bearing and siderite-bearing overburdens, the NP determination methodology has been modified in an attempt to provide siderite-free NP's. The modification is very simple. Because siderite is at least partially digested in the NP digestion step, a large amount of $Fe^{2+}$ can potentially go into solution. During the titration process, the $Fe^{2+}$ is partially oxidized to $Fe^{3+}$; however, this reaction does not necessarily go to completion over the short time span of the determination. Because the precipitation of ferric hydroxide generates $H^+$ it is necessary to accelerate the oxidation of $Fe^{2+}$ to $Fe^{3+}$ which will eventually precipitate as a ferric hydroxide (ferrihydrite) during the NaOH titration step. In order to speed up the Fe oxidation, a 30 wt% $H_2O_2$ solution is added. The overburden sample is digested as in the conventional NP determination; however, after cooling the leachate is filtered, five mL of 30 wt.% $H_2O_2$ is added and then reboiled for three to five minutes. After cooling, the leachate is titrated using the conventional NP guidelines.

Table 1 summarizes some of the results. The NP's of samples (173) and (171) are slightly different from those reported in Morrison, Atkinson, and Scheetz (1990) because they were redetermined from different splits. In the first set of data (173, 71, and 63) the addition of $H_2O_2$ had a profound impact on the NP value. In these samples, where siderite is present, the NP's were lowered by 54.07, 38.20, and 13.90%, respectively. In overburden sample (63), the NP value was not as affected as in samples (173) and (71) because calcite was the dominant carbonate phase.

The second set of data in Table 1, sample (171), focuses on the problem of incorrectly assigned fizz ratings. Because determining fizz ratings is very subjective, the addition of $H_2O_2$ is warranted. Overburden sample (171) is a siderite-bearing black to dark gray shale. Three sets of analyses were run, all in triplicate. In the first sequence, the sample was run according to the correct conventional NP determination. In the second sequence, an incorrect fizz rating was intentionally assigned to determine to

Table 1. Comparison of conventional and modified NP determinations among selected overburden samples.

| Sample | Fizz | Carbonate Mineralogy | NP | $NP_{hydrogen\ peroxide}$ |
|---|---|---|---|---|
| 173 | 0 | Siderite (12.54%) | 25.04 | 11.50 |
| 71 | 1 | Siderite (68.58%) Calcite ? | 52.98 | 32.73 |
| 63 | 1 | Calcite, siderite | 65.59 | 56.47 |
| 171 | 0 | Siderite (6.63%) | 14.41 ± 1.47 | - |
| 171 | 1 | Siderite (6.63%) | 23.51 ± 1.03 | 10.25 ± 0.13 |

what extent a larger volume of HCl would effect the NP value when using the conventional NP methodology. In the third sequence, an incorrect fizz rating was intentionally assigned; however, this time the modified NP determination was used. All of these determinations were performed manually, as opposed to using an automated titration system. Comparing these runs clearly indicates that an incorrectly assigned fizz rating, and therefore digestion procedure, can have a profound impact on the resulting NP. By adding $H_2O_2$, the error is lessened and the precision of the method increased.

Despite the addition of five mL of 30 wt.% $H_2O_2$, the NP's of overburdens in which siderite was the only identifiable carbonate phase, did not go to zero. The explanation for this may be two-fold.

The first explanation is that not all of the $Fe^{2+}$ has oxidized to $Fe^{3+}$ and has not precipitated as ferric hydroxide. Because this reaction has not gone to completion, erroneously high NP values could be reported.

The second explanation is more complex and requires further research to substantiate. Siderite is rarely phase pure in nature (Mozley 1989). Microprobe analysis presented in Morrison, Atkinson, and Scheetz (1990) has documented the occurrence of Ca and Mg present in siderites as well as Mn. If Ca and/or Mg is present in significant amounts it may explain why certain siderite-bearing overburdens still possess NP despite the addition of $H_2O_2$. The relationship between the field neutralization of acidic drainages by Ca-bearing siderites has yet to be documented.

## Conclusions

The characterization of the carbonates present in the coal-bearing strata of the eastern United States using the conventional NP determination can be inaccurate. This inaccuracy is due to the presence of siderite which affects the stability of the NaOH titration endpoint. Conclusions of the research completed thus far include: 1) siderite is ubiquitous in the coal-bearing strata examined thus far; 2) the presence of siderite can profoundly effect the NP value; 3) incorrectly assigned fizz ratings, and thus digestion procedures, can affect the accuracy of the NP determinations; 4) $CO_2$ coulometry can be used to very accurately determine the carbonate carbon contents of overburdens; 5) the $CO_2$ evolution rate during the carbonate carbon determination is a function of which carbonate phases are present, and these rates can be used to determine if the overburden contains calcite, siderite, or a mixture of carbonate phases; 6) the effect of siderite on the determination of NP can be minimized using $H_2O_2$; 7) because siderite occurs in overburdens which contain reactive carbonate phases (e.g. calcite and/or dolomite), the addition of $H_2O_2$ is warranted because there is no criterion to judge the validity of an NP whose assigned fizz rating is greater than zero; 8) the observed siderites were not phase pure, an observation in good agreement with the literature; and, 9) despite the addition of $H_2O_2$ many siderite bearing overburdens still possess NP. This may be attributable to the pH of the NaOH back titration endpoint (pH = 7.0) and/or the presence of Ca and Mg in the siderite.

## Acknowledgements

The authors would like to acknowledge the Pennsylvania Energy Development Authority for supporting this research under Grant #ME 486-043. Joe Bodkin provided assistance in the $CO_2$ coulometry determinations.

## Literature Cited


Despard, T.L. 1988. X-Ray Diffraction Evaluation of Coal Overburden Neutralization Potential, In: Bureau of Mines Information Circular 9183, United States Department of the Interior, 413 p.

Evangelou, V.P., K. Roberts, and G.W. Szekeres. 1985. The Use of an Automated Apparatus for Determining Coal Spoil Carbonate Types and Reactivity, p. 163-165. In: Proceedings of the 1985 National Symposium on Surface Mining, Hydrology, Sedimentology, and Reclamation. (Lexington, KY, Dec. 8 - 13, 1985).

Hammack, R.W. 1987. Evolved Gas Analysis - A New Method For Determining Pyrite, Bicarbonate, and Alkaline Earth Carbonates. In: Eighth Annual West Virginia Task Force Symposium, Morgantown, West Virginia, West Virginia Surface Mine Drainage Task Force and West Virginia Mining and Reclamation Association.

Huffman, E.W.D. 1977. Performance of a New Automatic Carbon Dioxide Coulometer: Microchemical Journal, 22: 567-573.

Jackson, M.L. 1964. Soil Chemical Analysis: Prentice-Hall, Englewood Cliffs, NJ, 496 p.

Morrison, J.L. 1988. A Study of Factors Controlling the Severity of Acid Mine Drainage in the Allegheny Group of Western Pennsylvania [M.S. thesis]: University Park, The Pennsylvania State University, 145 p.

Morrison, J.L., S.D. Atkinson, and B.E. Scheetz. 1990. Delineation of Potential Manganese Sources In the Coal Overburdens of Western Pennsylvania. In Proceedings of the 1990 Mining and Reclamation Conference and Exhibit.

Mozley, P.S. 1989. Relation Between Depositional Environment and the Elemental Composition of Early Diagenetic Siderite: Geology, 17: 704-706.

Noll, D.A., T.W. Bergstresser, and J. Woodcock. 1988. Overburden Sampling and Testing Manual: Commonwealth of Pennsylvania, 78 p.

Palmer, M.E. 1978. Acidity and Nutrient Availability in Colliery Spoil, p. 85-126. In: Goodman, G.T., and M.J. Chadwick, eds., Environmental Management of Mineral Wastes: Alphen aan den Rijn-The Netherlands, Sijthoff and Noordhoff.

Williams, E.G., A.W., Rose, R.R. Parizek, and S.A. Waters. 1982. Factors Controlling the Generation of Acid Mine Drainage: Final Report on Grant G5105086, Pennsylvania Mining and Mineral Resources Research Institute, 265 p.

# DELINEATION OF POTENTIAL MANGANESE SOURCES IN THE COAL OVERBURDENS OF WESTERN PENNSYLVANIA[1]

by

J.L. Morrison, S.D. Atkinson, and B.E. Scheetz[2]

Abstract. Manganese is a common and regulated metal in acidic mine effluents. Because the precipitation and removal of manganese from mine effluents is complex and expensive, it is advantageous to ascertain the distribution of manganese in the stratigraphic section prior to mining. By identifying certain stratigraphic horizons that have high amounts of manganese, it may be possible to implement special handling plans to minimize or prevent mine drainages from having elevated manganese contents. Because the source of manganese in mine drainages is poorly documented, the overburden from a mine site having drainages characterized by elevated manganese concentrations (70 to 80 ppm Mn) was examined in detail. Bulk manganese analyses determined the majority of the manganese to be present in a 13.5 foot thick sequence of black to dark gray shales which also contained highly variable amounts of sulfur. $CO_2$ coulometry, and x-ray diffraction determined this stratigraphic interval to be enriched in siderite ($FeCO_3$). Increases in the manganese were accompanied by increases in the siderite content. Manganese substitution in the siderite structure appears to be the source of manganese in this mine site.

Additional key words: Siderite, $CO_2$ coulometry, solid solution, microprobe analyses and bulk Mn determinations.

## Introduction

Because the source of the Mn in coal-bearing strata is poorly understood, predicting the occurrence of Mn in mine drainages is difficult. To gain a better understanding of why certain mine drainages have elevated concentrations of manganese, the overburden from a mine site having an acidic high-Mn drainage was examined using a variety of techniques.

The primary objective of the overburden analysis is to characterize the overburden from this particular mine site using a variety of techniques that will provide more information than the typical acid-base accounting data. Although the source of Mn in coal-bearing strata is poorly documented, the following sources are potential candidates: 1) pyrolusite ($MnO_2$), and/or 2) Mn in solid solution associated with the various carbonate phases (e.g. siderite and/or calcite).

One particular area of interest is the characterization of the various carbonate phases present. The characterization of carbonates involves: 1) determining both the nonreactive carbonate (e.g. siderite) and the reactive carbonate (e.g. calcite and/or dolomite) phases and 2) determining if the presence of Mn is related to the occurrence of carbonates. The occurrence of Mn in solid solution in carbonates is well documented in the literature (Hurlbut

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]J.L. Morrison, and S.D. Atkinson are research assistants and Dr. B.E. Scheetz an Associate Professor of Solid State Science at the Materials Research Laboratory. All of the authors hold faculty positions at The Pennsylvania State University.

and Klein 1977; Pearson 1979; Teodorovich 1961; Mozley 1989; and Pye 1984).

## Materials and Methods

The majority of overburden in this surface mine is characterized by low sulfur ($S_T$ < 0.35 wt. %) and low NP (NP < 20) in which fine to medium grained sandstones are the dominant lithology. The conventional acid/base accounting data (total sulfur, NP, and fizz rating) were obtained from Geochemical Testing of Somerset, PA. $CO_2$ coulometry, bulk Mn determinations, x-ray diffraction, and the simulated weathering experiments were performed at The Pennsylvania State University.

### Total Sulfur

Total sulfur analyses were performed using a Leco SC-32 sulfur analyzer. Vanadium pentoxide was used as an accelerator. This high temperature combustion technique is described in Noll, Bergstresser, and Woodcock (1988). Total sulfur can then be converted to maximum potential acidity by multiplying total sulfur by 31.25.

### Neutralization Potential

Neutralization potential, as the name implies, is used to determine the neutralization ability of a given sample. This entails determining the amount of calcite, and/or dolomite and expressing these contents as tons of $CaCO_3$ equivalent/1000 tons of overburden. The determination of neutralization potential is described in Noll, Bergstresser, and Woodcock (1988).

In theory, siderite does not report in the NP determination and is believed not to have the neutralization ability of calcite or dolomite. As a consequence, siderite may be very abundant in a given overburden, yet will go undetected in the NP determination. As a result of this, $CO_2$ coulometry was used to determine the carbonate content.

### $CO_2$ Coulometry

$CO_2$ coulometry is a highly accurate analytical technique that is routinely used in the determination of both inorganic and carbonate carbon. Because the various carbonate phases react at different rates, the $CO_2$ coulometer was used to measure the rate of $CO_2$ evolution when the sample is introduced to an acid. Because siderite reacts much slower than the other carbonate phases (e.g. calcite and/or dolomite), $CO_2$ coulometry was used to quantify the occurrence of siderite. The methodology employed here is described in detail by Morrison et al. (1990). In theory, this is a very similar approach to Evangelou, Roberts, and Szekeres (1985).

### Bulk Manganese Analyses

Bulk manganese analyses were performed on selected samples to determine the distribution of manganese throughout the stratigraphic sequence. Overburden samples were digested in a HF - $H_2SO_4$ solution, until only a minimal residue remained. The solution was boiled to drive off the volatile silica. The residue from the HF - $H_2SO_4$ leach was then introduced to a $HNO_3$ leach, to ensure complete digestion. Manganese, if present in any form, (e.g. pyrolusite and/or Mn in solid-solution) would leach out under these various acid leaches. The combined leachates from these digestions were analyzed for manganese using a Spectrametrics Spectra Span III atomic emission spectrometer. The excitation source was a DC plasma sustained in argon gas. Liquid aliquots were serially diluted to the linear range of 1 to 5 ppm. All unknowns were analyzed by first running a series of reference solutions and then the unknowns. This procedure was then repeated in reverse order to account for instrumental drift. The duplicate values were averaged and these averages reported. When operated in this manner, the experimental detection limits were estimated to be 0.01 ppm (10 ppb).

### Simulated Weathering Experiments

Simulated weathering experiments are a useful technique to compare overburden samples under controlled laboratory conditions. Their usefulness in predicting the field occurrence of acidic mine drainage is more problematic. Overburden samples were prepared by stage crushing to -1/4 inch. The samples were placed in one quart HDPE plastic storage containers (Figure 1). Deionized water was added to the plastic leaching vessels and allowed to react in an enclosed chamber in which humidified air was circulated. Leachates were collected on a weekly basis (hourly and weekly leachates combined) and characterized with respect to pH, acidity, sulfate, total iron, manganese, and aluminum.

### X-Ray Diffraction

Overburden samples were examined for their crystalline phase composition using conventional x-ray powder diffraction methods. All data were collected with copper K-alpha radiation on a fully automated Scintag PAD V diffractometer equipped with a cryogenic detector. All specimens were ground to -325 mesh and mounted on a zero background substrate cut from a single crystal of quartz which was oriented just off the c-axis. The range of data collection was from 15 to 55 degrees 2-theta at 1 degree/minute and a step size of 0.02 degrees. Phase identifications were made by comparison to the JCPDS-ICDD reference patterns. The lower detection limit is estimated to be approximately 1%.

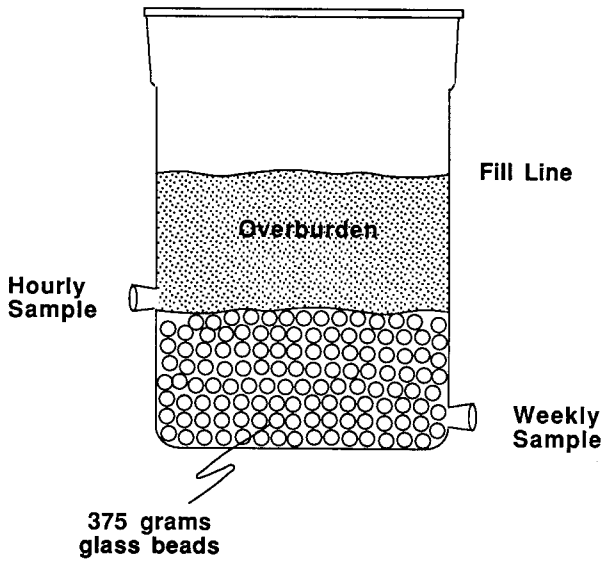



Figure 1: Simulated Weathering Vessles.

ETEC Microprobe

Overburden specimens were processed to -20 mesh and then pelletized. The -20 mesh particles were mixed with a Hexacol epolite resin and hardener, pressed into one inch diameter stainless steel molds at 10,000 psi for two minutes. After the two minute press, the resin was allowed to cure overnight. These pellets were polished with 400 and 600 grit paper for three minutes each. The next phase of polishing involved using 0.3 micron aluminum slurry on a Texmet cloth for two minutes, followed by the final polish using a 0.5 micron aluminum slurry on a silk polishing cloth. The samples were washed in an ultrasonic bath between all polishing steps. The polished pellets were wiped dry and analyzed within 18 to 24 hours.

An ETEC Auto Microprobe was employed to analyze the chemical composition of siderite by a combination of energy dispersive and wavelength dispersive techniques. Siderite grains were selected from the overburden fragments based upon the chemical composition determined by using semi-quantitative EDX. The criterion used to establish a grain of siderite versus clays was the presence of significant amounts of silica (> 1.00 wt %). In this case, these grains were not counted due to clay contamination. Grains which contained only traces of silica (< 1.00 wt %) were analyzed for Mg, Al, Ca, Mn, and Fe. This latter category of grains was analyzed individually by wavelength dispersive techniques using LIF, PET, and TAP crystal monochromators. All analyses were run at 15Kv acceleration potential with a sample current of 0.02 microamps.

Results and Discussion

Figure 2 summarizes the stratigraphic and overburden data from one of the core holes on the mine site. Total sulfur contents were generally less than 0.35% except in samples 170, 171, 185, and 189. NP's were typically less than two except for samples 170 to 175 and 183 to 185. The majority of the total sulfur and NP occurred within a 13.5 foot thick sequence of black to dark gray shale/siltstones. The NP from this horizon would typically be considered negligible as a neutralizer by Pa DER (Brady and Hornberger 1989) because of the assigned fizz rating and the low NP's. These NP's would be interpreted as being the result of siderite and therefore discounted because of siderite's inability to neutralize acidic drainages (Williams et al. 1982).

Because NP determinations do not accurately quantify all of the carbonates present (excludes the majority of siderite) carbonate carbon measurements were performed on selected samples. The majority of the carbonate present was associated with samples 170 - 175. $CO_2$ coulometry data, as interpreted and discussed by Morrison et al. (1990) indicated that the dominant carbonate phase present in these overburdens was siderite. This interpretation was also verified using x-ray diffraction.

Figure 3 shows a portion of the x-ray diffractograph for sample 173. The portion of the diffractograph presented here represented the 2 theta interval where the main carbonate peaks would occur if present (27 to 33 degrees 2 theta). The stick figures below the x-ray diffractograph are the JCPDS-ICDD reference patterns for siderite, calcite, and ankerite respectively.

Bulk manganese determinations revealed that the majority of manganese present in the overburden also occurred in this 13.5 foot thick shale sequence. Of particular importance to note here was that increasing carbonate carbon (siderite) contents were accompanied by both increasing amounts of NP and Mn. The increase in NP's with increasing siderite contents exemplifies the shortcomings of the conventional NP determination. As the amount of siderite increased, the accuracy of the NP determination became questionable because of the increased Fe in solution Morrison et al. (1990). The parallel increase in both siderite and Mn strongly suggested that the Mn is associated with the siderite probably occuring in solid solution.

Rosenberg (1960, 1963, 1967) described the subsolidus relationships in the system $CaCO_3$-$MgCO_3$-$FeCO_3$-$MnCO_3$. His data and more recent studies reported by Essene (1983) demonstrated that complete solid solution exists in the $MgCO_3$-$FeCO_3MnCO_3$ system. Figure 4 gives an example of the subsolidus relationships in the $MgCO_3$-$FeCO_3MnCO_3$

| Log Interval | Thick-ness | Scale | Graphic Log | Lithologic Description | OB Analys No. | % Total Sulfur | NP | Fizz Rating | CC | Mn |
|---|---|---|---|---|---|---|---|---|---|---|
| 0-3 | 36" | | | | 153 | 0.01 | 0.54 | none | | 95 |
| 3-6 | 36" | | | Light gray, medium grain sandstone | 154 | 0.00 | 0.26 | none | | 15 |
| 6-9 | 36" | 10 | | | 155 | 0.00 | 0.41 | none | | |
| 9-12 | 36" | | | | 156 | 0.00 | 0.51 | none | | 15 |
| 12-15 | 36" | | | | 157 | 0.00 | 0.28 | none | 0.01 | 15 |
| 15-18 | 36" | | | | 158 | 0.00 | 0.33 | none | | |
| 18-21 | 36" | 20 | | Light gray, medium to fine grained sandstone | 159 | 0.00 | 0.48 | none | | |
| 21-24 | 36" | | | | 160 | 0.00 | 0.66 | none | | |
| 24-27 | 36" | | | | 161 | 0.00 | 1.02 | none | | |
| 27-30 | 36" | 30 | | | 162 | 0.00 | 0.51 | none | | 20 |
| 30-33 | 36" | | | | 163 | 0.00 | 0.43 | none | | |
| 33-36 | 36" | | | | 164 | 0.00 | 0.64 | none | 0.00 | 15 |
| 36-39 | 36" | | | Light gray, medium grained sandstone with increasing amounts of organic debris with depth | 165 | 0.02 | 0.74 | none | | |
| 39-42 | 36" | 40 | | | 166 | 0.01 | 0.56 | none | | |
| 42-45 | 36" | | | | 167 | 0.03 | 0.46 | none | | |
| 45-47'1 | 25" | | | | 168 | 0.01 | 0.54 | none | | |
| 47'1-49'2 | 25" | 50 | | | 169 | 0.15 | 0.26 | none | | |
| 49'2-52'2 | 36" | | | | 170 | 4.42 | 2.81 | none | 0.04 | 220 |
| 52'2-55'2 | 36" | | | Dark-gray shale with increasing siltstone laminate with depth | 171 | 0.66 | 10.20 | none | 0.69 | 1650 |
| 55'2-58'2 | 36" | 60 | | | 172 | 0.20 | 16.43 | none | 1.23 | 2700 |
| 58'2-61'2 | 36" | | | | 173 | 0.26 | 18.29 | none | 1.30 | 3100 |
| 61'2-62'10 | 17" | | | | 174 | 0.32 | 10.25 | none | 0.80 | 2050 |
| 62'10-65'10 | 36" | | | Light gray medium-grain sandstone, | 175 | 0.25 | 4.03 | none | 0.36 | 550 |
| 65'10-68'10 | 36" | 70 | | black shale partings | 176 | 0.24 | 1.91 | none | | 290 |
| 68'10-71'10 | 36" | | | | 177 | 0.09 | -1.32 | none | | |
| 71'10-74'10 | 36" | | | | 178 | 0.02 | 0.20 | none | 0.00 | 30 |
| 74'10-77'10 | 36" | | | | 179 | 0.02 | 0.41 | none | | |
| 77'10-80'10 | 36" | | | | 180 | 0.00 | 0.00 | none | | |
| 80'10-83'10 | 36" | 80 | | Light to whitish gray, medium-grain sandstone with some pyritic nodules, decreasing organic debris with depth | 181 | 0.03 | 0.38 | none | 0.00 | 30 |
| 83-86'10 | 36" | | | | 182 | 0.07 | 0.64 | none | | 30 |
| 86'10-89'10 | 36" | | | | 183 | 0.01 | 3.01 | none | | |
| 89'10-92'5 | 31" | 90 | | | 184 | 0.13 | 13.47 | slight | 0.25 | 270 |
| 92'5-94'5 | 24" | | | Dark gray laminated siltstones | 185 | 0.51 | 2.58 | none | | |
| 94'5-96'5 | 24" | | | | 186 | 0.27 | -2.21 | none | | 240 |
| 96'5-99'5 | 36" | | | Olive to dark gray poorly indurated shale/underclay with plant fragments | 187 | 0.01 | -0.22 | none | 0.01 | 130 |
| 99'5-101'11 | 30" | 100 | | | 188 | 0.35 | 0.87 | none | | 105 |
| 101'11-104'11 | 36" | | | Coal | 189 | 1.50 | 1.38 | none | | |
| 104'11-106'5 | 18" | | | Dark gray underclay | 190 | 0.06 | 1.61 | none | | 115 |
| 106'5-107'10 | 17" | | | Black carbonaceous/boney shale | 191 | 0.23 | 1.35 | none | | |
| 107'-110'4 | 30" | 110 | | | 192 | 0.00 | 0.48 | none | 0.00 | 75 |
| 110'4-112'10 | 30" | | | Dark gray siltstone with small vitrinite bands | 193 | 0.02 | 0.66 | none | | 65 |
| 112'10-115'5 | 31" | | | | 194 | 0.08 | 1.05 | none | 0.01 | 90 |

Figure 2: Overburden Characterization Data.

system showing extensive solid solution between siderite and magnesite and a limited solid solution with Fe in calcite. The pure Fe-analog of dolomite, ankerite, was seen from this data not to exist, but exhibited a partial solid solution between dolomite and ankerite. Limited solid solution was reported (Reeder and Dollase, 1989) in the binary compounds in this system with increased amounts increasing with temperatures. A large three phase area was present in this diagram in which calcite solid solution/ankerite solid solution and siderite-magnesite solid solution coexist. The diagram was representative of the phase relationship at 400°C. All of these studies were reported on experimental or natural samples that were formed at elevated temperatures or pressures above 350°C and 400MPa.

In contrast, literature citations of sedimentary carbonates were restricted mainly to calcite and dolomite with only very limited data reported on siderite (Mozely 1989; Pearson 1979). Mozley (1989) reported on the differentiation between marine and fresh water diagenetic

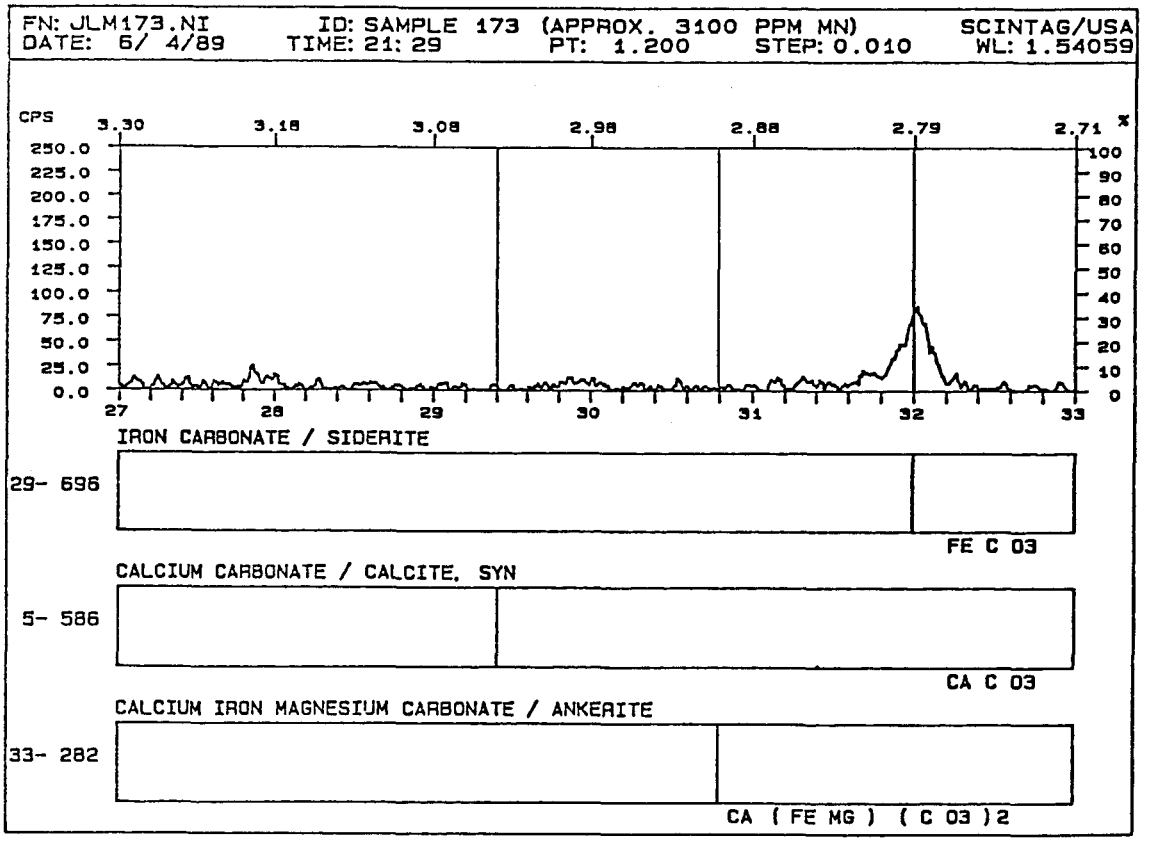



Figure 3: X-ray Diffractograph of Overburden Sample #173.

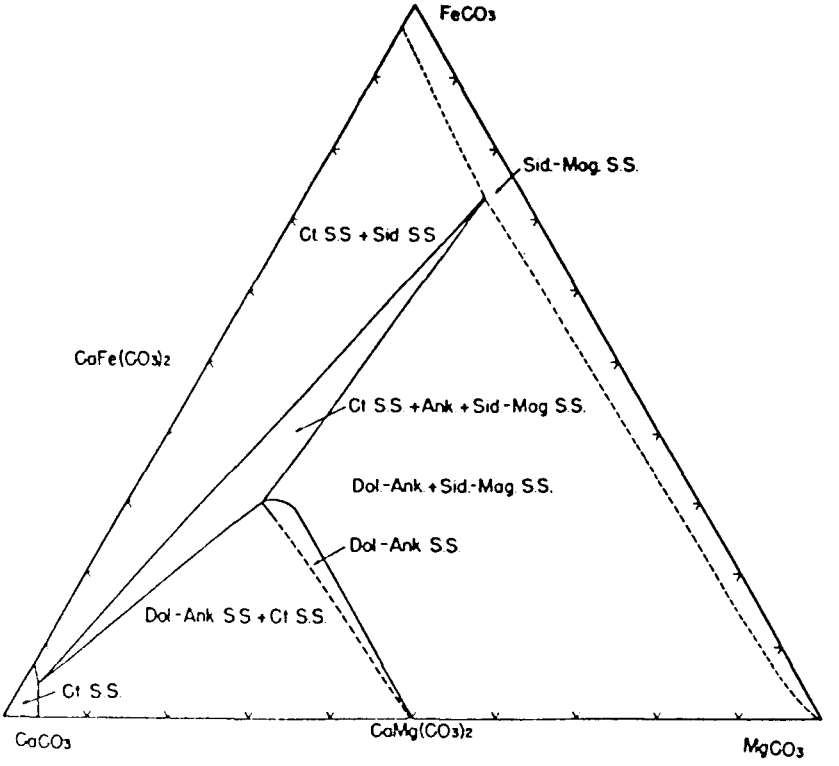



Figure 4: Subsolidus Relationships in the $MgCO_3$-$FeCO_3$-$CaCO_3$ System (after Rosenberg, 1960).

siderites, based upon the crystal chemical substitution of Ca, Mg and Mn for Fe in siderites. The observed degree of substitution in Mozley (1989) was larger at lower temperatures than reported by other authors, perhaps reflecting the differences in depositional environments. This sparcity of information was attributed by Dresel (1989) to the inability to identify fine-grained siderite in the field. As a consequence, very little is known with regards to the degree of solid solutions that form in siderite under these depositional conditions. The subsolidus phase relations of this system at lower temperatures (as in sedimentary siderite) will change from those presented in Figure 4.

Further data to support the observation that Mn is contained in siderite as a crystal chemical substituent was obtained from a siderite concretion (not from this mine site), (Morrison et al. 1990) for which 40 grains of siderite were analyzed for Fe, Mn, Mg and Ca. These data, Table 1, show an average Mn-substitution of 3500 ppm which is equated to a molar substitution of: $(Fe_{.86}Mg_{.02}Ca_{.1}Mn_{.01})CO_3$. Table 2 further supports the presence of a nearly constant, within experimental error, concentration of Mn in this 13.5 foot sequence of black shales. With the exception of the first three foot interval in this section (an interval with high pyrite concentrations) the concentration of Mn in the siderite (carbonate carbon) was relatively constant, showing perhaps a slight trend to higher concentrations with depth.

Table 1: Microprobe Analysis of Siderite (Wt.%)

| | Fe | Mn | Ca | Mg |
|---|---|---|---|---|
| Avg. | 39.05 | 0.35 | 3.52 | 0.38 |
| std. dev. | 1.42 | 0.26 | 0.93 | 0.10 |
| number | 41 | 41 | 41 | 41 |

Simulated weathering experiments were conducted on selected overburden samples within this 13.5 foot sequence of shales. Figure 5 summarizes the simulated weathering data for samples 170, 171, and 175. Sample 171 produced the highest weekly average of Mn among these three samples and also possessed both the highest amount of Mn and siderite in these samples. Mn concentrations in the mine backfill were approximately an order of magnitude greater (ranged from 78 to 84 ppm Mn) than the sample 171 simulated weathering Mn levels.

Table 2: Ratio of Mn to Carbonate Carbon

| Sample # | Carbonate Carbon (wt %) | Mn (ppm) | Mn/C. Carbon Ratio |
|---|---|---|---|
| 170 | 0.04 | 220 | $5.5 \times 10^{-1}$ |
| 171 | 0.69 | 1650 | $2.2 \times 10^{-1}$ |
| 172 | 1.23 | 2700 | $2.4 \times 10^{-1}$ |
| 173 | 1.30 | 3100 | $2.6 \times 10^{-1}$ |
| 174 | 0.80 | 2050 | $2.6 \times 10^{-1}$ |

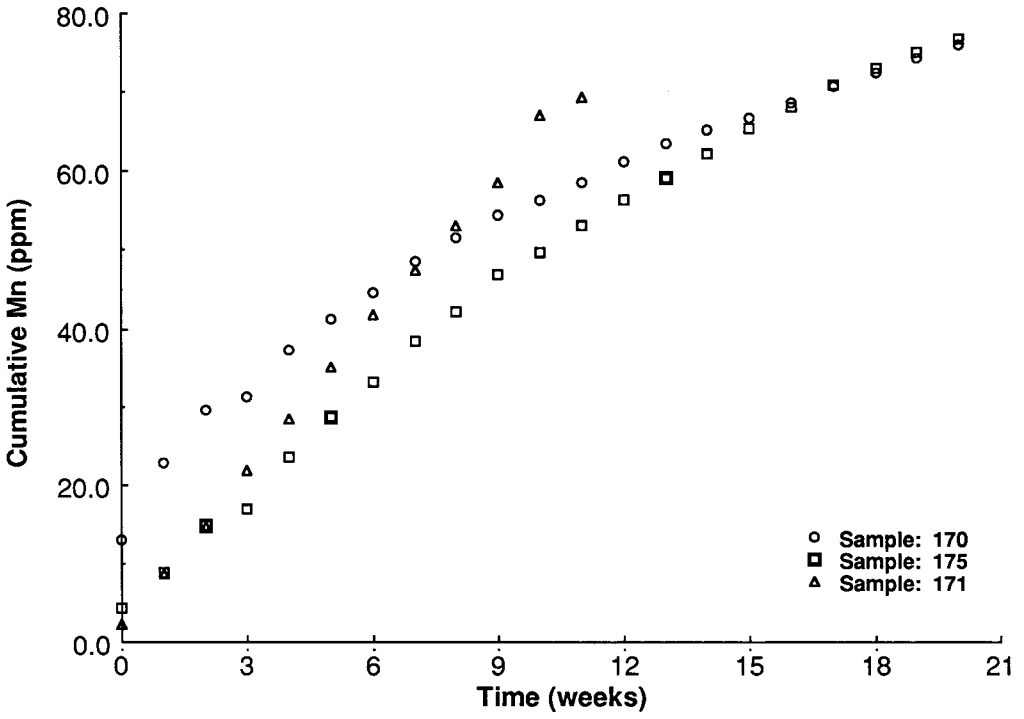
## Conclusions

It is clear that increasing amounts of manganese in this particular mine site, are accompanied with increasing amounts of both carbonate carbon and NP. $CO_2$ coulometry and x-ray diffraction data indicates that this carbonate is solely in the form of siderite. Therefore, at this particular site, the source of the manganese appears to be a Mn-bearing siderite. Of particular importance is that the siderite in this stratigraphic horizon occurs in intimate contact with some of the highest sulfur contents encountered. This accelerates the dissolution of siderite and therefore the release of Mn. Simulated weathering experiments produced leachates with Mn levels approximately an order of magnitude less than those values found in the mine backfill.

The occurrence of Mn in carbonates is well documented. Although the Mn in this particular mine site appears to be in the siderite, ongoing research at The Pennsylvania State University is establishing a Mn database in order to gain a better understanding of the distribution of Mn-bearing phases in coal-bearing stata. Such a database will hopefully lead to determining what level of Mn in the overburden has the potential to produce a mine drainage with elevated levels of Mn.

## Acknowledgements

The authors would like to acknowledge the Pennsylvania Energy Development Authority for supporting this research under Grant #ME 486-043. Lee Eminhizer also proved invaluable in providing assistance in the microprobe analysis.




| Sample | Acid / Base Account Data | | | Carbonate | Bulk Mn | Simulated Weathering | |
|---|---|---|---|---|---|---|---|
| | Total Sulfur (wt %) | Neutralization Potential | Fizz Rating | Carbon | (ppm) | Range Mn (ppm) | Weekly Avg. Mn (ppm) |
| 170 | 4.42 | 2.81 | none | 0.04 | 220 | 1.7-12.9 | 3.4 |
| 171 | 0.66 | 10.20 | none | 0.69 | 1650 | 2.2-7.1 | 5.8 |
| 175 | 0.25 | 4.03 | none | 0.36 | 550 | 1.8-6.0 | 3.5 |

Figure 5: Characterization of Manganese Levels in Simulated Weathering Experiments.

## Literature Cited


Brady, K.B.C. and R.J. Hornberger. 1989. Mine Drainage Prediction and Overburden Analysis in Pennsylvania, In: Proceedings of the 1989 Ninth Annual West Virginia Surface Mine Drainage Task Force Symposium. (Morgantown, WV, April 25-26, 1989).

Dresel, P.E. 1989. The Dissolution Kinetics of Siderite and its Effect on Acid Mine Drainage. PhD thesis. University Park, The Pennsylvania State University, 249 p.

Essene, E.J. 1983. Solid Solutions and Solvi Among Metamorphic Carbonates with Applications to Geologic Thermobarometry. p. 77-96. In Ribbe, P.H. ed., Reviews in Mineralogy, vol 11, Carbonates: Mineralogy and Chemistry. Mineralogical Society of America.

Evangelou, V.P., K. Roberts, and G.W. Szekeres. 1985. The Use of an Automated Apparatus for Determining Coal Spoil Carbonate Types and Reactivity, p. 163-165. In Proceedings of the 1985 National Symposium on Surface Mining, Hydrology, Sedimentology, and Reclamation. (Lexington, KY, Dec. 8-13, 1985).

Hurlbut, C.S. and C. Klein. 1977. Manual of Mineralogy. 19 ed., John Wiley & Sons, New York, 532 p.

Morrison, J.L., et al. 1990, The Use of $CO_2$ Coulometry in Differentiating and Quantifying the Carbonate Phases in the Coal-Bearing Strata of Western Pennsylvania: Its Applicability in Interpreting and Modifying Neutralization (NP) Measurements. In Proceedings of the 1990 Mining and Reclamation Conference and Exhibit.

Mozley, P.S. 1989. Relation Between Depositional Environment and the Elemental Composition of Early Diagenetic Siderite. Geology 17: 704-706.

Noll, D.A., T.S. Bergstresser, J. Woodcock. 1988. Overburden Sampling and Testing Manual. Commonwealth of Pennsylvania, 78 p.

Pearson, M.J. 1979. Geochemistry of the Hepworth Carboniferous Sediment Sequence and Origin of the Diagenetic Iron Minerals and Concretions. Geochimica et Cosmochimica Acta 43: 927-941.

Pye, K. 1984. SEM Analysis of Siderite Cements in Intertidal Marsh Sediments, Norfolk, England. Marine Geology 56: 1-12.

Reeder, R.J., and W.A. Dollase. 1989. Structural Variation in the Dolomite - Ankerite Solid-Solution Series: An X-ray, Mossbauer, and TEM Study. American Mineralogist 74: 1159-1167.

Rosenberg, P.E. 1960. Subsolidus Studies in the System $CaCO_3$-$MgCO_3$-$FeCO_3$-$MnCO_3$. PhD thesis. University Park, The Pennsylvania State University, 137 p.

Rosenberg, P.E. 1963. Subsolidus Relations in the System $CaCO_3$-$FeCO_3$. American Journal of Science 261: 683-690.

Rosenberg, P.E. 1967. Subsolidus relations in the System $CaCO_3$-$MgCO_3$-$FeCO_3$:. American Mineralogist 52: 787-796.

Teodorovich, G.I. 1961. Authigenic Minerals in Sedimentary Rocks. Consultants Bureau, New York, 113 p.

Williams, E.G., A.W., Rose, R.R. Parizek, and S.A. Waters. 1982. Factors Controlling the Generation of Acid Mine Drainage. Final Report on Grant G5105086, Pennsylvania Mining and Mineral Resources Research Institute, 265 p.

# THE EFFECT OF OXYGEN ON PYRITE OXIDATION[1]

By

Richard W. Hammack and George R. Watzlaf[2]

Abstract. The oxygen dependence of abiotic and biotic pyrite oxidation was examined on three scales: 1) pyrite surfaces to a depth of 20 angstroms, 2) 5 g quantities of pyrite in small columns, and 3) 175 kg quantities of pyritic shale in large columns. Results of these studies indicated that the initial abiotic oxidation of fresh pyrite surfaces was independent of oxygen partial pressures above 10% (0-order reaction) and was proportional to oxygen partial pressures below 10% (first-order reaction). However, the rates of abiotic oxidation measured in small columns were proportional to oxygen partial pressures between atmospheric (21%) and 5%. Small column and large column results indicated that with bacteria present, the rate of pyrite oxidation was independent of oxygen partial pressures down to 1%. Below 1% oxygen, the reaction rate was proportional to oxygen partial pressures (first-order reaction). The oxygen-impermeable and oxygen-consumptive barriers other than water that are economically feasible at present are not capable of maintaining oxygen levels below 1%. Therefore, no reduction in the rate of pyrite oxidation would be would be realized using these techniques unless bacteria were inhibited. The best method for limiting pyrite oxidation was to place pyritic material below a permanent water table. Results of large column experiments indicated that a 96% reduction in pyrite oxidation could be realized by inundating the pyritic material. Results of large column tests corroborated previous work by other researchers that indicated leaching frequency had no effect on pyrite oxidation and total sulfate load.

Additional key words: pyrite oxidation, oxygen dependency.

## Introduction

Early laboratory studies (Braley 1960; Clark 1965; Morth and Smith 1966; Rogowski and Pionke 1984; Nicholson, Gillham and Reardon 1988) showed that the rate of pyrite oxidation was dependent on the partial pressure of oxygen. However, there was no general concensus among researchers: reaction orders of 1/2, 2/3, 0.81, and 1 were proposed (Nicholson, Gillham and Reardon 1988; Lowson 1988).

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, WV, April 23-26, 1990.

[2]Richard W. Hammack is a geologist and George R. Watzlaf is an environmental engineer at the U.S. Bureau of Mines, Pittsburgh Research Center, Pittsburgh, PA 15236.

Although there was no agreement on reaction order, all studies indicated that the rate of pyrite oxidation could be reduced by limiting oxygen partial pressures. In the field, several strategies including clay capping, plastic liners, and even standard reclamation practices have reduced oxygen partial pressure in spoil and refuse to less than 10 or even 5% (Watzlaf and Erickson 1986; Lusardi and Erickson 1985). However, reducing oxygen partial pressure to these levels did not significantly slow the overall pyrite oxidation rate (Watzlaf and Erickson 1986). A laboratory study (Pugh, Hossner and Dixon 1984) partially explained the discrepancies between field and laboratory results. The study was carried out using framboidal pyrite in an aqueous suspension inoculated with a pure culture of Thiobacillus ferrooxidans. Results of this study indicated that the rate of pyrite oxidation was independent of oxygen partial pressures (0-order) above 8% when bacteria were present. Below 8% oxygen, the pyrite oxidation rate dropped sharply. Based on these results, no reduction in the pyrite oxidation rate would be expected in spoil or refuse piles unless the oxygen partial pressure could be sustained at levels less than 8% or unless bacteria could be inhibited.

Previous studies (Lusardi and Erickson 1985) indicated that the bulk of pyrite oxidation occurred in the unsaturated zone above the water table. Because many published laboratory pyrite oxidation rates (Braley 1960; Clark 1965; Morth and Smith 1966; and Pugh, Hossner, and Dixon 1984) were determined using aqueous pyrite suspensions, these results were not directly applicable to the field. This study was intended to be a comprehensive evaluation of abiotic and biotic pyrite oxidation rates under unsaturated conditions with different oxygen partial pressures. Pyrite oxidation was examined on three different scales: (1) pyrite surfaces to a depth of 20 angstroms; (2) 5 g quantities of pyrite in small columns; and, (3) 175 kg quantities of pyritic shale in large columns. The rationale of this approach was to determine the oxygen dependency of pyrite oxidation in the simplest case (ie. the oxidation of a surface) and observe apparent changes in oxygen dependency with increasing system complexity. Use of this approach could enable an interpolation to field scale.

## Methods and Materials

### Materials

The pyrite used in surface and small column studies was hand picked from a sample of the Pittsburgh coalbed collected in Barbour County, WV. The pyritic material was crushed to -60 mesh and subjected to a density separation using tetrabromoethane (2.96 g/cc). The sink portion was washed in boiling 6N HCl to remove sulfate and iron oxides and to kill bacteria. Evolved gas analysis indicated that the pyrite concentrate contained no sulfate sulfur and less than 1% organic sulfur. The material was determined to contain greater than 96% iron disulfide based on total sulfur measurements by combustion furnace.

The material used for large column experiments was a dark gray pyritic shale, superjacent to the Clarion coalbed in Clearfield County Pennsylvania. The shale was crushed to pass 3.8 cm and loaded into four columns containing approximately 175 kg each (Watzlaf and Hammack 1989). The crushed material was analyzed by the combustion furnace method and found to contain 3.5% sulfur.

### Surface Weathering Studies

X-ray photoelectron spectroscopy (XPS), a surface analysis technique, was used to examine weathering pyrite surfaces. Because mass transfer or diffusional effects are minimal during the early stages of pyrite surface oxidation, the measured rates better reflect the kinetics of the electron transfer component of pyrite oxidation.

XPS provided quantitative elemental information with a detection limit of approximately 1% of atoms comprising the outermost monolayers. Different formal oxidation states could be distinguished based on chemical shift information (Brundle 1982). Chemical shifts in the sulfur (2p) electron binding energy (Fig. 1) were used in this study to distinguish between pyritic sulfur (reactant) and sulfate sulfur (product). The XPS analysis depth for pyrite was estimated to be about 2.3 nm based on measured inelastic mean free paths for similar semiconductors (Buckley, Woods, and Wouterlood 1987). This depth corresponded to the thickness of 4.24 pyrite unit cells (0.54175 nm/unit cell).

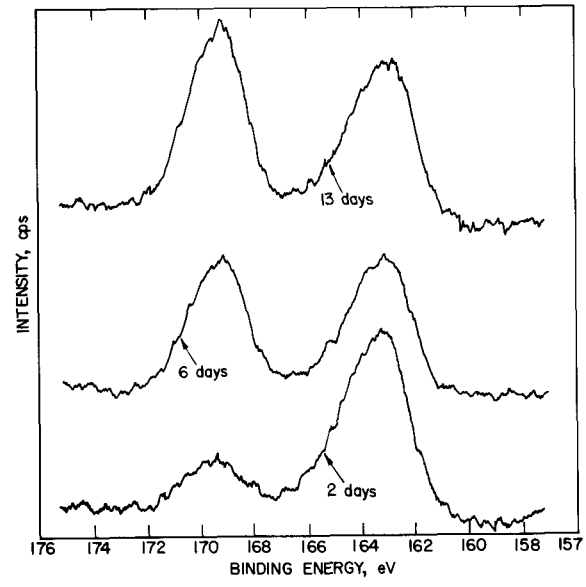



Figure 1. XPS scans of a pyrite surface showing the S (2p) electron region after 2, 6, and 13 days of weathering. Pyritic sulfur and sulfate sulfur peaks occur at 163.5 eV and 169.5 eV, respectively.

The pyrite was weathered at room temperature with a constant humidity of 88% and a oxygen partial pressure of 0.5, 5.0, 10.0, or 21%, balance nitrogen. The conversion of sulfide sulfur to sulfate sulfur was monitored. For each sample, a percent conversion was calculated from the integrated photoelectron intensities for sulfate sulfur (binding energy = 169 eV) and pyritic sulfur (binding energy = 163 eV):

$$\text{Pct. conv.} = \frac{I_{\text{sulfate sulfur}}}{(I_{\text{sulfate sulfur}} + I_{\text{sulfide sulfur}})} \times 100\%$$

where I = integrated photoelectron intensity, arbitrary units.

Because sulfate sulfur and sulfide sulfur were the only observed sulfur species, the equation can be rewritten:

$$\text{Percent conv.} = \frac{\text{I sulfate sulfur}}{\text{I total sulfur}} \times 100 \text{ pct}$$

Immediately prior to weathering, samples for XPS analysis were washed with boiling 4.8 N HCl to remove sulfate, rinsed with methanol, and dried under vacuum. Cylindrical wafers (13 mm diameter by 0.5 mm) were pressed under 700 kg/cm$^2$ pressure. Weathering chambers (Fig. 2) were constructed so that wafers could be removed without disturbing the atmosphere within the chamber. Prior to each experiment, the chambers were washed with an acidified surfactant (sodium lauryl sulfate) solution and then rinsed with methanol to reduce the likelihood that bacterial catalysis could occur.

Periodically during the test, one wafer from each oxygen atmosphere was removed and placed in a Leybold-Heraeus LHS-10 photoelectron spectrometer operated at a pressure of 2X10$^{-8}$ mbar or lower. X-rays from a magnesium anode (Mg K = 1253.6 eV) and an analyzer pass energy of 100 eV were used for the acquisition of S(2p) data. Binding energy calibrations were carried out by adjusting the measured binding energy for the C(1s) spectrum of adventitious carbon to 284.6 eV and shifting all other measured binding energies correspondingly. Peak areas within the S(2p) region attributable to sulfide sulfur and sulfate sulfur were determined with the Leybold-Heraeus DS-5 data system. The percent conversion was calculated for each wafer and plotted versus weathering time.

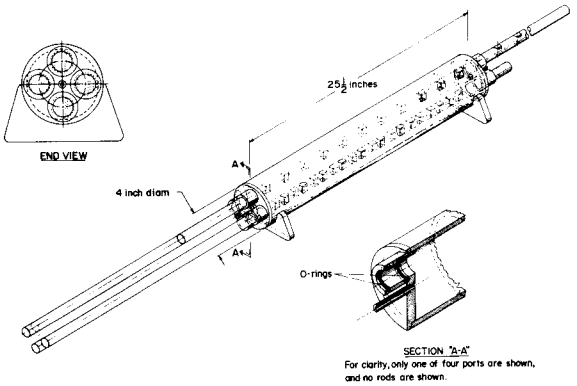



Figure 2. Schematic of chamber used for the XPS-monitored weathering of pyrite wafers. Wafers are placed in rod cut-outs where they can be removed one at a time without disturbing the environment in the chamber.

Small Column Studies

Four 40-cm by 2.54-cm I.D. columns (Fig. 3) were filled with 5 g each of pyrite dispersed in 100 g of 30 - 60 mesh acid-washed silica sand. Glass wool at the base of each column prevented the sample from washing out of the column. The columns were designed to allow pyrite to weather under unsaturated conditions in known oxygen atmospheres. Certified gas mixtures containing 0.005, 0.1, 0.3, 0.5, 1.0, 5.0, 9.6, or 14.5% oxygen, 5.0% carbon dioxide, and balance nitrogen were obtained in compressed gas cylinders. Gas mixtures were

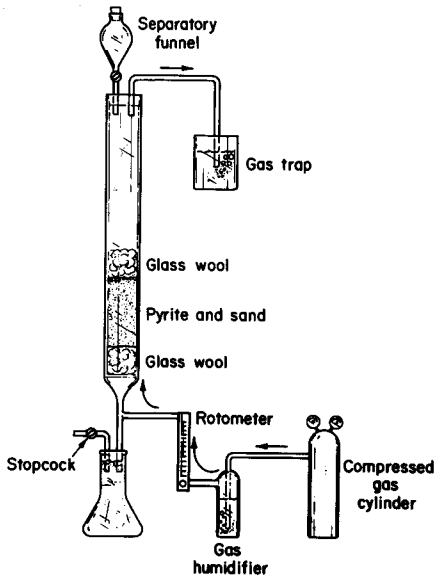



Figure 3. Schematic of set-up used for small column weathering tests.

bubbled through water to humidify the gas stream, passed through a rotometer, and then introduced into the base of each column (Fig. 3). The gases flowed up through the pyritic material and were exhausted to the atmosphere through a gas trap. A flow rate of 20 ml/min was maintained through each column. Analysis of gases exiting the columns showed that this flow rate was sufficient to maintain constant oxygen content throughout the column. The columns were leached weekly with 200 ml of 16 megohm deionized water (controlled oxygen atmospheres were maintained during leaching periods). A most probable number (MPN) type of bacterial enumeration (American Public Health Association 1985; Cobley and Haddock 1975) was performed on each leachate to determine the presence/absence or number of bacteria.

Large Column Studies

The large columns were constructed using 1.92-m lengths of 0.29-m diameter cast acrylic (Fig. 4). Three room temperature experiments were conducted with the large columns: (1) comparison of two applied oxygen partial pressures during unsaturated leaching; (2) comparison of four unsaturated leaching frequencies; and (3) comparison of unsaturated leaching with water-saturated leaching. Deionized water was added through the top of the columns for all leaching and flushing tests. For the first test, the tops of the columns were closed to the atmosphere. Certified gas mixtures containing 1.5% and 10.5% oxygen, balance nitrogen were humidified and flowed through the columns at a rate sufficient to maintain a constant oxygen partial pressure throughout the columns (3.1 L/min). For tests two and three, the top of the columns remained open to the atmosphere. During all unsaturated leaching tests, the valve on the final leachate collection system remained open with the end of the tube kept under water to avoid oxygen inflow. During flushing tests, the columns remained saturated with water with a constant water level maintained at 0.15 m above the surface of the shale. Twenty

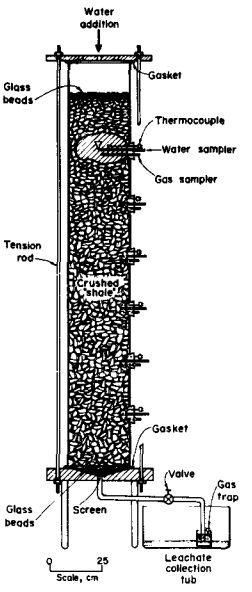



Figure 4. Schematic of set-up used for large column weathering tests.

liters of water were added and withdrawn simultaneously to maintain this water level. MPN bacterial enumerations were performed periodically to insure the presence of viable bacterial populations.

Analytical

The leachates from both the small column and large column tests were analyzed for sulfate using ion chromatography. Gas chromatography was used to determine oxygen in the effluent gases from the columns.

## RESULTS AND DISCUSSION

Surface Studies

The room temperature oxidation of pyrite surfaces was carried out under abiotic conditions at 0.5, 5.0, 10.0, and 21% oxygen and 88% relative humidity. The surface oxidation of pyrite at 21% oxygen is shown in Figure 5. Initial oxidation rates are rapid. The apparent passivation that occurs after 300 hrs of weathering is an artifact of the XPS technique resulting from the mobilization and recrystallization of oxidation products. However, the slope of the least-squares fit line through the initial points is indicative of the oxidation rate. Abiotic surface oxidation rates at 0.5, 5.0, 10.0, and 21% oxygen are shown in Figure 6. Oxidation rates measured by XPS in this study are based on the amount of pyrite at the surface and "available" for reaction and, are significantly faster (20 to 40 times) than rates typically reported. Published oxidation rates are based on bulk pyrite content (amount of sulfate produced or oxygen consumed per gram bulk pyrite per hour) although only a small fraction of the bulk pyrite is at the surface and available for reaction.

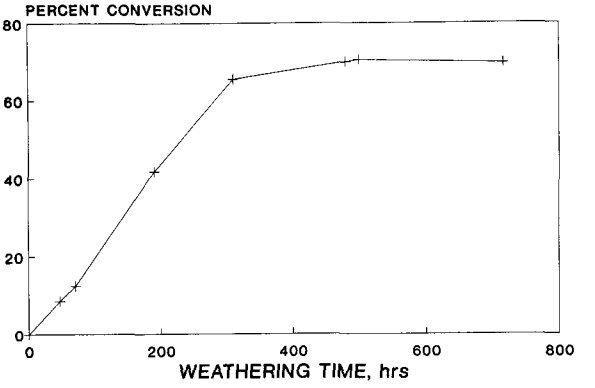



Figure 5. XPS-monitored abiotic weathering of a pyrite surface at 21% oxygen and 88% relative humidity.

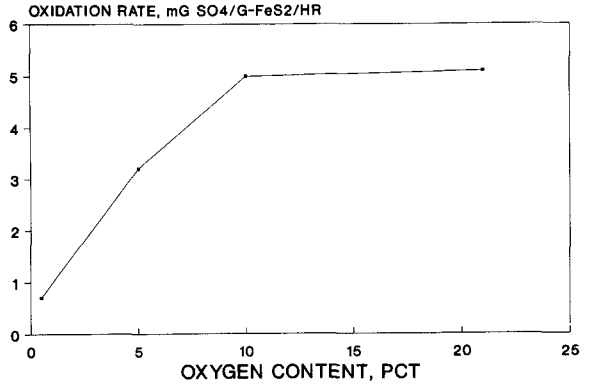



Figure 6. Abiotic surface oxidation rate at different oxygen partial pressures.

In this study, the abiotic surface oxidation rate (Table 1) is found to be independent of oxygen content above 10% (0-th order reaction). Below 10% oxygen, the rates are proportional to the oxygen content (first order reaction).

Table 1. Abiotic Oxidation of Pyrite Surfaces

| Oxygen | Reaction order | Rate equation |
|---|---|---|
| Below 10 pct | 1st | $dSO_4/dt = (0.45$ mg $SO_4/g\text{-}FeS_2/hr/pct\ O_2)(pct\ O_2) + 0.64$ |
| Above 10 pct | 0 | $dSO_4/dt = 5.0$ mg $SO_4/g\text{-}FeS_2/hr$ |

Small Columns

Oxidation without Bacteria. Abiotic oxidation experiments were carried out on HCl-washed pyrite under unsaturated conditions at oxygen contents of 0.5, 5.0, 9.6, and 14.5%. MPN bacterial counts performed on the leachates indicated that the columns were sterile throughout the experiment. Plots of cumulative sulfate in the leachates versus weathering time (Fig. 7) are linear and show that abiotic pyrite oxidation is highly dependent on the oxygen content of the column atmosphere. Oxidation rates were calculated from the slope of the least-squares line fit to cumulative sulfate plots. Abiotic oxidation rates at different oxygen contents are shown in Figure 8 (without bacteria) and Table 2. The abiotic oxidation of pyrite at small-column scale is 1st order above 5% oxygen. Below 5% oxygen, the reaction order appears to fall below 1 which agrees with published results (Nicholson, Gillham, and Reardon 1988). In surface studies, pyrite oxidation was independent of oxygen partial pressures above 10%. However, at small column scale, the reaction is first order in this range. It can be speculated that at surface scale, the diffusional component of pyrite oxidation (the most sensitive to oxygen partial pressure) is minimal.

Oxidation with Bacteria. The room temperature biotic oxidation of pyrite was carried out under unsaturated conditions at oxygen contents of 0.005, 0.1, 0.3, 0.5, 1.0, and 10% (5% oxygen column broke during test). Each column was inoculated with 30 ml of acid drainage from a coal mine in Westmoreland County, PA. MPN's were performed during the experiment to make certain that each column contained viable bacteria.

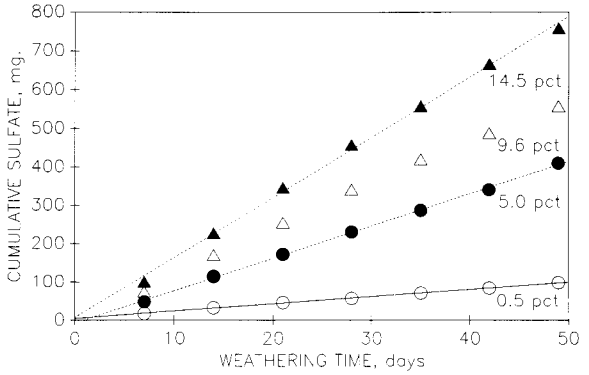



Figure 7. Plots of cumulative sulfate load versus weathering time for the abiotic weathering of pyrite in small columns under different oxygen partial pressures.

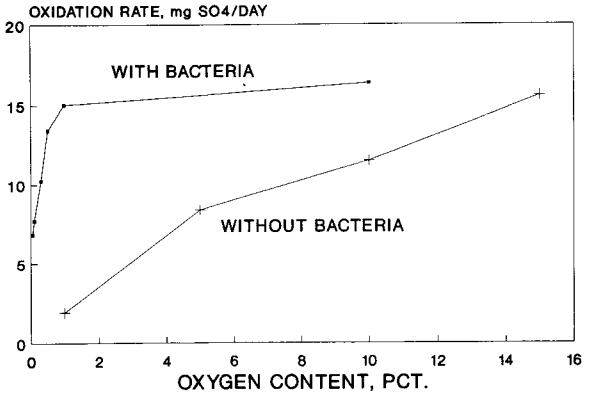



Figure 8. Abiotic rates of pyrite oxidation in small columns maintained at different oxygen partial pressures.

Cumulative sulfate versus weathering time plots for biotic oxidation are shown in Figure 9. Oxidation rates were determined from the slope of the least-squares fit line through the stationary phase. At this time, the effect of sulfate or ferric iron added with the inoculum would be insignificant. The inoculum could, at most, contribute only about 60 mg of sulfate including sulfate from the oxidation of pyrite by ferric iron in the inoculum. Unlike abiotic oxidation, biotic

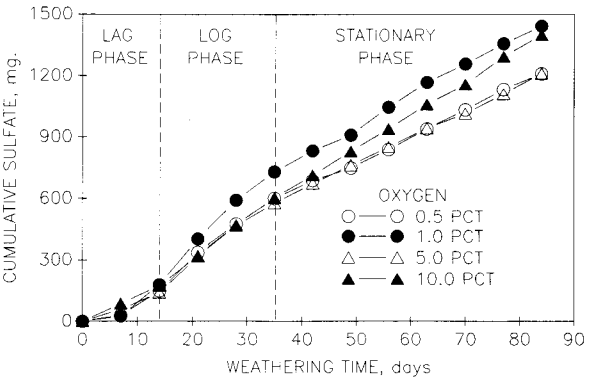



Figure 9. Plots of cumulative sulfate load versus weathering time for the biotic weathering of pyrite in small columns under different oxygen partial pressures.

Table 2. Abiotic Oxidation in Small Columns

| Oxygen | Reaction order | Rate equation |
|---|---|---|
| Below 5 pct | Fractional | Insufficient data |
| Above 5 pct | 1st | $dSO_4/dt$ = (0.006 mg $SO_4$/g-$FeS_2$/hr/pct $O_2$)(pct $O_2$) + 4.63 |

Table 3. Biotic Oxidation in Small Columns

| Oxygen | Reaction order | Rate equation |
|---|---|---|
| 0.1 - 1 pct | 1st | $dSO_4/dt = (0.073 \text{ mg } SO_4/g\text{-}FeS_2/hr/pct\ O_2)(pct\ O_2) + 7.22$ |
| Above 1 pct | 0 | $dSO_4/dt = 0.13 \text{ mg } SO_4/g\text{-}FeS_2/hr$ |

oxidation is not significantly inhibited by reduced oxygen contents (Fig. 8, with bacteria) and Table 3. In fact, the reaction rate is independent of oxygen partial pressures (0-order reaction) above 1%. Below 1% oxygen, the reaction appears to be first order until some point below 0.1%.

The oxidation rate at oxygen partial pressures above 1% (0.13 mg $SO_4/g\text{-}FeS_2/hr$) is in good agreement with published rates (Table 4). It is also apparent that the rates of abiotic and biotic pyrite oxidation are not significantly different at oxygen partial pressures above 15% (Fig. 8). The role of bacteria is to sustain high oxidation rates at low oxygen partial pressures.

Large Columns

Small column tests indicated that the rate of biotic pyrite oxidation was independent of oxygen partial pressures (0-th order reaction) above 1%. This indicates that measures taken to reduce oxygen partial pressures would be ineffective at reducing pyrite oxidation unless oxygen could be maintained at levels below 1%. This is not economically possible at present using barriers other than water. In large column tests, no difference in biotic pyrite oxidation rate was found between columns with applied atmospheres of 1.5% and 10.5% oxygen (Fig 10). These results corroborate the results of small column tests and indicate that current oxygen-impermeable or -consumptive barriers (except water) will not be effective at limiting pyrite oxidation without bacterial inhibition.

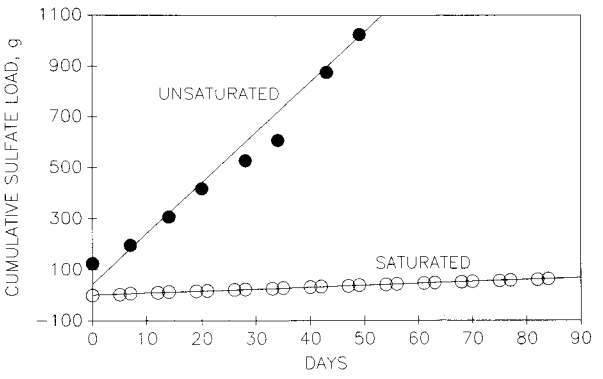



Figure 10. Plots of cumulative sulfate load versus weathering time for saturated and unsaturated pyritic shale in large columns.

Table 4. Pyrite oxidation rates, oxygen dependence, and water dependence reported by other authors.

| Investigator | Observed rate | $O_2$-dependence | $H_2O$-dependence | Pyrite type | Particle size | Temperature | Species Monitored | Remarks |
|---|---|---|---|---|---|---|---|---|
| Braley (1960) | 0.13 mg $SO_4^{-2}g^{-1}FeS_2$ $hr^{-1}$ | Linear (first-order) | NA | Sulfur ball | -8 + 40 mesh | | Sulfate | |
| Clark (1965) | 0.11 mg $SO_4^{-2}g^{-1}FeS_2$ $hr^{-1}$ | Linear (first-order) for dissolved oxygen | NA | Sulfur ball | -40 + 50 mesh | 20° C | Sulfate | |
| Morth and Smith (1966) | 0.06 mg $SO_4^{-2}g^{-1}FeS_2$ $hr^{-1}$ | Non-linear (fractional order) | Non-linear (order exceeds 1 as 100% relative humidity is approached) | Sulfur ball | -70 + 100 mesh | 25° C | Oxygen | |
| Rogowski and Pionke (1984) | 0.16 mg $SO_4^{-2}g^{-1}FeS_2$ $hr^{-1}$ | ND | ND | Not spec. | 2-8 mm | 25° C | Sulfate | Spoil sample |
| Nicholson and others (1988) | 0.13 mg $SO_4^{-2}g^{-1}FeS_2$ $hr^{-1}$ | Adsorption (Langmuir-type) | NA | Massive, from various locations | 76 um | 23° C | Sulfate | Carbonate buffered |

NA - Not applicable.
ND - Not determined.

The effect of leaching frequency is shown in Figure 11. Columns 1-4 were leached as follows: column #1, 1L every day; column #2, 7L every 7 days; column #3, 14L every 14 days; and column #4, 21L every 21 days. The results of this experiment indicated that the build up of weathered products did not retard pyrite oxidation by surface passivation. An alternative explanation is that the solubilization of sulfate minerals under these conditions may be slower than pyrite oxidation. Therefore, leaching frequency tests may only be indicative of the rate of oxidation product solubilization rather than pyrite oxidation.

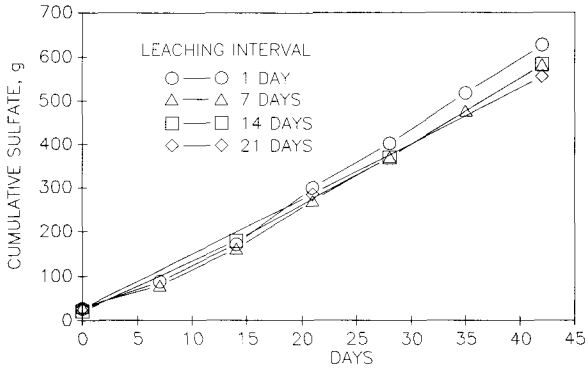



Figure 11. Plots of cumulative sulfate load versus weathering time at different leaching frequencies.

Results of this study indicate that attempts to limit pyrite oxidation under unsaturated conditions would be difficult and undoubtedly expensive. However, water is an effective oxygen barrier. Figure 12 compares the rates of release of sulfate under unsaturated and water-saturated conditions. For unsaturated leaching, sulfate was released at a rate of 18.2 g/day compared to 0.7 g/day when the shale remained inundated and flushed biweekly with 20 L of oxygen-saturated water (D.O. = 8 mg/L). Stoichiometrically, the amount of oxygen added through flushes with oxygen-saturated water can account for 0.2 g/day of sulfate from the oxidation of pyrite. The remaining 0.5 g/day of sulfate released could be due to additional pyrite oxidation from ferric iron resident in the columns or dissolved oxygen transferred from the atmosphere. It is also possible that some of the sulfate is from the continued solubilization of previously formed sulfate compounds. Additional tests using oxygen-depleted water for the 20-L flushes resulted in a sulfate release rate of 0.5 g/day.

The importance of maintaining complete water inundation was illustrated during these tests. A leak developed between tests that dropped the level of the water to 0.5 m below the suface of the shale. It is not known how long this upper 0.5 m of shale was exposed, but it was for no more than 32 days. During and for a short period after resaturation, the sulfate release rate into solution was a minimum of 6.9 g/day, ten times the rate during saturation. After resaturation, the sulfate release rate dropped to pre-leak levels.

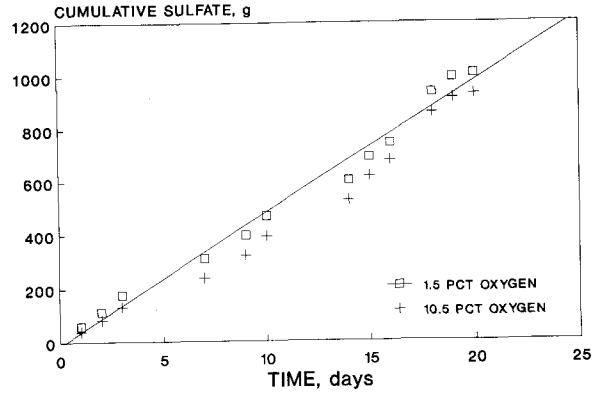



Figure 12. Plots of the cumulative sulfate load versus weathering time for large columns maintained at oxygen partial pressures of 1.5% and 10.5%.

Implications for Spoil Handling

When material is first exposed by mining, fresh pyrite surfaces occur along fractures in the newly broken rock. The pH of the material is near-neutral, which limits ferric ion solubility and the catalytic effect of bacteria on pyrite oxidation. In this scenario, the abiotic surface oxidation rates determined by XPS are applicable. These rates are rapid because the reaction is not diffusion-limited at this point. The oxygen partial pressure would have to be maintained below 10% to affect pyrite oxidation rates in fresh material. Although these conditions may only exist for a few days, the initial abiotic oxidation is important because it establishes the low-pH, ferrous iron-rich environment necessary for the growth of iron-oxidizing bacteria.

Once bacterially-mediated pyrite oxidation is established, the oxygen partial pressure would have to be maintained below 1% to realize any reduction in oxidation rate. Reducing oxygen partial pressures to this level with solid covers is not practical. However, if bacteria are inhibited, any reduction in oxygen partial pressure from atmospheric (21%) will result in a proportional decrease in pyrite oxidation rate.

The most effective strategy for limiting pyrite oxidation is to place pyritic material below a permanent water table. Results of this study indicate that pyrite oxidation rates can be reduced by at least 96% with inundation. However, it is important that pyritic spoil be immediately flooded. Any storage above the water table will result in rapid pyrite oxidation even at low lovels of oxygen unless bacteria are inhibited. The oxidation products that form during storage will be solubilized when the material is flooded, resulting in an slug of degraded water. However, once the stored oxidation products are removed, water quality will improve. Any field trial of inundation must allow for differentiation between sulfate derived from product solubilization and sulfate from on-going pyrite oxidation. It is also desirable that groundwater flow through the impoundment be slow and from an oxygen-depleted source, if possible. Oxidation rates determined for oxygen-saturated aqueous pyrite suspensions

(Nicholson, Gillham, and Reardon 1988) are equivalent to the rates determined in this study for unsaturated conditions. This indicates that flooding will not reduce pyrite oxidation if the pyrite is contacted by a high flow of oxygen-saturated water. However, these conditions are unlikely to exist in an inundated backfill.

Results of this study indicate that under unsaturated conditions, pyrite oxidation rates can only be reduced by bacterial inhibition in conjunction with an oxygen-consumptive or oxygen-impermeable barrier. Inundating pyritic material will reduce oxidation rates by at least 96% and is the the most effective means of controlling pyrite oxidation.

## Literature Cited


American Public Health Association. 1985. Standard Methods for the Examination of Water and Wastewater. 16th Edition. American Public Health Association, Inc., Washington, DC, p. 870-886.

Braley, S. A. 1960. The Oxidation of Pyritic Conglomerates. Spec. Rept. to Coal Ind. Advisory Comm. to Ohio River Valley Water Sanit. Comm., Res. Proj. No. 370-6, 32 p.

Brundle, C. R. 1982. Ultra-High Vacuum Techniques of Surface Characterization. p. 14-16. In Industrial Applications of Surface Analysis. L. A. Casper and C. J. Powell (eds.). ACS Symposium Series 199 (Washington, D.C.).

Buckley, A. N., R. Woods, and H. J. Wouterlood. 1987. The Deposition of Sulfur on Pyrite and Chalcopyrite from Sodium Sulfide Solutions. In Proceedings of the Royal Australian Chemical Institute Eighth National Convention (Sydney, Australia, 1987).

Clark, C. S. 1965. The Oxidation of Coal Mine Pyrite. Ph.D. Thesis, The Johns Hopkins Univ., 90 p.

Cobley, J. G. and B. A. Haddock. 1975. The Respiratory Chain of Thiobacillus ferrooxidans: the Reduction of Cytochromes by $FE^{+2}$ and the Preliminary Characterization of Rusticyanin a Novel "Blue" Protein. FEBS Letters 60: 29-33.

Lowson, R. T. 1988. Aqueous Oxidation of Pyrite by Molecular Oxygen. Chemical Reviews 82(5):461-496.

Lusardi, P. J. and P. M. Erickson. 1985. Assessment and Reclamation of an Abandoned Acid-Producing Strip Mine in Northern Clarion County, Pennsylvania. p. 313-321. In Proceedings of Symposium on Surface Mining, Hydrology, Sedimentology, and Reclamation. (Lexington, KY, Dec. 9-13, 1985).

Morth, A. H. and E. E. Smith. 1966. Kinetics of the Sulfide-to-Sulfate Reaction. Symp. Fossil Fuels Environmental Pollution, ACS Division, Fuel Chemistry Preprints 10(1):83-92.

Nicholson, R. V., R. W. Gillham, and E. J. Reardon. 1988. Pyrite Oxidation in Carbonate-Buffered Solution. Geochemica et Cosmochimica Acta 53(5):1077-1085.

Pugh, C. E., L. R. Hossner, and J. B. Dixon. 1984. Oxidation Rate of Iron Sulfides as Affected by Surface Area, Morphology, Oxygen Concentration, and Autotrophic Bacteria. Soil Science 137(5):309-314.

Rogowski, A. S. and H. B. Pionke. 1984. Hydrology and Water Quality on Stripmined Lands. U.S. Environmental Protection Agency EPA-IAG-D5-E763, 183 p.

Watzlaf, G. R. and P. M. Erickson. 1986. Topical Amendments of Coal Refuse: Effect on Pore Gas Composition and Water Quality. p. 255-261. In Proceedings of National Symposium on Mining, Hydrology, Sedimentology, and Reclamation (Lexington, KY, Dec. 8-11, 1986).

Watzlaf, G. R. and R. W. Hammack. 1989. The Effect of Oxygen, Iron-Oxidizing Bacteria, and Leaching Frequency on Pyrite Oxidation. In Proceedings of the Ninth Annual West Virginia Surface Mine Drainage Task Force Symposium. (Morgantown, WV, April 25-26, 1989).

# AN APPLICATION OF ACID-BASE ACCOUNTING FOR HIGHWAY CONSTRUCTION IN EAST TENNESSEE[1]

by

J.T. Ammons, C.B. Coburn, Jr., and P.A. Shelton[2]

Abstract. Concern over the environmental impact of highway construction on the foothills Scenic Parkway in east Tennessee prompted the use of acid-base accounting to evaluate overburden rocks. Samples were acquired during active construction to evaluate the acid producing potential of the overburden. A core was provided representing the middle portion of the overburden for acid-base accounting and mineralogical analyses. Substantial quantities of pyrite were noted in a few zones in the overburden. Most of the overburden contained sufficient neutralizers to neutralize the potential acidity if the overburden was blended in the fill during construction. Higher neutralization potential values in the overburden were directly related to the relative proportion of dolomite detected in the samples.

Additional key words: acid-base accounting, highway construction, east Tennessee.

## Introduction

The Tennessee Department of Transportation (TDOT) had expressed concern that acidity problems might be encountered during the construction of the Foothills Scenic Parkway in East Tennessee (Fig. 1). These concerns were based on sporadic vegetation failures observed on the previously completed section of the parkway.

Certain sections of the precambrian rocks of east Tennessee contain pyrite ($FeS_2$), (Byerly and Middleton, 1981) and the oxidation of pyrite will result in the formation of acidity. Drastic land disturbances, such as highway construction, expose iron disulfide to the air and water necessary to promote the following reaction presented in a simplified form:

$$4FeS_2 + 15O_2 + 14H_2 \text{ ----> } 4Fe(OH)_3 + 8H_2SO_4$$

The production of sulfuric acid affects water and soil quality. Minerals are dissolved in the acid environment developed during the oxidization process, and elements are mobilized in the soil which could impact water quality in the immediate watershed.

Jago and Byerly (1988) conducted a leaching study on pyritic precambrian rocks containing pyrite in east Tennessee. Using acid-base accounting, they established that the acid-producing potential for the material under evaluation was 23.7 tons per 1000 tons material ($CaCO_3$ equivalent). They concluded that additions of interlayer lime to fill material improved the leachate quality.

The objectives of this study were to identify the acid producing strata in the overburden core and to access the environmental impact for revegetation and water quality.

## Materials and Methods

Several samples were analyzed from the active construction site to assess whether the overburden had acid-producing potential. An overburden core was provided by TDOT from the geologic section affected during construction. Water quality sampling stations were in place to monitor

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]J.T. Ammons is Associate Professor, Department of Plant and Soil Science, The University of Tennessee at Knoxville. C.B. Coburn, Jr. is Professor of Biology, Tennessee Technological University, Cookeville, Tennessee. P.A. Shelton is Research Associate, Southern Illinois University at Carbondale.

chemical parameters for pre-construction, active-construction, and post-construction phases of the project.

The first sampling was completed on-site during active construction. Six samples were removed at random from disposal piles. Different rock types were selected to determine if materials placed in the fill were potentially toxic. The Tennessee Department of Transportation provided a thirty-two foot section of the overburden for laboratory analyses. The upper twenty-five feet of the core was not sampled. The overburden core and the pile sample sites were located near Caylor Gap on the Wear Cove, Tennessee Geologic Quadrangle (Fig. 1).

Both overburden core and pile samples were ground to less than 60 mesh and were analyzed using acid-base accounting (Smith et al. 1974; Sobek et al. 1978). Total sulfur values were used to calculate the acid-base account. The neutralization potential followed the guidelines presented by Shelton et al. (1984).

Mineralogical analyses were conducted on the overburden core samples. Ground samples (less than 60 mesh) were placed in metal powder sample holders and scanned from 2 degrees to 60 degrees 2 theta using copper (k) alpha radiation (McMurdie et al. 1986).

Water quality has been monitored since the inception of the construction on the parkway. Station I (Fig.1) is a water monitoring sampling station in the watershed draining Caylor Gap where the overburden core and the pile samples were collected. Data reported are an average of twelve months for each year.

## Results and Discussion

### Preliminary sampling

Results from the pile samples obtained during active construction are reported in Table 1. The rocks were identified as part of the Anakeesta formation, Precambrian age, from the Great Smokey Mountain Group. The Anakeesta formation consists of "dark gray, bluish-gray, black slates with interbeds of fine-grain sandstone" (Hardeman 1966). Three of the pile samples were potentially toxic (Table 1). Samples showing a deficiency of 5 tons/1000 tons material ($CaCO_3$ equivalent) are considered potentially toxic (Smith et al. 1974; Sobek et al. 1978).

### Overburden Analysis

All of the samples contained a significant concentration of total sulfur (>0.4%) (Table 2). In most samples, sufficient neutralizers were present to counteract potential acidity.

Potentially toxic zones were found at depths of 30 to 37 feet and 47 to 52 feet (Table 2 and Fig. 2). The potentially toxic

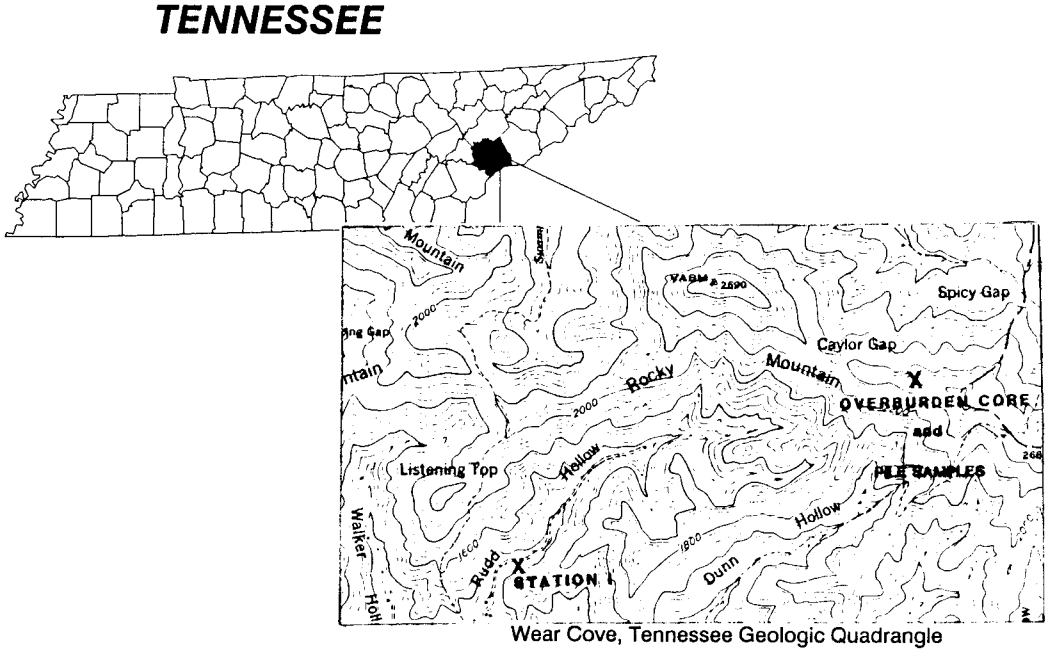



Figure 1. Location of Overburden Core, Pile Samples, and Water Monitoring Station on the Foothills Scenic Parkway.

Table 1. Acid-Base Account of Grab Samples. Foothills Scenic Parkway.

| Sample Number | Paste pH | Munsell Color | Fizz | Percent Sulfur | $CaCO_3$ equivalents (tons/1000 tons material) Max From Sulfur | N.P. | Deficiency | Excess |
|---|---|---|---|---|---|---|---|---|
| 1 | 5.8 | 2.5Y6/0 | None | .063 | 1.98 | 1.03 | .95 | |
| 2 | 5.4 | 5Y8/1 | None | .019 | .58 | .11 | .47 | |
| 3 | 7.4 | 5Y6/1 | Slight | .031 | .96 | 1.03 | | .07 |
| 4 | 6.2 | 2.5Y6/0 | Slight | 2.149 | 67.16 | 2.86 | 64.30 | |
| 5 | 7.3 | 2.5Y7/0 | Slight | .684 | 21.38 | .92 | 20.46 | |
| 6 | 7.0 | 2.5Y6/0 | Slight | 2.358 | 73.69 | 2.86 | 70.83 | |

rocks were sandstone conglomerates with a dark shale-slate intercalate. Pyrite was visually confirmed between the bands of shaly slate and the sandstone conglomerate with the aid of a 10X hand lens. Revegetation and water quality problems can be anticipated if the potentially toxic materials are deposited on the surface.

The physical stability of the pyrite containing rocks were not evaluated in this study, but the rapidity of rock disintegration in the fill indicates that water quality and revegetation may be impacted. Future studies on the Precambrian rocks in east Tennessee should include physical stability and\or weatherability evaluations. If these rocks are resistant to physical and chemical breakdown in the construction fill, products from the oxidation of pyrite will be minimized. This will enhance success of revegetation and reduce affect on water quality in the watershed. A rapid, chemical and physical breakdown may affect the stability of the fill plus increase surface area which will increase oxidation reactions in rocks containing pyrite.

## Mineralogy

Primary minerals present in the overburden samples were chlorite, muscovite, orthoclase, quartz, and dolomite (Table 3). Pyrite was not detected in the x-ray analysis. Samples containing less than 10% dolomite were potentially toxic (Table 2). The presence of dolomite contributed heavily to the high NP values. In the absence of dolomite, the NP's are low. The dolomite is probably responsible for the majority of the bases in these overburden rocks.

## Water Quality

Sulfate and magnesium concentrations in the water varied over the five-year period (Fig. 3), but showed a slight increase. The variation in sulfate was probably due to differences in rainfall over the five-year period. The increase in magnesium concentrations were probably related to weathering of dolomite in the fill.

## Conclusions

Substantial quantities of pyritic

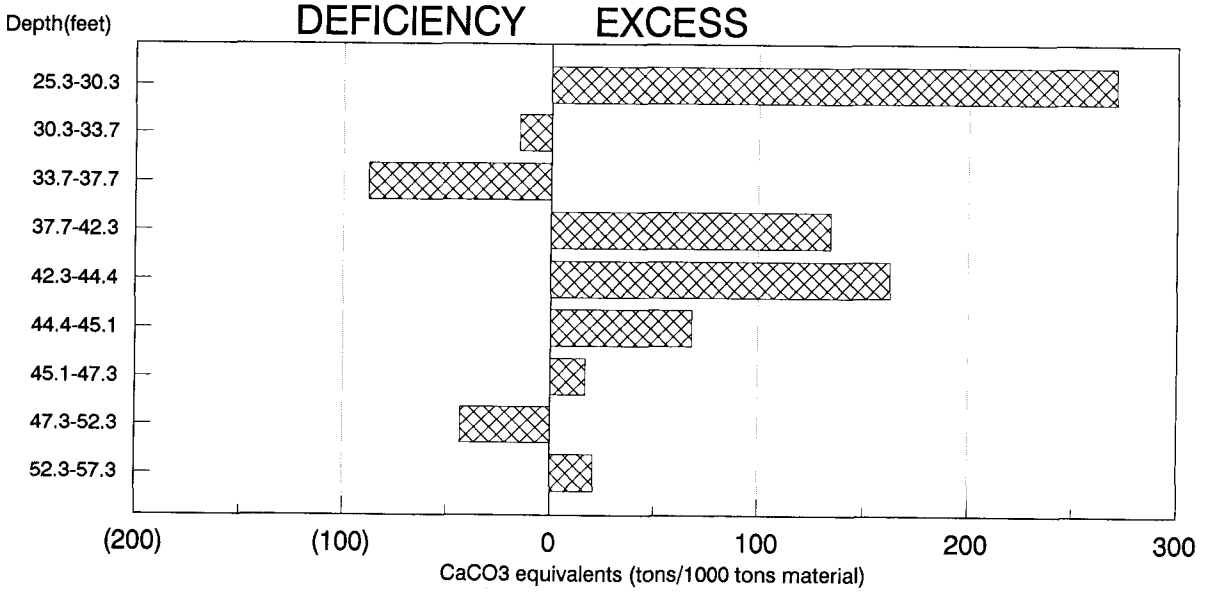



Figure 2. Acid-Base Account of Foothills Parkway Overburden Core.

Table 2. Acid-Base Account of Overburden Core. Foothills Scenic Parkway.

| Sample Number | Depth | Munsell Color | Fizz | Percent Sulfur | $CaCO_3$ equivalents (tons/1000 tons material) Max From Sulfur | N.P. | Deficiency | Excess |
|---|---|---|---|---|---|---|---|---|
| | ft | | | | | | | |
| 1 | 25.3-30.3 | N 6/0 | Strong | .843 | 26.35 | 297.43 | | 271.08 |
| 2 | 30.3-33.7 | N 6/0 | Slight | .717 | 22.40 | 7.00 | 15.40 | |
| 3 | 33.7-37.3 | N 5/0 | Slight | 2.840 | 88.88 | 1.00 | 87.88 | |
| 4 | 37.3-42.3 | N 6/0 | Strong | .674 | 21.07 | 155.38 | | 134.31 |
| 5 | 42.3-44.4 | N 5/0 | Strong | 1.160 | 36.25 | 199.09 | | 162.84 |
| 6 | 44.4-45.1 | N 5/0 | Moderate | .810 | 25.31 | 93.47 | | 68.16 |
| 7 | 45.1-47.3 | N 5/0 | Moderate | 1.443 | 45.09 | 61.90 | | 16.81 |
| 8 | 47.3-52.3 | N 5/0 | Moderate | 2.853 | 89.16 | 46.12 | 43.04 | |
| 9 | 52.3-57.3 | N 5/0 | Moderate | .925 | 28.91 | 49.76 | | 20.85 |

Table 3. Summary of Mineralogical Analyses for Overburden Core. Foothills Scenic Parkway.

| Sample | Depth | Minerals Present: Chlorite[1] | Muscovite[2] | Orthoclase[3] | Dolomite[4] | Quartz[5] |
|---|---|---|---|---|---|---|
| | ft | Approx. Percentage | | | | |
| 1 | 25.3-30.3 | 20-30 | 10-20 | ND | 10-30 | >30 |
| 2 | 30.3-33.7 | ND | 10-20 | 20-30 | ND | >30 |
| 3 | 33.7-37.3 | ND | 20-30 | 20-30 | ND | >30 |
| 4 | 37.3-42.3 | ND | 10-20 | 20-30 | 10-20 | >30 |
| 5 | 42.3-44.4 | ND | 20-30 | 20-30 | 20-30 | >30 |
| 6 | 44.4-45.1 | ND | 20-30 | 10-20 | 10-20 | >30 |
| 7 | 45.1-47.3 | ND | 20-30 | 10-20 | 10-20 | >30 |
| 8 | 47.3-52.3 | ND | 20-30 | 10-20 | 10-20 | >30 |
| 9 | 52.3-57.3 | ND | 20-30 | 10-20 | <10 | >30 |

[1] - Chlorite - $(Fe,Al,Mg)_6(Si,Al)_4O_{10}(OH)_8$

[2] - Muscovite - $KAl_2(Si_3,Al)_1O_{10}(OH,F)_2$

[3] - Orthoclase - $KAlSi_3O_8$

[4] - Dolomite - $Ca,Mg(Co_3)_2$

[5] - Quartz - $SiO_2$

materials are present in the Precambrian rocks studied on the Foothills Scenic Parkway. Byerly and Middleton (1981) reported acid-base account values for potentially deleterious rocks southwest of the Great Smoky Mountains in Monroe County, Tennessee. Values in this study ranged from a deficiency of 7.69 to 42.88 calcium carbonate equivalent (tons/1,000 tons material). This is compared to samples from the Foothills Parkway where the pile samples ranged from an excess of .07 to a deficiency 70.83 and the overburden samples ranged from an excess of 271.08 to a deficiency of 87.88 calcium carbonate equivalent (tons/1,000 tons material). Most zones in the overburden tested on the Foothills Parkway contain sufficient quantities of basic materials to neutralize potential acidity in the fill if the materials are blended. This is supported, in part, by results reported by Jago and Byerly (1988). In a pyritic material encapsulation study involving Precambrian rocks, they found the best leachate quality was achieved by interlayering limestone with the fill material.

Higher NP values in the Foothill's study are directly related to the relative proportion of dolomite detected in the samples. This is probably responsible for elevated levels of magnesium in the water quality samples.

The following are suggested guidelines to accomplish the best post-construction environmental quality:

1. Complete an overburden analysis (acid-base accounting) on the geologic section to be removed during construction.

2. During construction, monitor overburden for changes in depth or sequence of acid-toxic rocks.

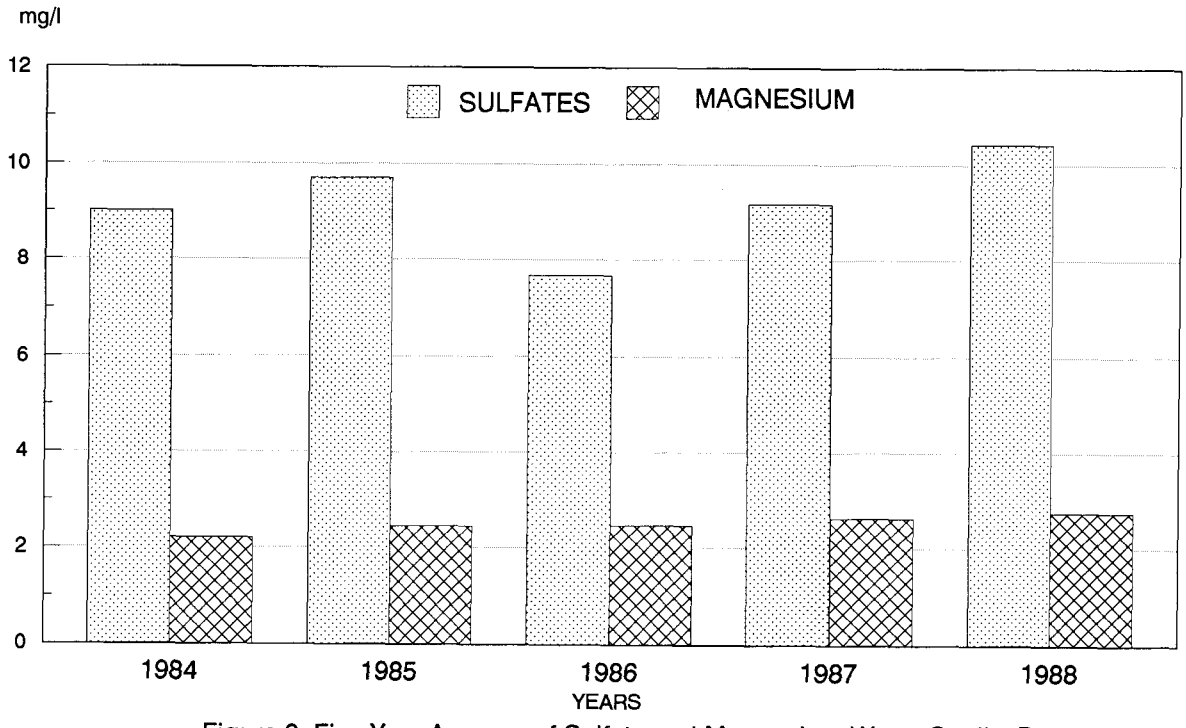



Figure 3. Five-Year Average of Sulfate and Magnesium Water Quality Data.

Station I, Foothills Scenic Parkway.

3. Examine older fills constructed from the same geologic materials for presence of acid-toxic rocks and vegetation success.

4. Study the physical stability of overburden rocks under controlled experimental conditions.

5. Monitor water quality in watersheds affected by construction activities.

## Literature Cited

Byerly, Don W. and Lloyd M. Middleton. 1981. Evaluation of the acid drainage potential of Certain Precambrian Rocks in the Blue Ridge Province. pp. 174-175. In 32nd Annual Highway Geology Symposium Proceedings. Tennessee Department of Transportation, Division of Soils and Engineering, Nashville, TN.

Hardeman, William D. 1966. East Sheet. Geologic Map of Tennessee. State of Tennessee Department of Conservation. Division of Geology. Nashville, TN.

Jago, William K. and Don W. Byerly. 1988. Preliminary Results of Field Leaching Methods to Evaluate Pyritic Material Encapsulation Techniques Related to Highway Construction. pp. 73-78. In 1988 Symposium on Mining, Hydrology, Sedimentology and Reclamation. (University of Kentucky. Lexington, KY 40506-0046).

McMurdie,H.F., M.C. Morris, E.H. Evans, B. Parentzkin and W. Wong-Nq. 1986. Methods of Producing Standard X-Ray Diffraction Powder Patterns. Powder Diffraction 1:40-43.

Shelton, P.A., J.T. Ammons and J.R. Freeman. 1984. Neutralization Potential: A Closer Look. Greenlands Quarterly. 13:4. West Virginia Surface Mining and Reclamation Association, Charleston, WV. pp. 35-37.

Smith, R.M., W.E Grube, Jr., T. Arkle, Jr., and A.A. Sobek. 1974. Mine Spoil Potential for Soil and Water Quality. EPA-670/2-74-070. US-EPA. Cincinnati, OH.

Sobek, A.A., W. Schuller, J. Freeman, and R.M. Smith. 1978. Field and Laboratory Methods Applicable to Overburdens and Minesoils. EPA-600/2-78-054. US-EPA. Cincinnati, OH.

This page intentionally left blank.

# Forestry

This page intentionally left blank.

# GUIDELINES FOR ESTABLISHING PRODUCTIVE FOREST LAND ON RECLAIMED SURFACE MINES IN THE CENTRAL APPALACHIANS[1]

J.L. Torbert and J.A. Burger[2]

Abstract. This paper presents an overview of forestry reclamation research conducted within the Forestry Department at Virginia Polytechnic Institute and State University since 1980. Research indicates that coal mining and subsequent reclamation in the Central Appalachian region of Virginia, West Virginia, and Kentucky presents a good opportunity to create productive forests on land that is often too steep or remote to be used for crop production, grazing, or commercial development. Unfortunately, coal companies seldom reclaim these lands in a fashion that will result in productive forest land. Overburden selection, surface grading, and ground cover establishment are usually oriented towards producing a hayland/pasture land-use. In order to develop productive forest land, special considerations need to be given to the requirements of forest tree species as opposed to agronomic forages. Preliminary results from our research program indicate that for best tree establishment and long-term growth, moderately-acidic sandstone overburdens should be used as topsoil substitutes when real topsoil is not recoverable, grading and tracking-in should be minimized to avoid compaction, and a tree-compatible ground cover of acid-tolerant, short grasses and legumes should be used for revegetation.

## Introduction

Forestry is a logical land-use for most of the reclaimed mined land in the Appalachian Mountains because most reclaimed sites are too steep or remote to be used for grazing, crops, or commercial development. Consequently, more than 80% of the coal mining permits issued in Virginia during the last few years have specified "forest land" as the designated post mining land-use (Virginia Division Mined Land Reclamation, personal communication). According to PL 95-87, the Surface Mining Control and Reclamation Act of 1977 (SMCRA), mined lands are supposed to be reclaimed in a fashion that promotes their intended land-use. Unfortunately, most land designated as forest land is not reclaimed in a way that favors tree establishment or (more importantly) long-term forest productivity.

A research program was started by the Forestry Department of Virginia Polytechnic Institute and State University (VPI&SU) in 1980 with initial funding from the Powell River Project, an industry-university research cooperative, to study reforestation of surface mined land. Early emphasis was placed on studying the establishment of trees on land reclaimed before 1977 (pre-SMCRA). A trial of fifteen species was established along with a study to evaluate the effects of a mycorrhizal fungus (Pisolithus tinctorius), fertilizer planting tablets, and chemical weed-control on the establishment and development of three species of containerized pine. Although these studies have yielded valuable results (Schoenholtz and Burger 1984; Torbert et al. 1985, and Schoenholtz et al., 1987), research emphasis shifted in subsequent years to concerns of site productivity and techniques for modifying reclamation processes to create better sites for tree establishment.

Our studies show that in many cases excellent sites can be created during reclamation such that tree growth often exceeds the pre-mining level of productivity. When a site is properly reclaimed and revegetated, virtually any tree species suitable to the climate can be established without the need for containerized seedlings,

---

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, WV,. April 23-26, 1990.

[2] John L. Torbert is Forest Soils Research Associate, and James A. Burger is Associate Professor of Forest Soil Science at the School of Forestry and Wildlife Resources, Virginia Polytechnic Institute and State University, Blacksburg, VA. 24061-0324.

mycorrhizal inoculation, fertilizer tablets, or chemical weed-control.

The objective of this paper is to outline the forestry reclamation research conducted at VPI&SU since 1980, and to present our research-based recommendations for reclaiming forest land in Virginia and surrounding states. Our results show that the greatest constraints to tree establishment and long-term growth in this region are 1) the use of inappropriate overburden for topsoil substitutes, 2) compaction from final grading and "tracking-in", 3) excessive levels of herbaceous ground cover during the year that trees are planted, and 4) improper planting techniques.

## Overburden selection

Proper minesoil construction is essential for long-term productivity of forest land. The inherent productivity of much forest land in the Appalachians is limited by shallow soils. Even though the area receives abundant rainfall, inadequate soil-water storage limits forest production in this region. Surface mining offers the opportunity to replace shallow and infertile soils, with carefully constructed minesoils that are deeper and often more fertile.

In order to take advantage of the deeper soils that can result from surface mining, overburden strata must have desirable physical and chemical properties for trees. Overburden strata differ in many important ways that influence plant growth. These properties include coarse fragment content, resistance to weathering, texture of the fine-earth fraction (<2mm), pH, soluble salt concentrations, and fertility. Forest tree species have different minesoil requirements than forage grasses and legumes, both in terms of quality and depth. Frequently, coal companies select finely textured overburden materials with a pH near neutral in order to maximize the production of ground covers without realizing the adverse effect that such overburden types can have on tree establishment and growth.

Several studies have been conducted to evaluate the effects of various overburden types on tree growth. In a greenhouse study, pine seed germination was significantly higher on a siltstone spoil than a sandstone spoil because the soil fraction associated with the siltstone was more finely textured and had greater water retention. However, seedling growth was greatest in the moderately acid sandstone (Preve et al. 1984). Many tree species grow poorly in soils with a high pH and prefer soils with a pH of 5 to 6.

In the field, an overburden-type study was established to evaluate the effect of various overburden mixes on tree establishment and growth. Treatment plots consisting of sandstone, siltstone, and different proportions of the two were carefully constructed to produce four feet of uncompacted growth media. Half of each plot was seeded with Ky-31 tall fescue (Festuca arundinacea Schreb.) and the other side was planted with pitch x loblolly hybrid pine (Pinus xrigitaeda) seedlings. After five years, the rock type had no effect on tree survival, but it had a large effect on tree growth. The average height of trees in the pure sandstone plots was 50% greater than the height of trees in pure siltstone (75 inches vs. 51 inches), and stem volume was five times greater (113 $in^3$ vs 23 $in^3$); (Torbert et al. 1990a). The sandstone spoil used in this study was a better growth medium for trees largely because it had a lower pH (5.7 vs 7.1) and lower levels of soluble salts (0.4 vs 1.3 dS $m^{-1}$). In contrast, the fescue performed better in the siltstone spoil during the first three years and was unaffected by spoil type by the fifth year (Roberts et al. 1988). It is important to note that selection of a spoil for maximum first-year ground cover performance in this case would have resulted in a significant sacrifice of tree performance. Furthermore, the early enhancement of ground cover growth did not persist through the entire bond period.

A minesoil-quality study (Torbert et al. 1988) was conducted to evaluate the relationship between minesoil properties and the growth of ten-year-old eastern white pines (Pinus strobus L.) that were planted across an area of several thousand acres in southwestern Virginia. Based on the height of trees at age ten and the use of Beck's (1971) equation for white pine polymorphic site index curves, the site index (base age 50) for each site was estimated and found to range from 40 ft to 110 ft. Based on this study, the minesoil properties most influential on tree growth were rooting volume and level of soluble salts. These results corroborate those of Ashby et al. (1984) who reported excellent tree growth on minesoils in Illinois, and who attributed the performance of those trees to the presence of a deep, uncompacted, non-toxic minesoil. Plass (1979) summarized results of several studies that found a negative relationship between soluble salt levels and performance of various pine species.

In our minesoil-quality study, the best tree growth was found on acidic, sandy minesoils derived from oxidized sandstone overburden found near the surface before mining. In addition to results obtained from our studies, observations across numerous sites in Virginia, West Virginia, and Kentucky, indicate that, with few exceptions, whenever excellent tree growth is found on minesoils, the trees are growing in an oxidized (brown) moderately acid, sandstone-derived minesoil. Oxidized sandstone is usually highly fractured and tends to produce more soil-sized particles during the blasting process than the harder sandstone, siltstone, and shale strate that occurs further below the surface. Furthermore, oxidized sandstone weathers more rapidly; rocks have been observed to weather completely to a loamy sand within a few years.

Unfortunately, even though oxidized sandstone is abundant in Virginia and is a superior growth medium for trees, reclamationists often avoid using this material as a topsoil substitute. Oxidized sandstone in southwestern Virginia and much of West Virginia and Kentucky has a pH of

4.5 to 5.5 and a high phosphorus fixing capacity that renders this material less suitable for agronomic grasses and legumes. However, this problem can be overcome by using a mixture of tree-compatible grasses and legumes that are tolerant of the chemical and physical minesoil properties that maximize tree growth.

## Surface grading

Prior to revegetation, the final surface is typically cleared of large boulders, gullies are filled, and the surface is graded smooth and "tracked-in" with bulldozer cleats to improve seed bed conditions. These practices promote the establishment of a uniform ground cover that facilitates bond release. While these practices may be conducive to grassland establishment, the harmful effects of such practices to trees due to minesoil compaction have been lamented by forestry reclamation researchers for years (Vogel 1981, Larson and Vimmerstedt 1983, Ashby et al. 1984). Poor seedling survival often results on compacted soils because trees are not planted deeply enough, and root growth is restricted by the low moisture holding capacity and high soil strength associated with compacted soils. A more subtle effect of compaction that is not readily recognized is the permanent decrease in minesoil quality and decreased growth rates of surviving trees.

To quantify and demonstrate the effects of compaction on the survival and growth of several tree species caused by final grading and "tracking-in", a compaction study (Torbert et al. 1990b) was established. Bare-root tree seedlings were planted on a sloping site, half of which was graded and tracked-in using standard operational practices, and half of which was left ungraded. Minesoil compaction due to grading was immediately apparent during tree planting. Even during the early spring when soils were relatively moist, it was difficult to make an adequate hole with the tree planting bar on the graded slope. On the ungraded slope, however, deep planting holes were easy to open. Survival and growth was much better on the ungraded slope. Overall, tree survival on the ungraded slope was nearly 70% compared to 42% on the graded and, tracked-in slope (Table 1). The effects of compaction were most severe on the hardwood species. All the black walnuts and sugar maples survived on the ungraded slope, but only about half of both species survived on the graded slope. Seventy percent of the sycamores survived, and some grew to more than four feet tall in two years on the ungraded slope; none survived on the compacted slope. Unfortunately, this study was removed during its third season by re-mining; however, it was observed that compaction effects on growth continued to be very dramatic during the third year. Red oak grew as much as two feet during the third spring on the uncompacted area, but only several inches on the compacted area.

Some compaction is inevitable due to the return-to-contour requirements of SMCRA. On long steep slopes, coal companies must grade and track-in sites to stabilize the site and ensure a uniform ground cover. On these areas there may be no alternative to current reclamation practices. On level, gently sloping, or short slopes, however, where compaction is not required for slope stabilization, grading should be minimized and tracking-in should be eliminated when forestland is the designated post-mining land use. Tree planting will be easier, survival will be greater, and better long-term tree growth and forest development will occur.

## Revegetation

Another aspect of post-SMCRA reclamation that conflicts with tree establishment efforts is dense herbaceous ground covers. Forage species commonly used to create hayland/pasture (Ky-31, clovers, etc.) are often not appropriate for reclamation of forest land. If they are successfully established, they grow too tall and dense for trees to emerge. On the other hand, when oxidized, acidic sandstones are used as a topsoil substitute, the ground cover is sometimes too sparse to satisfy ground cover requirements for bond release. When trees are to be planted, a tree-compatible ground cover should be used which protects the surface from erosion but still protects tree establishment (Vogel, 1981).

Our research has shown that a ground cover can be established which provides 90% cover during the first year, protects the surface from erosion, and does not excessively compete with trees (Torbert et al. 1986a,b). Tree compatible ground cover species that have performed well in several research projects and for operational reclamation include annual grasses such as foxtail millet (Setaria italica), rye (Secale cereale), or wheat (Triticum aestivum); less-competitive perennial grasses such as perennial ryegrass (Lolium perenne), redtop (Agrostis gigantea), orchardgrass (Dactylis glomerata), and weeping lovegrass (Eragrostis curvula); and legumes such as birdsfoot trefoil (Lotus corniculatus), kobe lespedeza (Lespedeza striata var 'Kobe') and Appalow lespedeza (Lespedeza cuneata var 'Appalow'). Vogel (1981) recommends a mixture of weeping lovegrass (2 lbs/acre), orchardgrass (5 lbs/acre), kobe lespedeza (10 lbs/acre), and Appalow lespedeza (8 lbs/acre).

These species have been successfully established with seeding and fertilization rates that are lower than typically used for reclamation. Unlike standard hayland/pasture ground covers that are lush during the first year and gradually decline without additional fertilizer, tree-compatible covers that we have tested are comparatively sparse during the first year and become increasingly lush with time. Despite relatively low seeding rates, ground cover establishment during the first year can satisfy the 90% cover requirement for partial bond reduction. Most of this first year-cover results from the annual grasses, while the legumes dominate after several years. In our trials, when birdsfoot trefoil and Appalow lespedeza were seeded at rates of 5 to 10 lbs/Ac, they started slowly by producing only a few plants per square foot which were generally less

Table 1. Tree survival and growth after two years on a minesoil in southwest Virginia as affected by surface grading compaction.

| Species | Survival | | Tree Height | | Height Response |
|---|---|---|---|---|---|
| | Graded & Tracked-in | Ungraded | Graded & Tracked-in | Ungraded | |
| | (%) | | (in) | | (%) |
| Black walnut | 55 | 100 | 17 | 22 | 29 |
| Red oak | 40 | 62 | 17 | 17 | 0 |
| Sugar maple | 60 | 100 | 23 | 46 | 100 |
| Sycamore | 0 | 69 | - | 51 | -- |
| Virginia pine | 63 | 42 | 18 | 22 | 22 |
| White pine | 37 | 43 | 10 | 17 | 70 |
| | -- | -- | -- | -- | -- |
| Average | 42 | 69 | 17 | 29 | 44 |

than six inches tall after the first season. However, by the third season, they developed into a complete ground cover. Appalow, a relatively new variety of sericea lespedeza, sprawls along the ground rather than growing upright. It has been established on sites with a pH as low as 4.0 and can rapidly invade surrounding bare areas. Despite the aggressiveness of Appalow and birdsfoot trefoil, their short stature during the first few years after seeding allows tree seedlings to grow above them. These legumes will persist beneath the trees, increasing soil nitrogen levels for several years, until they are eventually shaded out.

Another advantage of seeding these tree-compatible species at relatively low seeding rates is that native plant species can more easily invade the site. Dense covers of tall fescue can effectively prevent the establishment of native plants (Wade, 1989). In our ground cover experiments, we observed that native herbaceous and tree species were abundant, even when native topsoil was not replaced. Many of these plants persist in the ground cover above the legumes adding to the diversity of plant species and creating a plant community that resembles a natural forest. Lastly, a problem associated with fescue is its alleleopathic effect on tree growth (Walter and Gilmore 1976, Todhunter and Beineke 1979). This problem can be avoided by using other grass species.

## Tree Establishment

After a tree-compatible ground cover is established, tree seedlings can be planted. We use two general categories of trees; "crop trees" and nitrogen-fixing "nurse trees". Crop trees are long-term species that offer economic value to the landowner. Nurse trees are short-lived nitrogen-fixing trees or shrubs that are primarily planted to enhance the N status of the soil and help achieve the minimum number of stems required for bond release. Most nurse trees also provide food and cover for wildlife. Nurse trees are short lived and begin to drop out of the tree stand after 15-20 years, providing additional space for the crop trees as they develop.

### Crop tree establishment.

Many important hardwood species (oaks, ash, walnut, yellow-poplar, etc.) will grow on properly constructed minesoils. Ashby et al. (1984) reported that 30-year-old white oaks growing on a minesoil in Illinois were the fastest growing white oaks ever reported in the state. Similarly, yellow-poplar and black walnut grew very well. As a general rule, however, pines are easier and less expensive to establish than hardwood crop tree species. Pines can be harvested in fewer years than most hardwood species and offer more immediate economic opportunities for landowners.

In recent years, the most widely planted pine in the Central Appalachian coal fields has been Eastern white pine. On good sites in the Appalachians, white pine produces more timber per acre than any other species (Doolittle 1958) and can be harvested in thirty years. On properly constructed minesoils, white pines can reach ten feet tall after five years. White pine produces the most merchantable volume when planted on a wide spacing (Balmer and Williston, 1983). For sawlog production on minesoil, white pine should be planted on a 10 x 10 ft to 12 x 12 ft spacing (300-450 trees/acre). If trees are planted more densely, the stand might stagnate after crown closure and result in many low-value pulpwood-sized trees at an otherwise harvestable age.

Other pine species that have performed well in our studies include Virginia pine, loblolly pine, and pitch x loblolly hybrid pine. All of these species grew more rapidly than white pine during the first five years. These species should be planted more densely than white pine. A 6 x 8 ft spacing to an 8 x 8 ft spacing (680-900 trees/acre) is best. There is some concern that loblolly pine may not perform well over the long term because the coal fields are outside the species' natural range. The pitch x loblolly hybrid, which has the growth rate of loblolly pine and the cold-hardiness of pitch pine, has performed very well in our studies. Unfortunately, there is currently no commercial

source of hybrid pine seedlings for operational reclamation.

Nurse tree species selection.

Nitrogen-fixing trees and shrubs can increase soil nitrogen levels and improve the growth of interplanted crop trees (Plass, 1977; Vogel 1981, Jencks 1982, Vimmerstedt et al. 1989) Nitrogen-fixing trees appropriate for minesoils in the Central Appalachians include black locust (Robinia psuedoacacia L.), European black alder (Alnus glutinosa (L.) Gaertn.) and redbud (Cercis canadensis L.). Suitable shrub species include bristly locust (Robinia fertilis Ashe), autumn olive (Elaeagnus umbellata Thunb.), and bicolor lespedeza (Lespedeza bicolor Turcz.). Vogel (1981) recommends planting nurse species in every second or third row between crop trees.

Tree Planting Techniques.

A major reason trees die during the first year is due to mishandling before or during planting. Common problems that lead to seedling mortality are: 1) storing seedlings in warm places which breaks their dormancy, 2) allowing trees to dry out during storage or in the field during planting, 3) excessive root pruning, 4) shallow planting or 5) improper planting (not closing the planting hole).

Soil compaction is a contributing cause of poor planting techniques. It is difficult to make a deep planting hole in compacted soils. Because of this, trees are either improperly planted or severely root-pruned so that the root will fit in a small hole. These problems must be minimized by properly constructing the minesoil and properly training and supervising the planting crews. In Virginia, some coal companies have been able to obtain contracts with tree planting crews that guarantee minimum levels of survival.

Direct seeding trees.

We have experimented with direct seeding several pine and nurse tree species. Virginia pine and loblolly pine have been successfully established in several studies (Torbert et al. 1986b). These species can be established at seeding rates of 2 lbs/acre when used in conjunction with a tree-compatible ground cover and nitrogen fertilizer rates below 60 lbs N/acre. The cost of pine seed, however, is fairly high so that the cost of establishing these trees by direct seeding is probably not less than hand planting seedlings. White pine has been a difficult species to establish by direct seeding. White pine growth during the first season is slow, which makes them subject to frost-heaving during the first winter and overtopping by the ground cover during the second summer. Black locust and autumn olive have been successfully established in our studies and we have seen other sites where black alder and bicolor lespedeza were established with an herbaceous ground cover. Most nurse tree and shrub species have inexpensive seed; therefore, it may be most economical to direct seed the nurse species and handplant the crop trees.

Conclusions

Despite the fact that forestry is the most logical land use for much of the surface mined land in the Central Appalachians, most reclamation laws, practices, and guidelines in this region seem to have been designed to reclaim mined land for hayland or pasture with Kentucky-31 tall fescue and other forage species. An important concept that is not well understood by coal companies, regulatory agencies and many researchers is that in order to maximize land use productivity in accordance with PL 95-87, reclamation techniques to reclaim forest land should be different than tehcniques for reclaiming hayland/pasture.

To create productive forest land, a good minesoil for trees must be created during reclamation. An overburden material should be selected for placement at the surface that will maximize tree performance. When forest land is the designated land use, grading should be minimized. A tree-compatible ground cover should be used that will grow well on the site but not hinder tree establishment. On properly reclaimed sites, trees can be planted with ease, and survival and long-term growth will be ensured.

Literature Cited

Ashby, W.C., W.G. Vogel, C.A. Kolar, and G.R. Philo. 1984. Productivity of stony soils on strip mines. p. 31-44. In J.D. Nichols et al. (Ed) Erosion and productivity of soils containing rock fragments. Spec. Pub. No. 13. Soil Sci. Soc. Am. Madison, WI.

Balmer, W.E. and H.L. Williston. 1983. Managing eastern white pine in the southeast. USDA For. Serv. Forestry Rep. R8-FR 1. USDA For. Serv. Southeastern Area, State and Private Forestry, Atlanta, GA.

Beck, D.E., 1971. Polymorphic site index curves for white pine in southern Appalachians. USDA For. Serv. Res. Pap. SE-80. Southeastern For. Exp. Stn. Asheville, NC.

Doolittle, W.T. 1958. Site index comparisons for several forest species in the Southern Appalachians. Soil Sci. Soc. Proc. 22:455-458.

Jencks, E.M., E.H. Tyron, and M. Contri. 1982. Accumulation of nitrogen in minesoils seeded to black locust. Soi Sci Soc Am J. 46:1290-1293.

Larson, M.M. and J.P. Vimmerstedt. 1983. Evaluation of 30-year-old plantations on stripmined land in east central Ohio. Research Bulletin No. 1149. Ohio Agric. Res. and Dev. Center. Ohio State Univ. Wooster Ohio. 20pp.

Plass, W.T. 1977. Growth and survival of hardwoods and pine interplanted with European alder. USDA For. Serv. Res. Pap. NE-376. 10pp.

Plass, W.T. 1979. The use of southern pines for surface mine reclamation. pp 176-182. In H.L. Williston and W.E. Balmer (Eds). Proceedings Symposium for the Management of Pines of the Interior South. Nov 7-8, 1978, Knoxville, TN. USDA For. Serv. Southeastern Area State and Private Forestry. Tech. Pub. SA-TP2. mycorrhizae on pines seeded in greenhouse trays. J. Environ. Qual. 13:387-391.

Roberts,J.A., W.L. Daniels, J.C. Bell, and D.C. Martens. 1988. Tall fescue production and nutrient status on southwest Virginia mine soils. J. Environ. Qual. 17:55-62.

Schoenholtz, S.H., and J.A. Burger. 1984. Influence of cultural treatments on survival and growth of pines on strip-mined sites. Reclamation and Revegetation Research. 3:223-237.

Schoenholtz, S.H., J.A. Burger, and J.L. Torbert. 1987. Natural mycorrhizal colonization of pines on reclaimed surface mines in Virginia. J. Environ. Qual. 16:143-146.

Todhunter, M.N. and W.F. Beineke. 1979. Effect of fescue on black walnut growth. Tree Planters' Notes 30:20-23.

Torbert, J.L., J.A. Burger, J.N. Lien, and S.H. Schoenholtz. 1985. Results of a tree species trial on a recontoured surface mine in southwestern Virginia. Southern J. Applied Forestry 9:150-153.

Torbert, J.L., J.A. Burger, and T.J. Nichols. 1986a. Reforestation seed mixtures for hydroseeding reclaimed mined land. Va. Coop. Ext. Serv. Pub. 460-112. VPI&SU, Blacksburg VA. 6pp.

Torbert, J.L., J.A. Burger, and R.E. Preve. 1986b. The effect of ground cover species and nitrogen fertilizer rate on the successful establishment of hydroseeded trees. pp83-87. In D.H. Graves (Ed) Proceedings 1986 Symposium on Mining, Hydrology, Sedimentology, and Reclamation. Dec 8-11 1986. Univ. KY, Lexington, KY, UKY BU142.

Torbert, J.L., A.R. Tuladhar, J.A. Burger, and J.C. Bell. 1988. Minesoil property effects on the height of ten-year-old white pine. J. Environ. Qual. 17:189-192.

Torbert, J.L., J.A. Burger, and W.L. Daniels. 1990 a . Pine growth variation associated with overburden rock type on a reclaimed surface mine in Virginia. J. Environ. Qual. 19:88-92.

Torbert, J.L and J.A. Burger. (1990b). Comparison of tree survival and and growth on a graded and ungraded mine soil after two years. Tree Planters' Notes (In press).

Vimmerstedt, J.P., M.C. House, M.M. Larson, J.D. Kasile, and B.L. Bishop. 1989. Nitrogen and carbon accretion on Ohio coal minesoils: influence of soil-forming factors. Landscape and Urban Planning. 17:99-111.

Vogel, W.G. 1981. A guide for revegetating coal minesoils in the eastern United States. USDA For. Serv. Gen. Tech. Rep. NE-68. 190 pp.

Wade, G.L. 1989. Grass competition and establishment of native species from forest soil seed banks. Landscape and Urban Planning. 17:135-149.

Walter, D.T., and A.R. Gilmore. 1976. Allelopathic effects of fescue on the growth of sweetgum. J. Chem. Ecol. 2:469-479.

# INFLUENCE OF REFORESTATION RECLAMATION TREATMENTS ON NITROGEN DYNAMICS IN MINESOILS[1]

Stephen H. Schoenholtz, James A. Burger, and Richard E. Kreh[2]

Abstract. Reforestation of surface-mined land in the Appalachian region of the United States offers an environmentally and economically attractive reclamation alternative. However, reestablishment of productive forest ecosystems may be limited by minesoil nutrient deficiencies. Reclamation treatments which promote nutrient cycling are required to overcome these limitations. A tank lysimeter study was established to simulate field conditions for evaluation of reforestation reclamation treatments on nitrogen cycling in a minesoil from southwestern Virginia. The treatments were replacement of natural topsoil, yellow-poplar whole-tree chip amendment, and no amendment; each had 0 or 100 kg nitrogen $ha^{-1}$ applied. Lysimeters were then seeded with grass-legume reforestation groundcover mixtures in July 1987, and one-year-old pitch x loblolly hybrid pine seedlings were planted in March 1988. Results after three growing seasons suggest that inorganic nitrogen fertilizer effects on soil, leachate, and plant nitrogen cycling dynamics are rapid but temporary. In contrast, the topsoil and wood-chip organic amendments have provided sources of organic nitrogen and carbon which should influence the nitrogen dynamics of this minesoil system in a slower, more effective and longer lasting manner.

Additional Key Words: organic amendments, southwestern Virginia, lysimeters, ecosystem restoration, Pinus rigida x taeda.

## Introduction

The overall objective of most surface-mine reclamation is restoration of a stable, self-sustaining ecosystem that will minimize long-term environmental degradation and support the desired post-mining land use. Reforestation of surface-mined land in the Appalachian region of the United States is one land use that offers an environmentally and economically attractive reclamation alternative.

Restoration of productive forest ecosystems may be limited by minesoil nutrient deficiencies, particularly nitrogen (N). Mays and Bengtson (1978) reported that an inspection of revegetated surface mines in the eastern United States revealed the need for maintenance N fertilization in addition to the initial application at establishment. A lack of N in minesoils reported by others, including Czapowskyj (1973), Plass and Vogel (1973), Vogel (1981), Schoenholtz and Burger (1984), Reeder (1988), and Roberts et al. (1988), suggests that current reclamation programs do not always meet the goal of reestablishing an adequate N cycling capacity which would promote N accumulation, biomass production, and ecosystem stability.

Heavy applications of soluble fertilizers to recently exposed overburden materials are often used to overcome low N levels and to promote establishment of vegetation. However, in contrast to intact terrestrial ecosystems which have evolved stable nutrient cycling mechanisms, minesoils may not retain adequate amounts of the applied fertilizer, resulting in low use efficiency and in the potential for rapid N movement into adjacent surface water and groundwater. Stabilization of reclaimed mine sites with a desirable plant community that requires no long-term fertilizer inputs to maintain a satisfactory level of productivity is

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]Stephen H. Schoenholtz, James A. Burger, and Richard E. Kreh are Graduate Project Assistant, Associate Professor of Forest Soils, and Research Associate, respectively, Department of Forestry, Virginia Polytechnic Institute and State University, Blacksburg, VA 24061.

a major reclamation goal. The development of an active biological N cycle is necessary to realize this goal.

One of the first priorities in reestablishing N cycling in minesoil is the development of an organic-N reservoir through the establishment of an organic-matter pool. This has been accomplished by using organic amendments such as sawdust, wood residues, and sewage sludge (Moss et al., 1985, Schuman and Sedbrook 1984, Seaker and Sopper 1984). The establishment and maintenance of an organic-matter pool should provide an initial immobilization sink for fertilizer N via microbial uptake followed by slow release of inorganic N until an equilibrium between N immobilization and mineralization is reached.

Soil microbial activity is an intricate component of N cycling. Numbers and diversity of microorganisms are lower in minesoils than in most native soils (Wilson 1965, Stroo and Jencks 1982). Topsoil replacement is one means by which microbial populations can be reestablished. Furthermore, topsoil can provide a good rooting medium, improved infiltration, decreased runoff, increased vegetation species diversity, and a source of mineralizable N to the system.

Although mine operators in the Appalachian coalfields commonly stockpile and replace native topsoil as required by reclamation regulations, the value of this procedure has been questioned in steeply sloped areas with thin, infertile soils. At many sites, replacing native topsoil is time consuming and expensive. Improved mining techniques which isolate substitute overburden materials have been implemented and may provide equal or superior productivity compared with some native soils (Daniels and Amos 1985). Additional observations and quantification of the effect of topsoil additions on N dynamics would help assess the usefulness of this reclamation alternative.

Reliable guidelines for effective reforestation reclamation practices require a detailed understanding of the fundamental ecological processes occurring within the minesoil system. Assessments are not available comparing the effects of N fertilization, native topsoil replacement, and whole-tree wood-chip amendment on N cycling dynamics in Appalachian minesoils. The objective of this study was to evaluate the effects of these reclamation alternatives on selected N cycling dynamics in a minesoil from southwestern Virginia.

## Materials and Methods

To simulate field conditions, 18 concrete lysimeters (2.4 by 1.2 by 0.8 m deep) were set into native soil at the Reynolds Homestead Agricultural Experiment Station in Patrick County, Virginia. The lysimeters were designed to insure drainage and leachate collection. New minesoil from sandstone and siltstone overburden strata of the Wise formation in Wise County, Virginia was weighed and placed in the lysimeters. Minesoil bulk density within each lysimeter was calculated based on lysimeter volume and minesoil weight.

The reclamation treatments were the replacement of 5 cm of native topsoil, the addition of 8 cm of yellow-poplar whole-tree chips, and no organic amendment; each had 0 or 100 kg N $ha^{-1}$ applied as $NH_4NO_3$. All lysimeters also received 100 kg phosphorus (P) $ha^{-1}$ and 60 kg potassium (K) $ha^{-1}$. Selected properties of these organic amendments are presented in Table 1. Prior to fertilization and seeding, organic amendments were tilled into the minesoil to a depth of approximately 25 cm. Selected properties of the minesoil after tilling are presented in Table 2.

Treatments were in place by early July 1987. At this time, a mixture of grass and legume species was broadcast over the surface of each lysimeter (Table 3).This mixture was selected for its compatability with tree establishment (Torbert et al., 1986). Ten 1-0 pitch x loblolly hybrid (*Pinus rigida* x *taeda*) pine seedlings were planted in each lysimeter in March 1988. Pitch x loblolly pine was selected because of its desirable growth habit and wood quality.

Composite minesoil samples from 20 randomly located subsamples were collected from 0-15 cm depth in each lysimeter at intervals for the duration of the study. Total Kjeldahl N and KCl-extractable $NH_4$+$NO_3$ representing total soil N and plant-available inorganic soil N pools, respectively, were analyzed. Available P was extracted with 0.5 N $NaHCO_3$ (pH 8.5). Available K, calcium Ca), and magnesium (Mg) were extracted with 1 N $NH_4OAc$ (pH 7.0).The volume and inorganic-N ($NH_4$+$NO_3$) content of leachate draining from each lysimeter was measured after each precipitation event.

Herbaceous vegetation was sampled at the end of each growing season using two randomly located 0.10 $m^2$ clip-plots in each lysimeter for dry-weight yield and N content. Survival, height, and ground-line diameter of the pine seedlings were measured after each growing season. A pine stem-volume index calculated from height x $diameter^2$ was also derived.

The study was a completely randomized design with three types of organic amendment (control, topsoil, and wood chips) and two levels of fertilizer (control and 100 kg N $ha^{-1}$). The six possible treatment combinations were replicated three times. Analysis of variance followed by Duncan's multiple range tests for mean separations were used to compare treatment effects on response variables. Percent survival values were statisically compared following an arcsine transformation.

## Results and Discussion

### Total Soil N

Total soil N levels were not affected by the fertilizer treatment. However, the topsoil amendment significantly increased this N pool

Table 1. Selected properties of the organic amendments.

| Amendment | pH | %C | %N | C/N | Dry wt. added | C added | N added |
|---|---|---|---|---|---|---|---|
| | | | | | ----------kg ha$^{-1}$---------- | | |
| Topsoil[1] | 5.0 | 3 | .20 | 15/1 | 500,000 | 15,000 | 1,000 |
| Wood Chips | - | 47 | .18 | 261/1 | 50,000 | 23,500 | 90 |

[1]Predominately A horizon of Jefferson series, loamy typic hapludult.

Table 2. Selected chemical and physical properties of the upper 15 cm of minesoil following addition of organic amendments.

| Amendment | Total N | Available P | Available K | Available Ca | Available Mg | Bulk Density |
|---|---|---|---|---|---|---|
| | ----------mg kg$^{-1}$---------- | | | | | --Mg m$^{-3}$-- |
| Control | 565 | 2.7 | 52 | 654 | 211 | 1.84 |
| Topsoil | 838 | 3.6 | 73 | 768 | 234 | 1.88 |
| Wood Chips | 568 | 2.0 | 59 | 625 | 208 | 1.64 |

Table 3. Reforestation groundcover seed mixture.

| Species | Scientific Name | Rate |
|---|---|---|
| | | kg ha$^{-1}$ |
| Foxtail millet | Setaria italica | 8.4 |
| Perennial ryegrass | Lolium perenne | 8.4 |
| Annual ryegrass | Lolium multiflorum | 8.4 |
| Redtop | Agrostis gigantea | 5.1 |
| Birdsfoot trefoil | Lotus corniculatus | 8.4 |
| Korean lespedeza | Lespedeza stipulacea | 8.4 |
| 'Appalow' Sericea lespedeza | Lespedeza cuneata cv. "appalow" | 16.8 |

throughout the course of the study (Figure 1). The decrease in total soil N which occurred with the wood-chip amendment at the start of the study was a result of decreased bulk density under this treatment and not a function of lower N concentration (Table 2). By the end of the first growing season (October 1987) the wood-chip treatment total N value was comparable to the control. These results indicated that a single fertilizer application had no effect on total N in the soil during the first three years. In contrast, topsoil replacement increased this pool and may provide a long-term source of N for ecosystem restoration.

Extractable Inorganic N

Potassium chloride-extractable inorganic N provides an index of N available for plant uptake. Fertilization significantly increased extractable inorganic N during the first three months of the study (Figure 2). This increase in extractable inorganic N was beneficial during the critical period of plant establishment when promotion of groundcover was a primary objective. Fertilization did not result in an increase in extractable inorganic N beyond this initial period.

The topsoil amendment also significantly increased extractable inorganic N levels during the initial groundcover-establishment stage (Figure 3). The lack of a wood-chip effect on extractable inorganic N suggests that immobilization of fertilizer N was not a significant process under this treatment. This may be a function of rapid fertilizer loss in leachate N or rapid plant uptake of fertilizer N.

Leachate N

High levels of available N during the initial vegetation-establishment stage of reclamation can result in degradation of off-site water quality if this N is not retained in the plant-soil system. Inorganic N levels measured in the leachate indicate that N loss totaled 17.5 and 1.9 kg N $ha^{-1}$ for fertilizer and control, respectively, during the first growing season (Table 4). In contrast, leachate N loss during the second growing season was not affected by fertilization. The initial difference between the two fertilizer treatments illustrates the potential risk to off-site water quality when a single fertilizer application is used prior to vegetation establishment.

The quantity of inorganic N in the leachate was not significantly affected by the organic amendments. This suggests that any short-term effects of these treatments on net N mineralization were probably precluded by rapid plant uptake and by rapid initial fertilizer N leaching.

Aboveground Biomass

The development of abundant, high-quality aboveground biomass for site protection, nutrient retention, and post-mining land use is the primary concern of reclamation programs. Fertilization increased the dry-weight yield and total N of the aboveground herbaceous biomass after the first growing season (Table 5). This supports the hypothesis that fertilizer N dynamics were strongly influenced by both plant uptake and leaching during the first growing season. However, by the second growing season, fertilization effects on herbaceous biomass were absent.

The wood-chip amendment decreased dry-weight yield and total biomass N after the first and third growing seasons (Table 5). This treatment caused an initial phytotoxicity to the grass species in the revegetation mix resulting in only legume establishment during the first growing season. The topsoil amendment increased total biomass N after the first growing season, but did not differ from the control in subsequent years. The relatively stable nature of the organic carbon (C) and organic N provided by the two organic amendments may provide a long-term treatment effect on N cycling dynamics.

Tree Performance

First- and second-year pitch x loblolly hybrid pine survival were highest in the fertilized treatments (Table 6). However, stem-volume index was not significantly affected by this treatment.

After two growing seasons, pine survival and stem-volume index were highest with the wood-chip amendment (Table 6 and Figure 4). This may be directly related to the lower level of herbaceous competition and improved soil moisture relationships resulting from this treatment. Aboveground herbaceous biomass development was slowest and competitive grasses were minimal with the wood-chip amendment.

## Conclusions

Early results of this long-term minesoil reclamation study suggest that inorganic N fertilizer effects on soil, leachate, and plant N cycling dynamics are rapid but temporary. In contrast, the topsoil and wood-chip organic amendments have provided sources of organic N and C which should influence the N dynamics of this minesoil system in a slower, more effective, and longer lasting manner.

The two organic amendments used in this study are often available in the southern Appalachian coalfields. If productive post-mining ecosystem restoration is a reclamation objective, then amendments which produce more stable changes in N cycling dynamics than inorganic N applications may be required.

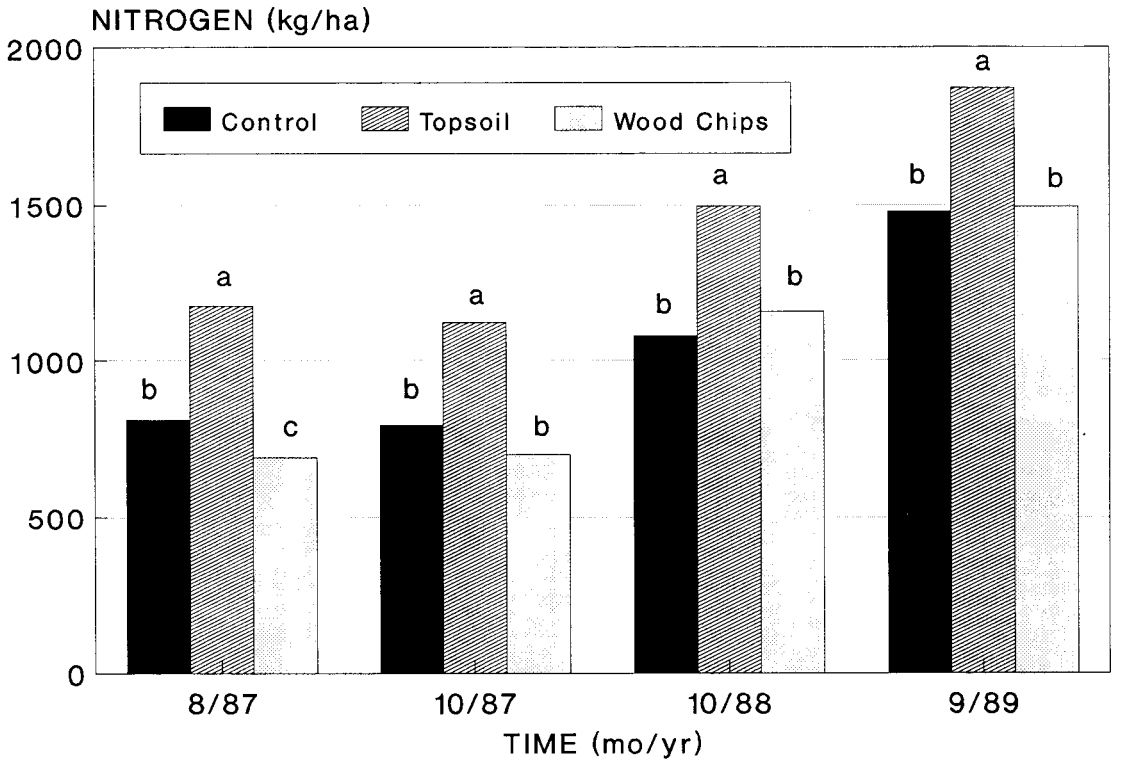



Figure 1. Effect of organic amendments on total minesoil N. For each date, bars with different letters are significantly different at the 0.05 level.

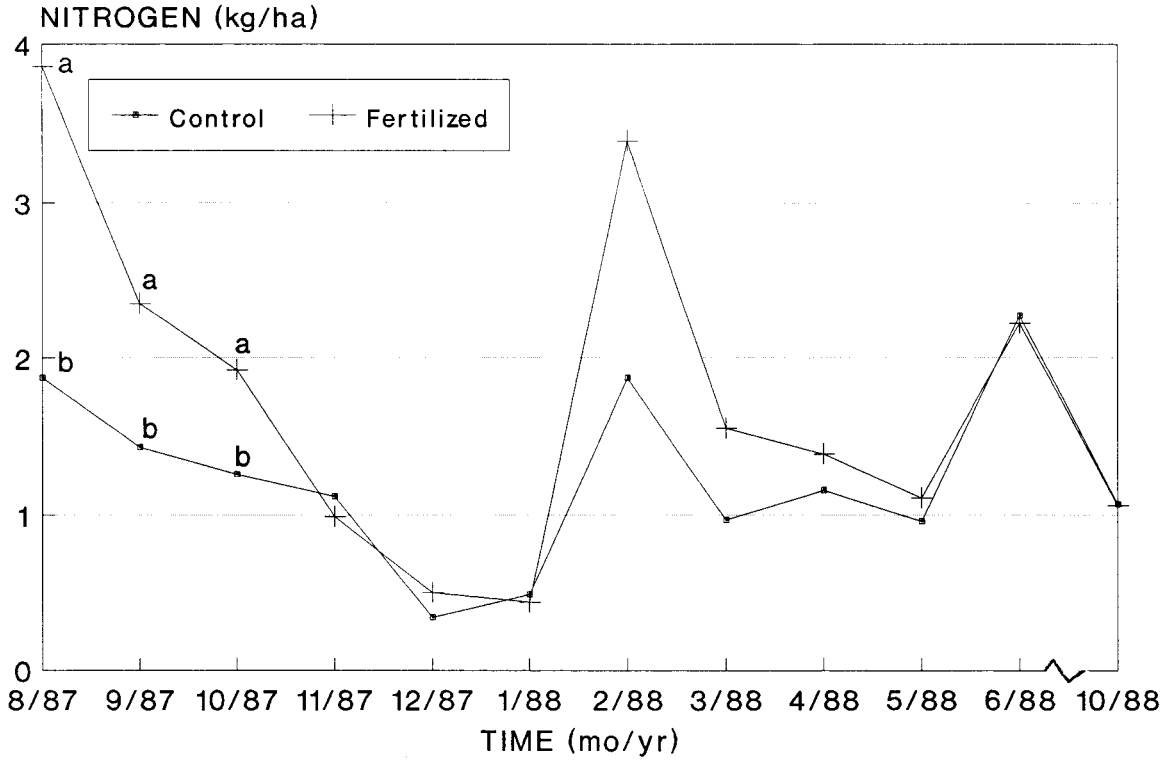



Figure 2. Effect of fertilization on KCl-extractable N. For each date, points with different letters are significantly different at the 0.05 level.

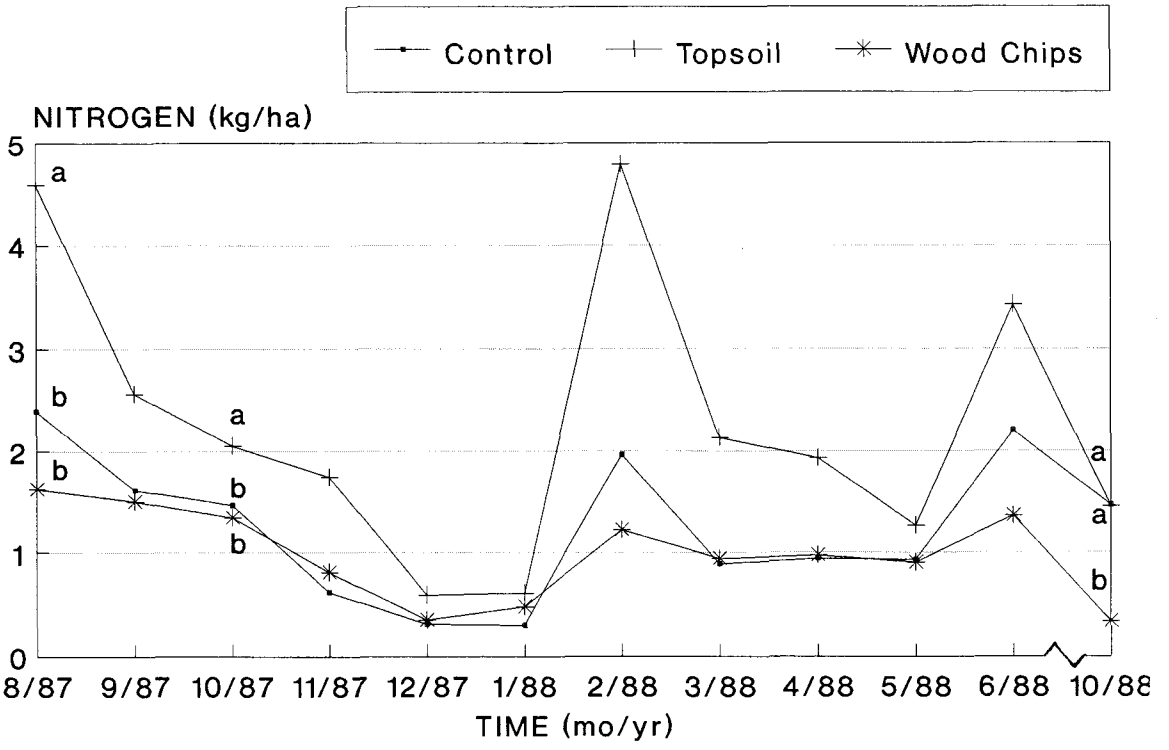



Figure 3. Effect of organic amendments on KCl-extractable KCl-extractable N. For each date, points with different letters are significantly different at the 0.05 level.

Table 4. Effect of fertilization on cumulative leachate N loss.[1]

| Fertilizer treatment | Time Period 7/87-10/87 | 11/87-10/88 | Total |
|---|---|---|---|
| | ----------kg N ha$^{-1}$---------- | | |
| Control | 1.9b[2] | 0.9a | 2.8b |
| Fertilized | 17.5a | 1.3a | 18.8a |

[1]Pooled across organic amendment treatments.

[2]Values within a column followed by different letters are significantly different according to Duncan's multiple range test at the 0.05 level.

Table 5. Effect of reclamation treatments on aboveground herbaceous biomass and N.

| Treatment | 10/87 | | 10/88 | | 10/89 | |
|---|---|---|---|---|---|---|
| | dry weight | N | dry weight | N | dry weight | N |
| | kg ha$^{-1}$ | | | | | |
| Organic amendment | | | | | | |
| Control | 4108a[1] | 52.7b | 4564a | 102.8a | 5998a | 135.7a |
| Topsoil | 5684a | 70.2a | 4106a | 94.1a | 6670a | 153.4a |
| Wood chips | 1109b | 32.4c | 4583a | 93.6a | 5043b | 101.3b |
| Fertilizer | | | | | | |
| Control | 2518a | 38.2b | 4936a | 110.4a | 5801a | 131.3a |
| Fertilized | 4749b | 65.3a | 3900a | 83.2a | 6006a | 129.0a |

[1]Values within a column for organic amendments or for fertilizer treatments followed by different letters are significantly different according to Duncans's multiple range test at the 0.05 level.

Table 6. Effect of reclamation treatments on pitch x loblolly hybrid pine survival.

| Treatment | 1st-year survival | 2nd-year survival |
|---|---|---|
| | % | |
| Organic amendment | | |
| Control | 87ab[1] | 83ab |
| Topsoil | 73b | 60b |
| Wood chips | 98a | 98a |
| Fertilizer | | |
| Control | 76b | 71b |
| Fertilized | 97a | 90a |

[1]Values within a column for organic amendments or for fertilizer treatments followed by different letters are significantly different according to Duncans's multiple range test at the 0.05 level.

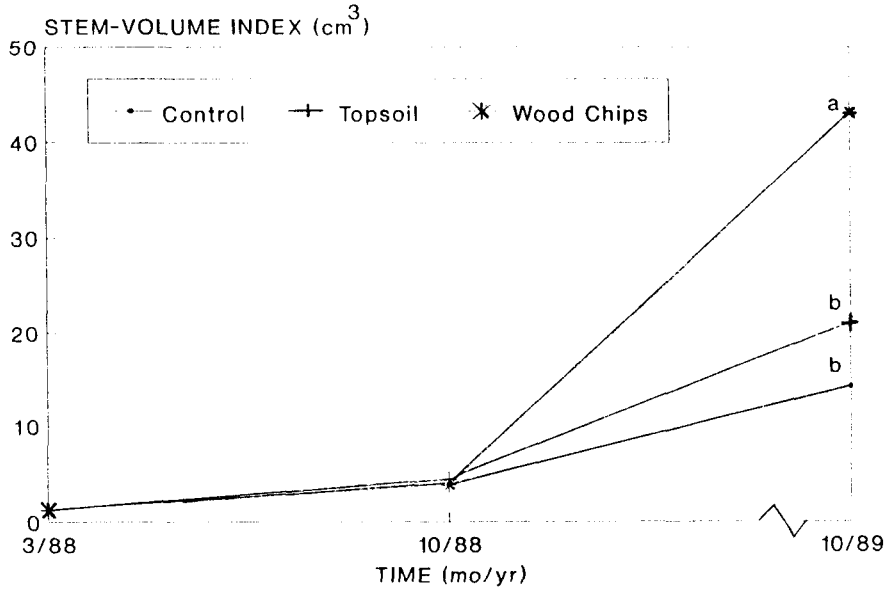



Figure 4. Effect of organic amendments on stem-volume index of pitch x loblolly hybrid pines. Points with different letters are significantly different at the 0.05 level.

Literature Cited


Czapowskyj, M.M. 1973. Establishing forest on surface-mined land as related to fertility and fertilization. p. 132-139 In Forest Fert. Symp. Proc. USDA For. Serv. Gen. Tech. Rep. NE-3.

Daniels, W.L., and D.F. Amos. 1985. Generating productive top soil substitutes from hard rock overburden in the southern Appalachians. Env. Geochem. & Health 7:8-15.

Mays, D.A., and G.W. Bengtson. 1978. Lime and fertilizer use in land reclamation in humid regions. p. 397-428. In F.W. Schaller and P. Sutton (ed.) Proc. Symp. Reclam. of Drastically Disturbed Lands. Wooster, OH.

Moss, S.A., Burger, J.A., and W.L. Daniels. 1985. The effects of organic amendments on nitrogen availability. p. 94-103. In C.A. Kolar (ed.) Proc. Symp. of Better Reclam. with Trees Conf. Carbondale, IL.

Plass, W.T., and W.G. Vogel. 1973. Chemical properties and particle size distribution of 39 surface-mine spoils in southern West Virginia. USDA For. Serv. Res. Pap., NE-276.

Reeder, J.D. 1988. Transformations of nitrogen-15-labeled fertilizer nitrogen and carbon mineralization in incubated coal mine spoils and disturbed soil. J. Environ. Qual. 17:291-298.

Roberts, J.A., W.L. Daniels, J.C. Bell, and D.C. Martens. 1988. Tall fescue production and nutrient status of southwest Virginia mine soils. J. Environ. Qual. 17:55-62.

Schoenholtz, S.H., and J.A. Burger. 1984. Influence of cultural treatments on survival and growth of pines on strip-mined sites. Reclam. Reveg. Res. 3:223-237.

Schuman, G.E., and T.A. Sedbrook. 1984. Sawmill wood residue for reclaiming bentonite spoils. For. Prod. J. 34:65-68.

Seaker, E.M., and W.E. Sopper. 1984. Reclamation of bituminous strip mine spoil banks with municipal sewage sludge. Reclam. Reveg. Res. 3:87-100.

Stroo, H.F., and E.M. Jencks. 1982. Enzyme activity and respiration in mine spoils. Soil Sci. Soc Am. J. 10:369-371.

Torbert, J.L., J.A. Burger, and T.J. Nichols. 1986. Reforestation mixtures for hydroseeding reclaimed mined land. Virginia Coop. Ext. Serv. Pub. 460-112. Virginia Polytech. Inst. and State Univ. Blacksburg, VA. 6 pp.

Vogel, W.G. 1981. A guide for revegetating coal mine soils in the Eastern United States. USDA For. Serv. Gen. Tech. Report, NE-68.

Wilson H.A. 1965. The microbiology of strip-mine spoil. W. Va. Agr. Exp. Sta. Bull. 506T.

# FACTORS LIMITING TREE GROWTH IN SOUTHERN ILLINOIS UNDER SMCRA[1]

by

W. Clark Ashby[2]

Abstract. Tree survival and height growth of green/white ash (Fraxinus pennsylvanica/F. americana), black walnut (Juglans nigra), and red oak (Quercus rubra) were determined in 1988-89 on three mine sites in southern Illinois. All sites had been graded, were found to be compacted, and had dense ground covers of tall fescue (Festuca arundinacea) and other herbaceous species. Trees were established in 1981 and 1982 with the ash as bare-root seedlings and the walnut and oak as seedlings and as seed. Survival and height growth were low at all three graded and compacted sites. In one study herbicide application enhanced tree performance and fertilizer had little effect. Trees grew more poorly in tall fescue than in other cover types. The other two studies were notable for animal damage to trees. Red oak was most severely damaged by voles, and black walnut by deer.

Additional Key Words: Compaction, stripmine, reclamation, reforestation, green ash (Fraxinus pennsylvanica Marsh.), black walnut (Juglans nigra L.), red oak (Quercus rubra L.), tall fescue (Festuca arundinacea Schreb., selection Ky-31), herbicide, animal damage, deer, vole, ground cover.

## Introduction

Trees have been extensively planted on mined lands, more than seventeen million in Illinois alone prior to 1968. By the time Public Law 95-87 "Surface Mining Control and Reclamation Act of 1977" (SMCRA) was passed, tree planting had declined from the major to a relatively minor post-mining land use. Under SMCRA the number of acres annually planted to trees increased greatly. Many persons with experience from the pre-law operations, however, have a perception that the tree planting under SMCRA has been less successful than earlier. Data from pre-law plantings at several ages for two of the mines in this study are available from publications and are compared to current growth and survival rates. Any such comparisons must be approximate because of differences in planting dates of thirty or more years, with associated differences in weather conditions and other growth factors.

Major impacts of SMCRA on planting sites have included grading, replacement of surface soils, and establishment of planted ground covers. Numerous studies have shown that these graded sites have high soil

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]W. Clark Ashby is Professor, Department of Plant Biology, Southern Illinois University, Carbondale, IL 62901.

Table 1. Site description, management, and planting plan for green/white ash, black walnut, and red oak at CONSOL, Freeman and Sahara. The ash planted was nominally green at CONSOL and Freeman and white at Sahara. Green ash was later reported at Sahara. Ash was planted only as seedlings, and black walnut and red oak were planted as both seed and seedlings at Freeman and Sahara.

| | Mine | | |
|---|---|---|---|
| *Site Features* | *CONSOL* | *Freeman* | *Sahara* |
| Year mined | 1976 | 1978 | 1978 |
| Equipment | Wheel and shovel | Wheel and shovel | Shovel |
| Grading | Nearly flat-dozers | 15% slope-dozers | Slight slope-dozers |
| Topsoil depth | 8+ inches | None | 10+ inches |
| Subsoil depth | 40 inches | Remined cast overburden | 16 inches |
| Bulk density | 1.8 $gcm^{-3}$ | 1.8 $gcm^{-3}$ | 1.6 $gcm^{-3}$ |
| Typical pH | 8.2 | > 7.0 | 5.0 - 7.4 |
| Available P | Low | Very low P and K | Low |
| Fertilizer | 9g tablet (22-8-2) per tree | 112 $kgha^{-1}$ (33-44-0) | As required under SMCRA |
| Ground cover in 1989 | See Table 2[1] | Tall fescue, some[2] alfalfa | Tall fescue[3] |
| Herbicides | Amitrol-T and simazine: 4-foot band | Roundup and simazine: 5-foot circle @ tree | Roundup and simazine: 5-foot circle @ tree |
| Established | 1981 | 1981, 1982 | 1981, 1982 |
| Trees/acre | 1360 | 650 | 650 |
| Row length | 100 trees | 12 trees | 50 trees |
| Plot numbers | 4 (split) | 40 | 6 |
| Total trees | 1200 | 4800 | 1300 |
| Planting | Machine | Hand | Hand |
| Material | Bare-root seedling | Seed and seedling | Seed and seedling |
| Spacing | 4' in row, 8' between rows | 2.5m (8.2') by 2.5m | 2.5m (8.2') by 2.5m |

[1]Raisanen 1982. Seedling mixes in Table 2. Wheat as cover crop. Planted 1980.
[2]Ashby et al. 1988. Seeded tall fescue, alfalfa, and orchard grass with oats as cover crop. Planted 1980.
[3]Kolar et al. 1981. Seeded tall fescue, alfalfa, orchard grass, ladino clover (Trifolium repens). Planted 1980.

bulk density and other indications of compaction from traffic by heavy equipment. Compaction has been shown both indirectly by comparing plant growth on ungraded or ripped minesoils with those graded, and directly to limit root penetration and plant growth (Josiah 1986).

This paper reports still further impacts of practices mandated under SMCRA on tree growth and survival. These include direct effects of ground cover, and indirect effect of ground cover on animal damage to trees. Another observed impact was an effect of slope position on tree growth in contrast to absence of slope effects on pre-law ungraded mined lands.

## Materials and Methods

Studies were carried out at the Consolidation Coal Company Burning Star

Mine No. 2 east of Pinckneyville in Perry County, Illinois, the Freeman United Coal Company Fidelity Mine west of DuQuoin in Perry County, Illinois, and the Sahara Coal Company Mine No. 6 west of Harrisburg in Saline County, Illinois. These studies have as a common denominator plantings of green/white ash (Fraxinus pennsylvanica Marsh./F. americana L.), black walnut (Juglans nigra L.), and red oak (Quercus rubra L.). Their performance is compared with one another on each mine site and by species among sites, and for Freeman and Sahara with pre-law plantings. The CONSOL and Sahara studies had respectively a total of 17 and 20 rows of trees per plot. We are reporting on five rows each, including black walnut and red oak planted as seed and as seedlings and green ash planted only as seedlings. Seed was collected locally, and seedlings came from the state nurseries.

The site characteristics, management, plant materials, and planting dates and methods are shown in Table 1. Soils were described at CONSOL and Sahara by a retired SCS soil surveyor (Miles 1989) and at Freeman by Norland (Ashby et al. 1988). Average bulk densities from earlier records were by Shelby tube at CONSOL and sand cone method elsewhere. Tree rows, except where herbicide was an experimental variable, were sprayed with a post-/pre-emergent herbicide mix at or soon after planting. Experimental variables included herbicide, fertilizer, and ground cover at CONSOL, and slope at Freeman. The four ground cover seed mixes at CONSOL are shown in Table 2. Because of the differences in experimental design, tests of significance between sites were not appropriate. Statistical analyses of the height data at CONSOL were carried out as ANOVA with one replication each for herbicide and fertilizer, and four replications for each ground cover. With only one datum per treatment percent survival was not statistically analyzed. The effects of slope on tree growth at Freeman were based on ANOVA with eight replications per five slope positions, and a Scheffe test as appropriate.

Table 2. Herbaceous cover mixes seeded during September 1980 at CONSOL (Raisanen 1982). The species of each mix dominant in 1988-89 is designated by *.

| Mix A | Lbs./Ac. |
|---|---|
| Perennial ryegrass (Lolium perenne) | 20 |
| Red clover (Trifolium pratense) | 14 |
| *Red top (Agrostis alba) | 6 |
| **Mix B** | |
| Orchard grass (Dactylis glomerata) | 25 |
| Korean lespedeza (Lespedeza stipulacea) | 15 |
| *Timothy (Phleum pratense) | 5 |
| **Mix C** | |
| Orchard grass (Dactylis glomerata) | 25 |
| *Birdsfoot trefoil (Lotus corniculatus) | 10 |
| Red clover (Trifolium pratense) | 10 |
| **Mix D** | |
| *Ky. 31 tall fescue (Festuca arundinacea) | 20 |
| Alfalfa (Medicago sativa) | 15 |
| Orchard grass (Dactylis glomerata) | 15 |
| Red clover (Trifolium pratense) | 10 |

Table 3. Survival and average height after 7 or 8 years of green/white ash, black walnut, and red oak on three graded, compacted mine sites with dense ground cover.

| Mine Site | Survival (%) | | | Height (cm) | | |
|---|---|---|---|---|---|---|
| | Ash | Walnut | Oak | Ash | Walnut | Oak |
| CONSOL | 62 | 45 | 41 | 309 | 130 | 201 |
| Freeman | 41 | 8 | 2 | 115 | 83 | 61 |
| Sahara | 66 | 41 | 59 | 186 | 84 | 144 |

## Results

Species performance in these studies is shown by site and species in Table 3. Ash had the greatest survival and height on all sites, while values for black walnut and red oak were substantially lower. Treatment effects were found at CONSOL. For all three tree species, increased survival and growth were found on the herbicided plots (Table 4). Fertilizer did not affect tree survival or height. Ground cover effects were significant for walnut and oak, but not for ash. Tree height, though not survival, was consistently greatest with the birdsfoot trefoil ground cover, and successively less with the redtop, timothy, and fescue cover.

Reasons for the relatively low survival and height growth at Freeman shown in Table 3 will be discussed later. Differences in growth were related to slope position (Table 5). All three species showed a strong trend for greatest survival and height growth on the lower north-facing slope. Tree heights were significantly different by the Scheffe test for the upper and lower slope positions for ash and walnut. The number of surviving red oaks was too low for an ANOVA to be appropriate.

Table 4. Tree survival and average height at CONSOL after 8 years for green ash, black walnut, and red oak as affected by herbicide, fertilizer, and ground cover.

| | | Survival (%)[1] | | | Height (cm)[2] | | |
|---|---|---|---|---|---|---|---|
| | | Ash | Walnut | Oak | Ash | Walnut | Oak |
| Herbicide: | + | 79 | 46 | 59 | 339 | 165 | 237 |
| | - | 44 | 44 | 23 | 255 | 93 | 110 |
| Fertilizer: | + | 62 | 44 | 38 | 302 | 129 | 197 |
| | - | 60 | 46 | 44 | 316 | 130 | 204 |
| Ground cover: | Trefoil | 58 | 46 | 37 | 351 | 180 | 250 |
| | Redtop | 70 | 57 | 53 | 341 | 129 | 196 |
| | Timothy | 53 | 39 | 55 | 324 | 119 | 187 |
| | Fescue | 65 | 38 | 19 | 222 | 80 | 160 |

[1]An unknown, apparently small number of trees had been removed with a tree spade for a neighboring golf course.

[2]Heights of all three species were significantly different ± herbicide. A statistically significant difference was found within the tree heights for walnut and oak related to type of ground cover.

Table 5. Slope position effects on survival and height of ash, black walnut, and red oak at Freeman on a 15° north - facing slope after 8 years.

| Position | Survival (%) | | | Height (cm) | | |
|---|---|---|---|---|---|---|
| | Ash | Black Walnut | Red Oak | Ash | Black Walnut | Red Oak |
| Top | 34 | 1 | 2 | 86cd[1] | 67ab | 53 |
| Upper slope | 51 | 4 | 1 | 107bc | 54b | 62 |
| Mid slope | 23 | 4 | 0 | 111abc | 56b | 33 |
| Lower slope | 55 | 20 | 2 | 129a | 88ab | 50 |
| Bottom | 70 | 14 | 5 | 126ab | 93a | 71 |

[1]Heights within a column that do not have letters in common are significantly different at the 0.05 level. Cell sizes were too small for ANOVA with red oak..

## Discussion

A persistent question is how well trees planted on pre-law sites grew compared to trees of equivalent age on post-SMCRA sites. Such comparisons, while highly instructive, are confounded by differences in plot designs, weather after planting, ages at which measurements were taken and reported, and other factors. Approximate comparisons are available for Freeman and Sahara (Table 6). At Freeman the 8-year heights on the pre-law ungraded banks were greater than on the post-SMCRA plots (Table 3). Survival could only be compared at three years, and was greatest on the pre-law plots. At Sahara the pre-law percent survival after 10 years was greater than that of the post-SMCRA plantings after only 8 years. The pre-law trees were taller, with substantially greater values than those extrapolated to 10 years for the post-SMCRA trees. Pre-law oaks at Freeman and Sahara, on which early measurements were not taken, formed forest stands 30 years after planting. It is doubtful that post SMCRA oak plantings will be so successful.

Of the possible reasons for the apparently better performance of the pre-law versus the post-SMCRA plantings, several seem consistently evident. One is soil compaction resulting from traffic by heavy machinery during grading operations, and consequent limited root-system development (Philo et al. 1982, Josiah 1986, Vance et al. 1987). Little is known of the effect of compaction on root-system development of eastern hardwood trees, or of rooting depths of these species. There was no reason to think root systems were limited on pre-law spoils, other than on a limited acreage of acidic or toxic materials, and roots were scarcely studied. In observations of fresh road cuts across forested mine banks, woody roots were always found at the bottom of the cuts, ten or more feet deep.Compaction effects can otherwise be recognized. One such effect is that while little or no differences in tree survival and height related to slope were found in more than 30 studies on pre-law ungraded mined lands (Limstrom 1960), and we have similar unpublished results from an equal number of pre-law plots, major differences were found in the Freeman study. Moisture availability, often a function of slope, is greatly affected by grading. Runoff from compacted soils would increase moisture availability and tree performance on lower slopes. Herbaceous cover was also relatively well developed on the lower slopes.

Pasture-type ground cover has been shown markedly to limit tree survival and growth (Philo et al. 1983, Ashby et al. 1988, von Althen 1989). Ground cover was an experimental variable in the study at CONSOL and the factor having the greatest effect on tree performance. Better herbicide practices for growing trees in reclamation are likely needed. More work is also needed on suitable ground cover species in which to plant trees, with tall fescue clearly a species to avoid (Vogel et al.1984).

Table 6. Survival and average height of ash, black walnut seedlings, and black walnut seed planted in 1947 at Freeman and at Sahara. Typical pH values are listed.

| Mine Site and Treatment | Survival (%) | | | Height (cm) | | |
|---|---|---|---|---|---|---|
| | Ash | Black Walnut | Black Walnut Seed | Ash | Black Walnut | Black Walnut Seed |
| Freeman, ungraded banks | | 3-year data[1] | | | 8-year data[2] | |
| Black locust cover, pH > 7.0 | 95 | 84 | 65 | 189 | 488 | 469 |
| Shortleaf pine cover, pH > 7.0 | 97 | 97 | 81 | 125 | 119 | 125 |
| Sahara | | 10-year data[3] | | | 10-year data[3] | |
| Ungraded, pH < 4.0 | 77 | 27 | 32 | 305 | 183 | 122 |
| Partially leveled, pH > 6.4 | 83 | 44 | 71 | 396 | 305 | 244 |

[1]Deitschman 1950. Three-year survival in the 1981 Freeman planting on graded spoil was 92% for ash, 25 % for black walnut seedlings, and 3% for black walnut seed.
[2]Boyce and Neebe 1959.
[3]Deitschman 1956.

The non-significant effects of fertilizer on tree growth at CONSOL likely reflect both an absence of deficiencies in the minesoils for those elements and an adverse effect on tree growth of more vigorous growth of the ground cover when fertilized.

Several factors affected the tree performance at Sahara. Although survival was comparable to that at CONSOL, growth was less. The relatively uniform fescue sward on a highly compacted rooting medium (Josiah 1986) was probably a major factor. Another major factor for the black walnut was deer damage, especially buck rub on rows of trees within the fescue meadow. Tops were broken off and trees were delimbed. This damage was so extensive that the measurements of walnut height in early fall 1988 were repeated in early spring 1989. Deer damage was evident on 76% of the saplings. The walnut, ash and oak had some browse damage. Although plantings in another study with alternating walnut and autumn olive of good size were browsed, they had only moderate buck rub damage.

A major factor adversely affecting tree growth, especially red oak, on the post-SMCRA plantings at Freeman was very extensive vole damage in 1984. The trees survived mainly as sprouts and the growth of trees relative to the herbaceous cover was set back and has not yet recovered. Some deer browsing on these rows of trees in the grassy meadow was observed.

## Conclusions

Although compaction was not an experimental variable in these studies, the relatively poor tree survival and growth as compared to pre-law plantings or to those on ripped ground or ungraded sites in other studies indicate that grading limited tree growth. Grading brought about significant effects of slope position, in contrast to pre-law lands. Adverse effects of pasture-type ground cover were clearly evident, both directly as shown by herbicide effects and indirectly by enhancing animal populations. Further attention needs to be given to the extensive damage caused by animals to trees in reclamation. Well-spaced trees on grassy meadows serve as deer parks. The meadows are also ideal habitats for voles (field mice). Populations of deer have increased greatly in recent years and thus the damage they cause has increased. Unless

needed control measures are instituted, survival and growth of walnut or other trees favored by these animals will be limited.

Failure to realize the need for extensive and continuing weed control may hinder reclamation with trees. The remarkable increases in black walnut growth with annual simazine sprays compared to mowing or limited herbicide application (von Althen 1989) need to be evaluated for mine lands.

Problems attributed to compaction may actually be caused by inadequate ground cover control. Some alternatives of less damaging ground covers are well illustrated by the CONSOL results. Still other species, such as red fescue (Festuca rubra L.), have promise (Ashby et al. 1989).

The role of fertilizers in tree plantings continues to be ambiguous. If fertilizers increase ground cover growth, they may do more harm than good. Fresh minesoils have good nutrient reserves and using an overburden mixture in place of the original topsoil could reduce a need for fertilizer. Allowing rock fragments on the soil surface would decrease erosion and reduce the need for ground cover plantings under present regulations (Ashby et al. 1984). Coarse textured materials also withstand compaction better than clay loams or silty clay loams found in southern Illinois.

Properly handled, mined lands are attractive sites to achieve alternative uses for trees. These uses include plantings for renewable energy (biomass) production and to help solve world $CO_2$ and air pollution problems. These promising uses of trees should be combined with new opportunities in reclamation suggested in part by this paper.

## Acknowledgments

The author expresses appreciation for use of the study sites and support by Consolidation Coal Company, Freeman United Coal Mining Company, and Sahara Coal Company, Inc. Clay A. Kolar and Gary R. Philo were instrumental in planting the Sahara study, and Michael R. Norland the Freeman study. Tree seedlings were supplied by the Illinois Department of Conservation. Height measurements were carried out by Ray Smith and Mike Woods. Jim McCarthy contributed to the study.

## Literature Cited

Ashby, W. C., K. P. Hannigan and D. A. Kost. 1989. Coal mine reclamation with grasses and legumes in southern Illinois. J. Soil & Water Cons. 44(1):79-83.

Ashby, W. C., M. R. Norland and D. A. Kost. 1988. Establishment of trees in herbaceous cover on graded Lenzburg minesoil. USDI Bur. Mines Info.Circ. 9184:48-53.

Ashby, W. C., W. G. Vogel, C. A. Kolar, and G. R. Philo. 1984. Productivity of stony soils on strip mines. p. 31-44 IN Erosion and Productivity of Soils Containing Rock Fragments. Soil Sci. Soc. Amer., Madison, WI.

Boyce, S. G. and D. J. Neebe. 1959. Trees for planting on strip-mined lands in Illinois. Tech. Paper 164. Central States For. Exp. Sta.,Columbus, OH. 33 p.

Deitschman, G. H. 1950. Comparative survival and growth of trees planted under three types of overhead cover on strip-mined land in southern Illinois. Sta. Note 63. Central States For. Exp. Sta., Columbus, OH. 2 p.

Deitschman, G. H. 1956. Growth of underplanted hardwoods in black locust and shortleaf pine plantations. Sta. Note 94. Central States For. Exp. Sta., Columbus, OH. 2 p.

Josiah, S. J. 1986. The effects of minesoil construction techniques and ripping on the long term survival and growth of black walnut. p. 183-193 IN Proc. Amer. Soc. Surface Mining and Reclam. National Meeting, Jackson, MS.

Kolar, C. A., W. C. Ashby and G. R. Philo. 1981. Differential performance of trees planted on reclaimed surface-mined land. pp. 271-278 IN 1981 Symposium on Surface Mining Hydrology, Sedimentology and Reclamation. Univ. Kentucky, Lexington, KY.

Limstrom, G. A. 1960. Forestation of strip-mined land in the central states. USDA For. Serv. Agric. Hdbk. 166. 74 p.

Miles, C. C. 1989. Unpublished soil descriptions.

Philo, G. R., C. A. Kolar and W. C. Ashby. 1982. Effects of ripping on minesoil compaction and black walnut establishment. p. 551-557 IN 1982 Symposium on Surface Mining, Hydrology, Sedimentology and Reclamation. Univ. Kentucky, Lexington, KY.

Philo, G. R., J. A. Spaniol, C. A. Kolar and W. C. Ashby. 1983. Weed control for better black walnut on strip mines. Tree Planters' Notes 34:13-15.

Raisanen, D. L. 1982. Survival of selected tree species on sites reclaimed to various reclamation standards. p. 93-102 IN Proceedings 1982 Better Reclamation with Trees Conference. Southern Illinois Univ., Carbondale, IL.

Vance, S. L., I. J. Jansen, and C. L. Hooks. 1987. A comparison of soil construction methods used after surface mining for coal. p. 149-152 IN Proceedings 1987 National Symposium Mining, Hydrology, Sedimentology, and Reclamation. Univ. Kentucky, Lexington, KY.

Vogel, W. G., T. W. Richards, and D. H. Graves. 1984. Survival of northern red oak and white oak seedlings planted in tall fescue and black locust-crownvetch covers. p. 33-41 IN Proceedings 1984 Better Reclamation with Trees Conference. Owensboro, Kentucky.

von Althen, F. W. 1989. Effects of weed control and irrigation on 7- and 8-year growth of planted black walnut. p. 103-113 IN Proceedings Fourth Black Walnut Symposium. Walnut Council, Indianapolis, IN.

# SUCCESSFUL OAK ESTABLISHMENT ON A RECLAIMED SURFACE MINE[1]

Walter H. Davidson, Albert W. Freeland, and Bradley Elison[2]

Abstract. We evaluated an 8-year-old northern red oak (Quercus rubra L.) plantation on a reclaimed anthracite surface mine in Pennsylvania. Survival exceeded 70 percent and some of the saplings were growing 2 feet or more each year. The tallest trees were 14 feet and the plantation averaged 8 feet. Absence of a heavy herbaceous cover, little or no soil compaction, good planting technique, and sufficient moisture all contributed to getting the seedlings off to a good start. The success of this plantation shows that red oak can survive and grow well on reclaimed minesoils.

Additional Key Words: Northern red oak, Quercus rubra L., Pennsylvania, anthracite, reforestation.

## Introduction

Establishing hardwood trees on reclaimed surface mines is a revegetation technique that coal producing states of the Eastern United States would like to practice. However, the combination of competition from herbaceous species and soil compaction makes successful establishment of oaks (and other woody species) on reclaimed minesoils more of a rarity than a commonplace occurrence. Vogel and Gray (1987) reported that compacted minesoils prevented root penetration of tree seedlings, retarded infiltration of precipitation, and increased the potential for rapid runoff and erosion.

For successful tree establishment on reclaimed surface mines, competing vegetation must be controlled to allow seedlings sufficient time to become established (Byrnes et al. 1984; Frame and Hicks 1984). Ashby et al. (1982) and Davidson et al. (1984) also concluded that soil compaction and competition from herbaceous vegetation are deterrents to tree survival on reclaimed surface mines.

Several hardwood species will grow at a satisfactory rate after they become established, but successfully planted oak often have slow early growth (Kolar et al. 1981). These factors have discouraged planting hardwoods, especially the oaks, on reclaimed mine sites.

## The Plantation

We recently found an exception to both of these "truisms" concerning compaction and competition. A surface-mined area in the Southern Anthracite Coal Field of Pennsylvania was successfully planted to northern red oak and some saplings are growing at a rate in excess of 2 feet a year. The overall results are better than one would expect in a forest planting (personal correspondence, H.C. Smith, USDA Forest Service, Parsons, WV, 1989).

The planting report describes the area as having 3 acres affected with 60 feet of shale overburden removed to recover the coal. No top soil was reserved. The initial herbaceous seeding in May 1980 consisted of birdsfoot trefoil (Lotus corniculatus) at 8 pounds/acre, tall fescue (Festuca arundinacea) at 30 pounds/acre, and winter rye (Lolium multiflorum) at 56 pounds/acre.

Amendments included 500 pounds/acre of 10-20-20 fertilizer and 2 tons of agricultural lime/acre. The seeding failed due to droughty conditions during the growing season. Seeding and amendments were repeated in the fall of 1980. Therefore, amendments total 1,000 pounds of fertilizer and 4 tons of lime/acre. In April 1981, 500 red oak (Quercus rubra), 500 white pine (Pinus strobus), and 500 autumn olive (Elaeagnus umbellate) were hand-planted on the site.

By the end of the 1981 growing season, ground cover was estimated at 80 percent and tree and shrub survival estimated at nearly 100 percent. Ground

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]Walter H. Davidson is Research Forester, Northeastern Forest Experiment Station, USDA Forest Service, Princeton, WV 24740; Albert W. Freeland is Forester, Northeastern Area, USDA Forest Service, Berea, KY 40403; and Bradley Elison is Surface Mine Conservation Inspector, PA Department of Environmental Resources, Pottsville, PA 17901.

cover diminished somewhat over the next 3 years and composition changed with the invasion of pioneer species such as dewberry (Rubus sp.), blackberry (Rubus sp.), quaking aspen (Populus tremuloides), and fire cherry (Prunus pensylvanica). During this time, white pine survival declined to about 80 percent. Red oak and autumn olive had virtually no natural mortality. However, all species suffered some mortality from damage by off-road recreational vehicles.

## Evaluation

An evaluation of the oak portion of the planting during the winter of 1988-89 showed good survival with several individuals exhibiting excellent growth. Ground cover was not estimated because evaluations were made during the dormant season.

Four randomly spaced soil samples were taken, two on the bench and two on the slope. They showed a range of physical characteristics from sandy loam to sandy clay loam. Chemical analysis showed an average pH of 5.05 (Table 1). Exchangeable and extractable minerals are shown in Table 2. The average cation exchange capacity was calculated to be 3.82 meq/100 g.

We attempted to measure compaction with a probe type compaction tester, but the soil was too stony to obtain accurate measurements. However, we concluded that the site was not extremely compacted since this was a small operation and heavy vehicle traffic would have been minimal. In addition, the stoniness and sandy clay loam texture of the soil are not conducive to extreme compaction (Table 3).

Table 1. Chemical analysis

| Sample No. | pH | CON M.Mho | Total acid meq/100 g | EXCH ALUM --- ppm --- | Total $PO_4$-BR |
|---|---|---|---|---|---|
| 5207 | 4.9 | .05 | 3.92 | 3.20 | 2.38 |
| 5208 | 4.9 | .03 | 3.28 | 2.71 | 2.52 |
| 5209 | 5.2 | .05 | 2.16 | 1.64 | 5.15 |
| 5210 | 5.2 | .05 | 2.44 | 1.84 | 7.02 |
| Mean | 5.05 | .045 | 2.95 | 2.34 | 4.27 |

We made a 100-percent tally of the oaks, measuring diameter at breast height (DBH) on all trees 4.5 feet and taller. Heights were estimated on several trees to obtain maximum heights of the tallest trees and a stand average. To determine any differences between trees planted on the bench to those planted on the outslope, we made a separate tally for each area.

## Results

There were 140 trees on the bench and 216 on the slope, or 71 percent of the trees planted. We were not able to determine percent survival on outslope vs bench, but it appeared to be about equal. Results of the tally showed that 8 years after planting the tallest trees were 14 feet (4.27 m), with many trees having annual height increments of 2 to 3 feet (60 to 90 cm). Figure 1 shows one of the taller trees. Our estimated average height of the entire planting was 8 feet (2.44 m). The diameter measurements showed that 29 percent (41 trees) on the bench had no diameter; i.e., less than 4.5 feet tall vs 19 percent (40 trees) on the outslope with no diameter. Diameters of the bench trees taller than 4.5 feet averaged 0.7 inch (1.8 cm) while outslope trees averaged 0.6 inch (1.6 cm). Since a greater percentage (81 vs 71) of the trees on the outslope had reached 4.5 feet or taller, it appears they are growing slightly faster than the trees on the bench.

## Conclusion

Our evaluation shows that red oak can be established successfully on reclaimed surface mines if they are planted properly and if compaction and competition are not severe. The oaks in this planting have survived well and are growing at a rate equal to or better than plantations on undisturbed sites.

Table 2. Exchangeable and extractable metals

Exchangeable metals (ammonium chloride extract)

| Sample | meq/100 g | | | | ppm | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| No. | Ca | Mg | K | Na | B | Si | Zn | P | Fe | Cu | Mn | Co | Al | Ni | Ti | Cr | Pb |
| 5207 | 1.68 | .82 | .20 | .12 | .35 | 2.60 | 1.80 | 1.62 | 1.62 | .65 | 18.14 | 1.54 | 226.00 | 1.66 | .17 | .39 | 4.22 |
| 5208 | 1.02 | .54 | .15 | .13 | .30 | 3.70 | .87 | 1.48 | 2.83 | .59 | 1.82 | .87 | 203.15 | .71 | .20 | .39 | 3.67 |
| 5209 | 2.73 | 1.44 | .15 | .13 | .25 | 3.31 | .79 | 2.14 | 2.10 | 1.16 | 8.76 | 1.32 | 125.44 | 1.12 | .21 | .38 | 3.45 |
| 5210 | 2.16 | 1.13 | .29 | .13 | .28 | 3.78 | .49 | 2.79 | 1.98 | .54 | 8.30 | .92 | 130.01 | .87 | .21 | .35 | 3.26 |
| Mean | 1.90 | .98 | .20 | .127 | .30 | 3.35 | .99 | 2.06 | 2.13 | .76 | 9.26 | 1.16 | 171.15 | 1.09 | .20 | .38 | 3.65 |

Extractable metals (weak acid extract)

| Sample | meq/100 g | | | | ppm | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| No. | Ca | Mg | K | Na | B | Si | Zn | P | Fe | Cu | Mn | Co | Al | Ni | Ti | Cr | Pb |
| 5207 | 1.54 | .77 | .16 | .02 | .58 | 4.58 | 2.92 | 3.46 | 51.48 | 5.67 | 38.94 | 3.02 | 802.59 | 2.64 | .31 | .30 | 8.71 |
| 5208 | .97 | .51 | .12 | .02 | .36 | 9.09 | .89 | 1.91 | 24.29 | 1.79 | 4.50 | .70 | 440.78 | .49 | .18 | .15 | 5.28 |
| 5209 | 2.99 | 1.59 | .12 | .02 | .49 | 6.62 | 1.59 | 4.77 | 51.06 | 6.54 | 25.95 | 2.60 | 616.79 | 1.56 | .22 | .31 | 8.53 |
| 5210 | 1.95 | 1.01 | .24 | .02 | .41 | 9.44 | .72 | 3.64 | 23.16 | 1.68 | 32.29 | 1.99 | 421.44 | .75 | .23 | .22 | 6.77 |
| Mean | 1.86 | .97 | .16 | .02 | .46 | 7.43 | 1.53 | 3.45 | 37.49 | 3.92 | 25.42 | 2.08 | 570.40 | 1.21 | .235 | .245 | 7.32 |

Table 3. Soil texture

| Sample No. | % Sand | % Silt | % Clay | Texture |
|---|---|---|---|---|
| 5207 | 63.9 | 20.0 | 16.1 | Sandy loam |
| 5208 | 49.9 | 28.0 | 22.1 | Sandy clay loam |
| 5209 | 63.9 | 18.0 | 18.1 | Sandy loam |
| 5210 | 53.9 | 24.0 | 22.1 | Sandy clay loam |
| Mean | 57.9 | 22.5 | 19.6 | Sandy clay loam |

Literature Cited


Ashby, W. C., C. A. Kolar, M. L. Guerke, et al. 1978. Our reclamation future, the missing bet on trees. IIEQ Document No. 78/04, Southern Illinois University, Carbondale. 99 p.

Byrnes, W. R., P. E. Pope, W. R. Chaney, B. H. Bussler, and C. R. Anderson. 1985. Ground cover control essential for establishment of hardwood seedlings on reclaimed mineland. p. 22-23. In Better reclamation with trees conference: Proceedings of the Fourth Annual Conference; Owensboro, KY, June 7-8, 1984. (Sponsored by Amax Coal Company and Purdue University.)

Davidson, Walter H., Russell J. Hutnik, and Delbert E. Parr. 1984. Reforestation of mined land in the northeastern and north central U.S. Northern Journal of Applied Forestry 1(1): 7-11.

Frame, R. C. and R. R. Hicks. 1987. Establishment of woody plants on reclaimed surface mines. West Virginia Forestry Notes 13: 25-28.

Kolar, C. A., W. C. Ashby, and G. R. Philo. 1981. Differential performance of trees planted on reclaimed surface mined land. p. 271-278. In Proceedings, 1981 National Symposium on Surface Mining Hydrology, Sedimentology, and Reclamation; University of Kentucky; Lexington. December 7-11, 1981.

Vogel, Willis G. and B. Gray. 1987. Will trees survive on topsoiled surface mines? p. 302-305.

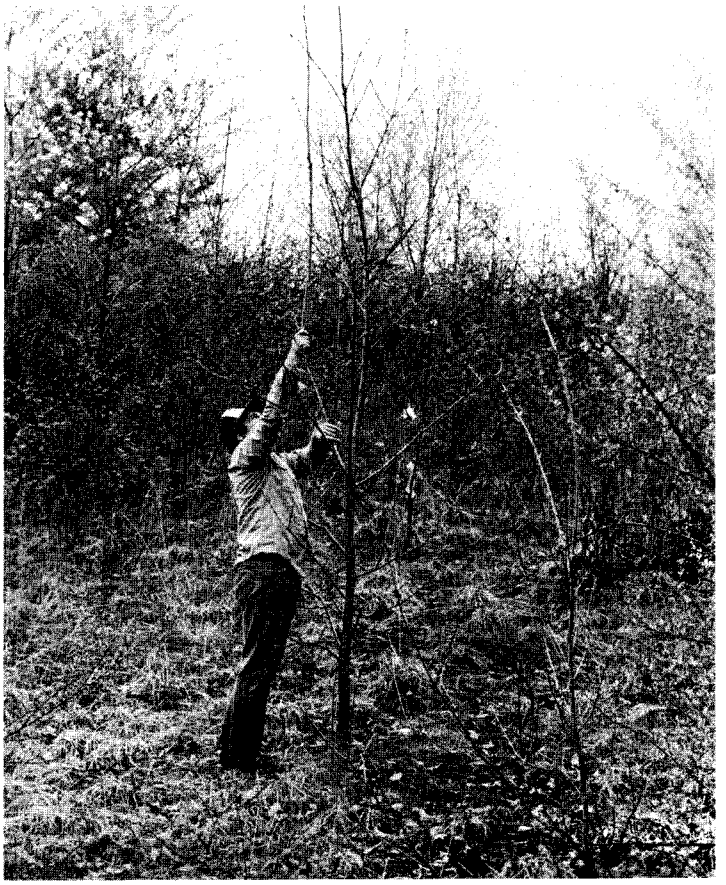



Figure 1. An 8-year-old northern red oak, 14 feet tall.


In Proceedings, 1987 National Symposium on Mining Hydrology, Sedimentology, and Reclamation; University of Kentucky; Lexington. December 7-11, 1987.

Vogel, Willis G. and T. W. Richards. 1985. Survival of northern red oak and white oak seedlings planted in tall fescue and black locust-crownvetch covers. p. 33-41. In Better Reclamation with Trees Conference: Proceedings of the Fourth Annual Conference. Owensboro, Kentucky. June 7-8, 1984. (Sponsored by Amax Coal Company and Purdue University.)

This page intentionally left blank.

# RESULTS OF EASTERN WHITE PINE ESTABLISHMENT STUDY IN SOUTHERN WEST VIRGINIA AFTER FOUR YEARS[1]

by

Timothy Probert, Ronald E. Gallimore,
John L. Torbert, James A. Burger[2]

Abstract In 1986 a study was established in Wyoming County, WV to compare several methods of establishing Eastern white pine (Pinus strobus) and black locust (Robinia pseudoacacia) on reclaimed surface mined lands in order to determine whether direct seeding was a practical alternative to handplanting seedlings. Three tree establishment treatments and two ground cover treatments were factorially arranged and replicated three times with 0.1 acre plots. Results after the fourth growing season indicate that the most cost-effective way to establish a commercial white pine plantation is to plant bare rootstock 2-0 seedlings and direct seed the black locust at a rate of less than 0.5 pounds per acre. Although some white pines were established by direct seeding, the spacing was erratic and the growth of direct seeded trees lagged behind the hand planted trees by several years.

Additional Key Words: Reclamation, Reforestation, Revegetation

## Introduction

Most lands surface-mined for coal in the Appalachians were originally forested. However, in many instances surface mined lands have been reclaimed as hayland or pasture land. These post mining land uses may be useful on lands mined by mountain top removal methods if lands will indeed be used for grazing or growing and harvesting hay. However, lands with steep slopes that are mined by contour mining methods and reclaimed as hayland and pasture land are not ideally suited for grazing or harvesting hay. In most instances, these lands, along with a large percentage of reclaimed mountain tops are abandoned following bond release and revert to relatively unproductive stands of understocked and undesirable tree species. Unless the intent is to actually manage these lands following bond release, the best and most logical approach for effective reclamation is to reforest these lands in a manner that will increase the likelihood that an economically viable forest will be established.

Forest land does not require intensive management during and after bond release and a crop of trees will grow into a valuable product for the landowner, while providing soil stability and wildlife habitat. Reforestation will be more successful and less expensive in the long run if the final stages of reclamation (overburden selection, topsoiling and grading) and revegetation are modified to recognize establishment and long term growth requirements of trees.

Eastern white pine is a good tree species for planting on reclaimed mine land in the Appalachians. On good sites, white pine is unrivaled in its ability to produce sawtimber. A natural stand of white pine has about three times the merchantable volume of a natural oak stand at age 50. (Doolittle, 1958). White pine is relatively free of insect problems in the Southern Appalachians and will probably be avoided by the oncoming gypsy moth infestation. Several characteristics of white pine make it adaptable for use on reclaimed mine land. White pine has the ability to withstand limited amounts of shade compared to other trees. This results in better survival rates when planted in grass cover. White pine is less nutrient demanding than some hardwood species and can tolerate moderately acidic sites. Since white pine stands produce more merchantable timber when seedlings are planted on a wide spacing (300 trees/acre) (Balmer & Williston, 1983), fewer trees need to be established for reclamation purposes.

---

[1]/ Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, WV

2/ Timothy Probert is Forester and Ronald E. Gallimore is Manager Mining and Reclamation, Pocahontas Land Corporation, Bluefield, WV 24701. John L. Torbert is Research Associate and James A. Burger is Associate Professor, VPI and SU, Blacksburg, VA 24061

Pines can be established by hand planting seedlings or direct seeding. Planting is the best way to ensure establishment and proper spacing, and it enables the use of genetically superior trees and fertilizer tablets which can give seedlings an early boost. However, since planting must be limited to a short period of time (late winter), labor availability and weather restrictions may constrain the amount of land that can be reforested by planting each year. Hydroseeding may be a good technique for reforesting large tracts of land quickly if tree seed can be sown together with a compatible ground cover species in the fall or spring.

A major problem associated with attempting to establish trees by direct seeding on minesoils is competition from the herbacous ground cover. Grasses and legumes commonly established to control erosion have historically made tree establishment difficult. Competition effects become especially severe when trying to establish trees from seed. A reforestation ground cover mixture of short grass and legumes was recommended by Vogel (1981). A similar ground cover mixture has been used in conjunction with direct seeded trees (Torbert and Burger, 1989a,b).

This study was designed to demonstrate the productive potential of white pine on properly constructed minesoils and to compare several tree-establishment techniques. Specifically, the objectives of this study were:

1) To evaluate the establishment of white pine and black locust by hand planting vs hydroseeding.

2) To compare two ground cover mixes for use with trees: a conventional hayland/pasture land mix vs. a reforestation mix.

3) To evaluate the effects of fertilizer tablets (applied at planting) on the establishment of white pine.

## Materials and Methods

The study was established in Wyoming County, WV on a 40% return-to-contour slope. The area had been backfilled with brown sandstone overburden. Final grading was kept to the minimum requirements under the regulatory guidelines and "tracking-in" was eliminated in order to minimize minesoil compaction. At time of seeding and tree planting the soil was loose and uncompacted. Although there were numerous large rocks, trees were easily planted in the soil around the rocks.

The study consisted of three tree-establishment treatments and two ground cover treatments factorially arranged to produce six treatment combinations. Each treatment was replicated three times producing a study with 18 plots. Plot size was 72 ft. x 60 ft. (0.1 acre).

The three tree-establishment treatments were:

1. Hydroseeding in one mix ground cover, black locust, and white pine. Seeding rates were 2 lbs/acre for white pine and 0.5 lbs/acre for black locust.
2. Hydroseeding ground cover and black locust and hand planting (with dibble bar) Eastern white pine on a 12 ft. x 12 ft. spacing. These plots were split so half the trees received a fertilizer tablet.
3. Hand planting Eastern white pine on a 12 ft. x 12 ft. spacing and interplanting black locust on a 12 ft. x 24 ft. spacing. Half the Eastern white pines received a fertilizer tablet.

The two ground cover treatments included a "conventional" hay-land/pasture land seed mix and a "reforestation" seed mix which included grass and legume species considered to be more compatible with tree seedlings (Table 1).

Table 1. Seed and fertilizer rates used for ground cover treatments.

| - - - - - - - - CONVENTIONAL MIX - - - - - - - - | |
|---|---|
| Species | Rate (lbs/acre) |
| Kentucky-31 tall fescue | 33.8 |
| perennial ryegrass | 17.0 |
| annual ryegrass | 11.5 |
| orchard grass | 10.7 |
| highland bentgrass | 2.0 |
| birdsfoot trefoil | 7.3 |
| mammoth red clover | 8.0 |
| yellow sweet clover | 4.8 |
| Nitrogen | 72.0 |
| Phosphorus | 89.0 |
| Potassium | 16.7 |
| - - - - - - - - REFORESTATION MIX - - - - - - - - | |
| Species | Rate (lbs/acre) |
| foxtail millet | 5.0 |
| perennial ryegrass | 5.0 |
| redtop | 3.0 |
| birdsfoot trefoil | 5.0 |
| 'Appalow' serecia lespedeza | 10.0 |
| Nitrogen | 36.4 |
| Phosphorus | 92.8 |
| Potassium | 55.1 |

Ground cover seed and fertilizer were applied by hydroseeding in early May 1986. Since it was not logistically possible to hydroseed tree seed onto each randomly selected plot, Eastern white pine and black locust seed was spread by hand prior to the hydroseeding. To simulate the conditions to which the tree seed might be exposed to in the hydroseeder, the seed was soaked in water for 30 minutes prior to application. White pine seedlings (2-0) were obtained from the Virginia Department of Forestry and black locust seedlings (1-0) were obtained from the Kentucky Division of Forestry. Seedlings were heavily graded such that only half the trees with the greatest root caliper were used. White pines had root calipers that measured about 0.25 inches and a stem height between 8 inches and 12 inches. Black locust stem heights averaged 24 inches.

Seedling roots were hand pruned to a length of about eight inches and dipped in a soil moisturizer slurry. Hand planting with dibble bars occurred during mid April, 1986. Soil moisture levels were good for planting, and several days of cool, rainy weather followed.

In the fall of 1987, 1988 and 1989, measurements were made of all plots. Measurements included ground cover percentage, number of trees established from seed, and survival and growth of hand planted Eastern white pines. Ground cover was determined for each plot by averaging visual estimates for the four quadrants in each plot. These estimates were made independently by the same two individuals each year. The number of hydroseeded trees was determined by counting the seedlings that grew within two 75 ft. x 3 ft. transects which ran along two diagonals in each plot. Planted pines were measured for total height and ground line diameter. From these measurements, tree volume was calculated as 1/3 $(radius)^2$ x height.

Analysis of variance techniques were used to determine the effects of ground cover treatment on percent ground cover, number of pines and locusts established by seeding, and planted pine survival and growth. Analysis of variance was also used to evaluate the effects of fertilizer treatment on the survival and growth of planted pine.

## Results and Discussion

### Minesoil properties.

The minesoil used in this study consisted primarily of a mix of native topsoil and brown sandstone overburden. The soil texture was a sandy loam, composed of 67% sand, 21% silt, and 12% clay. Typical of most brown sandstones, the pH (4.8) was low compared to desirable levels for most grasses and legumes, but it was appropriate for reforestation. Soluble salt levels were only 0.3 umhos/cm. Because the minesoil had not been tracked-in, the soil was uncompacted and easily planted.

### Comparison of Ground Covers.

The reforestation ground cover treatment contained less than one-third as much seed and was fertilized with only half as much nitrogen as the hayland/pasture treatment. (Table 1). The reforestation cover did not include Kentucky-31 tall fescue. Perennial ryegrass (Lolium perenne) and redtop (Agrostis gigantea) were the grass species included in the reforestation cover. Both treatments included birdsfoot trefoil (Lotus corniculatus) which was a dominant legume in this study. The reforestation treatment also included 'Appalow' lespedeza (Lespedeza cuneath cv. 'Appalow') which is a cultivar of serecia which tends to sprawl along the ground rather than grow upright.

The conventional cover increased from 87% during the first year to 93% after four years. (Figure 1). The reforestation cover was 80% during the first year and increased to 97% after four years. There were no significant treatment effects on ground cover in any years.

There was a large variation in ground cover across the study site, mostly reflecting variable site conditions. Several conventional ground cover plots were located in an area that consisted largely of replaced topsoil; these plots had nearly 100% ground cover. At the other extreme, the poorest ground cover occurred in conventional and reforestation plots in an area where the

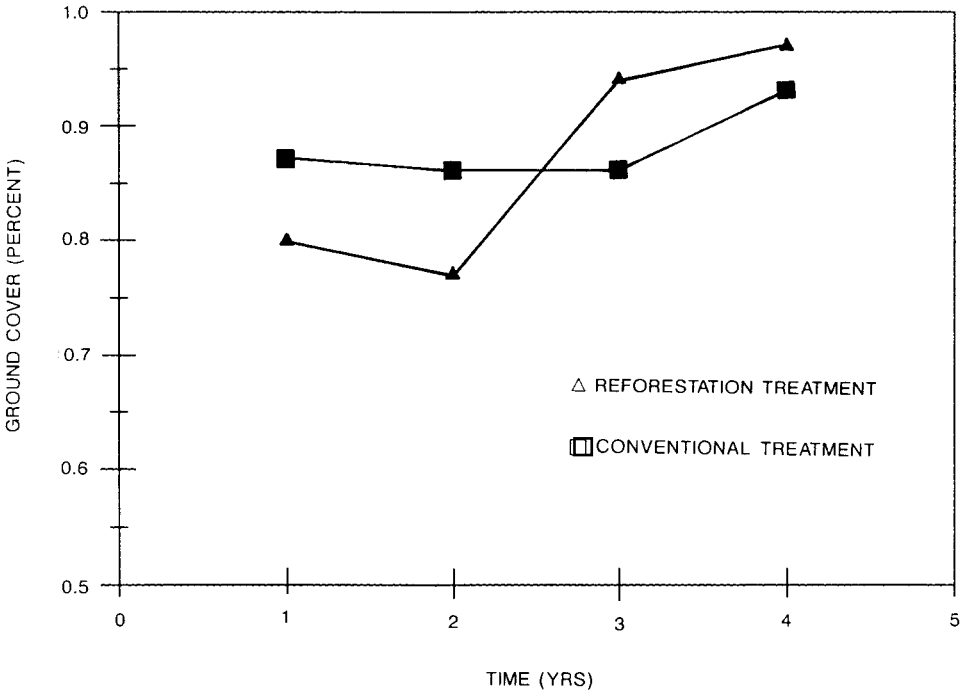



FIGURE 1. EFFECTS OF GROUND COVER TREATMENT ON PERCENT OF GROUND COVER ESTABLISHED

presence of pyritic minespoil produced a pH of 4.0. Some gullies were bare after the first growing season, but became vegetated by the end of the second season.

When the study was designed, it was anticipated that the two ground cover treatments would produce drastically different ground covers. However, throughout the four years of this study the two treatments have appeared to be very similar. Because the average mine soil pH was only 4.8, many components of the conventional ground cover treatment did not grow as vigorously as they would have on a minesoil with a higher pH. Consequently vegetation in the conventional treatment was not as high and lush as it often is on other sites.

Birdsfoot trefoil and 'Appalow' lespedeza have performed well in this study. Both species were sparse during the first year but became dense by the third year. Both treatments included birdsfoot trefoil. 'Appalow' lespedeza was seeded in the reforestation treatment but invaded the conventional treatment plots over a period of three years presumably by seed dispersal. The birdsfoot trefoil seemed to be most vigorous in the spring and fall while 'Appalow' lespedeza dominated during the summer. Thus the combination of these two species in the reforestation treatment provided a good legume cover throughout the growing season.

'Appalow' lespedeza was observed to be a very aggressive species capable of growing on a very low pH. In one of the conventional cover plots where the pH was 3.8, all of the species in the conventional mix died and by the third year the plot had been invaded by Appalow which is now the dominant component of cover.

Direct Seeding of Trees

Eastern white pines. The number of direct seeded Eastern white pines averaged 290/acre in the conventional ground cover treatment plots (range was 0 to 484 trees/acre) and 100 trees/acre in the reforestation treatment plots (0 to 194 trees/plot). Because of the large variability between plots within each ground cover treatment, the differences between numbers of trees established were not statistically significant (Table 2).

The growth of direct seeded pines was slow compared to planted trees. Most seeded pines were less than 12 inches tall (smaller than the size of the bare root nursery stock). It may take several years for direct seeded pines to become as large and vigorous as two-year-old seedlings that were hand planted. In addition to having a slower establishment growth rate, the distribution of direct seeded pines was too uneven for direct seeding to be a reliable method for the establishment of a well stocked forest.

Black locust. Black locust was successfully established by direct seeding. The use of 0.5 lb/ac of black locust seed per acre produced more than 1000 stems per acre which ranged from approximately 1 to 6 ft tall after the fourth season. This rate has produced too many locusts for use as a nurse tree with white pine. A rate of 0.1 - 0.3 lb/ac should result in a more desirable level of 100-300 locusts per acre.

Hand Planted Trees

Eastern white pine. Survival and growth of hand planted Eastern white pines has been very good. Despite two very dry summers, overall survival was 74% on the conventional ground cover treatment plots and 71% on the reforestation ground cover treatment plots. There was no significant difference between the two ground cover treatments on tree survival. Most of the mortality in this study occurred when seedlings were washed from the ground during several unusually heavy storms that occurred shortly after planting. During the second through fourth growing season, very few trees have died. Growth of these trees was excellent. Overall, the average tree height was 4 ft, and some trees were more than 8 ft tall. Research has shown that excellent white pine growth can result in the Appalachians when deep minesoils are created with acidic sandstone overburden (Torbert et al. 1988). This study further demonstrates the growth potential that can be achieved when an overburden is selected to favor tree growth and compaction associated with grading practices is minimized.

Black locust. Almost all the black locust trees planted in this study have survived, and many are more than 15 feet tall after the fourth season. These trees were planted at a rate of 100 trees/ac and are intended to provide a long-term source of

Table 2. Effect of ground cover treatment on percent ground cover and number of trees established by direct seeding.

| | GROUND COVER TREATMENT | | | |
|---|---|---|---|---|
| | Conventional | | Reforestation | |
| | Mean | Range | Mean | Range |
| Year 1 | 87 | (58-100) | 80 | (51-92) |
| Year 4 | 93 | (64-100) | 97 | (93-100) |
| Eastern white pine | 290 | (0-484) | 100 | (0-194) |
| black locusts | 1080 | (97-2,320) | 1,050 | (388-1,548) |

biologically fixed nitrogen. The effects of locust on pine needs to be monitored in the next few years, and the benefits and disadvantages of seeding locust versus planting locust needs to be evaluated.

## Effect of Ground Cover Mix and Fertilizer Tablets on Eastern White Pine Growth

Tree volume was significantly affected by ground cover treatment for all four years. Tree volume was significantly greater in the conventional treatment than the reforestation treatment (Table 3). At year 2, trees in the conventional treatment averaged 87% more volume than trees in the reforestation treatment. At year 4, the trees in the conventional treatment averaged 54% more volume than trees in the reforestation treatment (Figure 2). Although the initial results show a greater relative difference in tree volume between the conventional cover and the reforestation cover, as time goes on the relative difference has become smaller.

During the design of this study, it was hypothesized that trees in the reforestation ground cover treatment would be larger because there would be less competition compared to the conventional ground cover treatment. However, because the conventional ground cover was not as vigorous as expected, due in part, to acidic site

Table 3. Effect of ground cover treatment on average tree volume.

| Year | Conventional | Reforestation |
|---|---|---|
| | - - - - - - - - - - - - - - $cm^3$ - - - - - - - - - - - - - - | |
| 1 | 1.51 a | 1.23 b |
| 2 | 7.27 a | 3.88 b |
| 3 | 34.78 a | 20.35 b |
| 4 | 145.80 a | 94.25 b |

Values for a given year followed by different letters are significantly different, (p = 0.05).

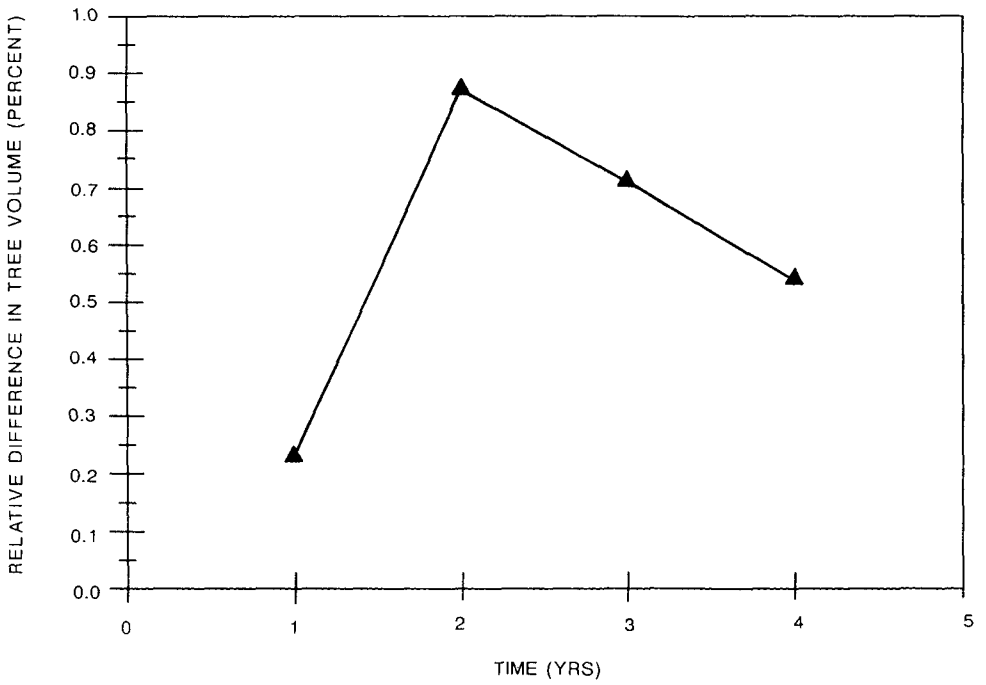



FIGURE 2. RELATIVE DIFFERENCE IN TREE VOLUME BETWEEN CONVENTIONAL GROUND COVER TREATMENT AND REFORESTATION GROUND COVER TREATMENT

conditions and a conventional treatment seed mixture that did not perform well, the competition effect never occurred. The greater tree growth in the plots with the conventional ground cover treatment over the plots with reforestation cover treatment most likely resulted from the higher rate of nitrogen fertilizer in the conventional treatment (72 lbs. vs 36 lbs. N/Ac.).

Fertilization with planting tablets resulted in a significant increase in tree volume in the conventional treatment during the first year, but the effect was not statistically significant in subsequent years. The effect of fertilization was more evident in the reforestation treatment, where there was a significant effect for the first three years (Table 4). The most probable reason for the fertilizer tablet not having a lasting effect on the pines in the conventional treatment was that during establishment enough fertilizer had been applied to the conventional cover treatment that little additional benefit was realized by the use of a fertilizer tablet.

white pine stands. If the goal of the landowner is to establish commercial forest land on the reclaimed mine site, hand planting pines and hydroseeding locust or other nitrogen fixing nurse trees/shrubs is recommended. It is also recommended that the amount of locust seed be reduced to .25 lbs-.40 lbs/acre to produce the desired number of stems per acre to benefit the Eastern white pine.

It is recommended that reforestation seed mix be used to establish ground cover as it provides a better cover over time under variable soil conditions. Initially a less dense cover allows seedings to grow and overtop the ground cover before it thickens in successive years. It also provides less competition early on to the desired crop tree and maintains a higher percentage of ground cover over time. Also, the reforestation seed mix reduces planting costs by producing a thicker cover with less seed. 'Appalow' lespedza was an excellent cover and did very well in acidic sites.

Table 4. Effect of ground cover treatment and fertilization on average tree volume.

| | Conventional | | Reforestation | |
|---|---|---|---|---|
| Year | Control | Fertilized | Control | Fertilized |
| | - - - - $cm^3$ - - - - - | | - - - - $cm^3$ - - - - - | |
| 1 | 1.24 a | 1.81 b | 1.02 a | 1.40 b |
| 2 | 7.16 a | 7.40 a | 2.68 a | 4.67 b |
| 3 | 35.09 a | 34.4 a | 15.20 a | 23.67 b |
| 4 | 144.37 a | 146.9 a | 76.30 a | 105.50 a |

Values for a given year within each treatment followed by different letters are significantly different (P = 0.1)

## Conclusions

The goal of this project was to develop a practical reclamation strategy that would establish an effective ground cover to prevent erosion and to allow mine operators to gain bond release while establishing forest lands with potential for future economic returns for Pocahontas Land Corporation.

Hand planting appears to be the most cost effective way to establish Eastern white pines on mined lands. Direct seeding on the other hand does not appear to be as reliable. The difference in cost between purchasing and planting 300 pines per acre versus purchasing and hydroseeding 2 lbs of Eastern white pine seed is negligible. The cost of direct seeding could even be higher if the expense of re-planting failed areas is included. The irregular spacing and slower establishment rate of direct seeded Eastern white pine will greatly decrease the economic potential of Eastern

This study has also shown that steep land can be successfully revegetated without "tracking in". The excellent tree growth on this uncompacted minesoil documents less final grading may be needed when the post mining land use is forests.

If trees are hand planted, higher rates of nitrogen fertilizer (50-70 lbs/ac) can be used with the reforestation mix. Rates in this ground cover mix were initially low so that the ground cover would not overtop pines germinating from seed. Since hand planted seedlings are taller and more vigorous, additional nitrogen can be added to stimulate the early growth of pines and aid initial ground cover establishment. Fertilizer tablets are not necessary for tree establishment. Any improvement in growth that might occur during the first several years will not be economically justified over the long-term.

## Literature Cited


Balmer, W. E., and H. L. Williston. 1983. Managing Eastern Eastern white pine in the Southeast. USDA For. Service Forestry Rep. R8-FR.

Doolittle, W.T. 1958. Site index comparisons for several forest species in the southern Appalachians. Proceedings Soil Science Society of America; 22 (5):450-458.

Torbert, J. L., A. R. Tuladhar, J. A. Burger and J. C. Bell. 1988. Minesoil Property Effects on Height of Ten-Year Old Eastern white pine. Journal of Environmental Quality. 17:189-192.

Torbert, J. L., and J. A. Burger. 1986a. The effects of ground cover species and nitrogen fertilizer rate on the successful establishment of hydroseeded trees. p. 83-87. In D.H. Graves (Ed.) Proceedings 1986 Symposium on Mining, Hydrology, Sedimentology, and Reclamation. Dec. 8-11, 1986 Univ. KY, Lexington, KY, UKY BU142.

Torbert, J. L. and J. A. Burger, and T. J. Nichols. 1986b. Reforestation seed mixtures for hydroseeding reclaimed mined land. Virginia Coop. Ext. Serv. Pub. 460-112. VPI-SU, Blacksburg, VA, 6pp.

Vogel, W.G. 1981. A guide for revegetating coal minesoils in the Eastern United States. USDA For. Serv. Gen. Tech. Rep. NE-68. 190 pp.

This page intentionally left blank.

# USING LAND RECLAMATION PRACTICES TO IMPROVE TREE CONDITION IN THE SUDBURY SMELTING AREA,ONTARIO, CANADA[1].

by

Peter J. Beckett[2] & John Negusanti[3]

Abstract. Extensive areas of Great Lakes-St. Lawrence mixed forest in the Sudbury area have declined over the past 100 years through a history of logging, fire, soil erosion, sulphur dioxide fumigation, acidification, particulate nickel and copper contamination and frost action. Pollution controls enacted since 1972 have not resulted in widespread recolonization and tree invasion as expected in the barren and depauperate White Birch-Red Maple (*Betula papyrifera - Acer rubrum*) communities. Birch growing in the toxic soils shows marginal leaf chlorosis during the mid summer while the maple exhibits regressive dieback. Since 1978, approximately 3000 hectares have been revegetated by the Sudbury Land Reclamation Programme. Dolomitic limestone, fertilizer and a grass-legume mixture were applied in the first year and in subsequent years a total of 1 million bare root and paper pot tree stock have been planted into the reclaimed sites. Trees introduced include Red Pine (*Pinus resinosa*), White Pine (*P. strobus*), Jack Pine (*P. banksiana*), White Spruce (*Picea glauca*), Red Oak (*Quercus rubra*), and Black Locust (*Robinia pseudoacacia*). Amounts of macro-elements in the foliage are below normal, except for nitrogen levels in Black Locust. Nickel and Al concentrations are high in tree tissue but not at toxic concentrations, whereas, Cu is average and Zn is present in low amounts. There is little evidence of chlorosis in the needles or in birch or maple leaves in the treated areas. Growth rates of the conifers are typical for trees found on acid, nutrient poor sites of northern Ontario. Trees lost due to insect or fungal disease have been minimal.

additional key words: rehabilitation, metals, conifers, hardwoods, landscape ecology.

[1] Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, April 24-26, 1990, Charleston, WV, USA.

[2] Biology Department, Laurentian University, Sudbury, Ontario, Canada P3E 2C6;

[3] Ontario Ministry of the Environment, 199 Larch Street, 11th Floor, Sudbury, Ontario, Canada P3E 5T9

## Introduction

### Background to the programme.

Sudbury, Ontario, Canada (46° 30' N 81° 00' W) is situated on the southern edge of the Canadian Precambrian Shield, in a glaciated landscape characterized by a mosaic of rock outcrops, glacial till deposits and lakes. The area supported a transitional forest of boreal and hardwood elements; the Great Lakes-St. Lawrence Forest (Rowe, 1959). There were extensive stands of Red (*Pinus resinosa* ) and White Pine (*P. strobus*) which were extensively logged from 1872 onwards, together with Yellow Birch (*Betula lutea*), Sugar Maple (*Acer saccharum*) and Red Oak (*Quercus rubra*). Regrowth supported stands of White Birch (*Betula papyrifera*) and Trembling Aspen (*Populus tremuloides*).

The mining and smelting of sulphide ores began in 1882, and smaller trees were used as mine timbers and as fuel for the open roast beds. Fires were started by sparks from the wood-burning locomotives of the Canadian Pacific Railway, and prospectors often burned the vegetation and duff to reveal the bedrock below. Sulphur dioxide from the ground-level roast yards also killed vegetation in their vicinity between 1888 and 1929, while smoke from the smelters, containing not only sulphur dioxide, but also copper, nickel and iron particulates, leached and acidified the soil, contaminating it with copper and nickel. Acidification caused aluminum from the soil minerals to be solubilized. The soil, having lost its protective vegetation cover, suffered extensive erosion, exacerbated by the intense frost-heaving and needle ice formation (Amiro and Courtin 1981; Winterhalder 1984)

The net result was the creation of 10,000 ha of completely barren land, as well as 36,000 ha of open woodland dominated by stunted and coppiced *Betula papyrifera* (White Birch), *Acer rubrum* (Red Maple) and *Quercus borealis* (Red Oak) (DeLestard 1967). Peatlands have also been affected by the industrial operations (Gignac and Beckett 1986).

In 1972, one of the three existing smelters, as well as an iron ore sintering plant, were closed. In addition, the three smokestacks at another smelter were replaced by a 381 metre stack, and emissions reduced. Following the improvement in atmospheric quality, some of the barren areas were colonized by metal-tolerant grasses such as *Deschampsia caespitosa* (Tufted Hairgrass), *Agrostis gigantea* (Redtop) and *A. scabra* (Tickle Grass) and the moss, *Pohlia nutans*.

On a few stony hillsides germination and re-establishment of scattered White Birch seedlings began approximately ten years following smelter closure. Near poor fens, *Betula pumila* var. *glandulifera* (Dwarf Birch) began, un-characteristically, to move up the slopes.

On barren sites, soil pH ranged from 2.0 to 4.5, while copper and nickel levels frequently reached 1,000 $\mu g\ g^{-1}$. Root growth was so inhibited that seedlings quickly dried out and died. Bioassay experiments showed that barren soils could be detoxified by addition of ground limestone. Field trials showed that a grass-legume cover could be achieved on the stony slopes by the manual application of ground dolomitic limestone, fertilizer and a seed mixture. The stones acted as a mulch, trapping the seeds and cutting down moisture loss. The surface application of limestone was sufficient to "promote" colonization by woody plants, even in the absence of fertilizer and seed (Winterhalder 1985).

Early rehabilitation attempts in 1969 and 1970 using trees (both bare root and container stock) on unamended barren soils had little success. (summarized by Winterhalder 1984). However, trials with bare root Jack (*Pinus banksiana*) and Red Pine in 1975 and 1976 using lime and fertilizer and in 1978 using the same species in Japanese Paper Pots (Negusanti 1978) showed excellent growth. Subsequently, as reported by Lautenbach (1985,1987, 1988) the Sudbury Land Reclamation Programme was initiated (1978) under the direction of the Vegetation Enhancement Technical Advisory Committee (VETAC) that reports to the Regional Municipality of Sudbury. In addition, both mining companies have been operating their own reclamation projects.

Philosophy governing tree planting

Before tree planting could be successful on a large scale, a herbaceous cover together with proper soil treatment was necessary. The grass-legume mixture must not be too dense so that it chokes the woody species. Nor must the herbaceous cover be sufficient to support a large population of small mammals that may damage the small trees. The grass legume mixture provides a mulch and, before the trees are fully established, acts as a nutrient reservoir. However, the nutrients must become available to the trees to avoid permanent stunting. The 10 - 30% herbaceous cover achieved in the programme appears suitable for adequate tree growth.

The presence of legumes should be of longterm benefit to the trees as a source of

nitrogen. Indeed some of the more successful treeplots contain Birdsfoot Trefoil (*Lotus corniculatus*) as a dense understory. However, the creation of woodland with an understory of agronomic species may seem unusual. Long-term monitoring has shown that the agronomic species decrease in importance over time and woody species through colonization increase in importance (see Winterhalder 1988).

The establishment of an open grass cover initiated colonization by seedlings of White Birch, *Salix* spp. (Willows) Trembling Aspen and in some localities, Dwarf Birch. Since natural colonization of woody species is occuring so rapidly, one might question the cost and expense of artificially introducing herbaceous and tree species on these sites. In time, one might expect a slow colonization by climax trees including the pines, spruces and oaks. However, the monocultural problems and the visual appearance of uniform stands of these colonizers is not very appealing.

By directly introducing trees into the grassed areas, total diversity could be quickly increased and a large number of niches can be available for other components of the ecosystem. In addition, one of the aims of the programme is the partial restoration of the mixed forest that was characteristic of Sudbury one hundred years ago. Once established, the introduced trees should act as a local source of seeds.

Mature trees in the landscape are much more intruding than a herbaceous cover. Thus it has been the intention of the Programme to create semi-natural, attractive, diverse and stable bio-ecosystems. Hence, the use of extensive "plantation" planting and very high stocking rates is opposed. Before planting, evaluating the site aids in the determination of a harmonious grouping of trees with the landscape, helps create openings in the arrangement, maintains sight lines, and makes the landscape appealing upon maturation of the trees. This aim is helped by the rocky terrain with small and large pockets of soil that make it difficult to completely cover the terrain with trees. Pines and spruces have been extensively used to provide year-round greenery (Beckett 1988)

Since one of the aims is to recreate, at least partially, the appearance of the original forest cover, the use of exotics is minimized. Black Locust has, however, been used because of its ability to fix nitrogen and to withstand windy locations. It is often mixed with pines or Red Oak to act as a nurse tree. For autumn colour, European

and Japanese Larch has been used in well-drained localities. There is evidence that the native Tamarack (*Larix larcina*) will grow satisfactory in these locations and creating trials with this species are in progress.

## Materials and Methods

### The Regional Land Reclamation Program - the Process

The major activities of the programme are summarized in Table 1.

a) Establishment of Grass Cover. Depending on the soil pH and colloid content, between 4.5 and 11 t $ha^{-1}$ of ground limestone were applied. The limestone was allowed to react with the soil for a few weeks before the addition of a N-P-K fertilizer having a high P:N ratio (usually 4:1). Sudbury soils are more deficient in phosphorus than nitrogen and phosphorus can decrease the amount of toxic metals, (especially Al) available to roots. Fertilizer in 50 kg bags was hand-spread at a rate of approximately 390 kg $ha^{-1}$ The fertilizer application usually occurred late in the summer to coincide with the seeding operation.

Seeding began in mid-August and continued through September to coincide with the autumn rains. A blended mixture of five agronomic grasses and two legume species was spread using Cyclone-seeders at a rate of 30 - 45 kg $ha^{-1}$

During the period 1978-89 approximately 3,000 ha have been treated, with the ten-year old areas showing a good cover of volunteer birchs and poplars, often up to 2.5 m in height and a diverse assemblage of herbs.

b) Tree Planting Programme. In spite of occasional relict Jack Pines (*Pinus banksiana*) and Red Pines, there is little evidence so far of volunteer pine establishment. For this reason, a tree-planting programme has been in effect for almost as long as the grassing programme. Both bare-root nursery stock and paperpot stock have been used, and over one million tree seedlings have been planted. Apart from some early test plots, seedlings are not planted in rows, but in groups that conform with topography and appear relatively natural. It is hoped that these trees will form a seed-source for later spread, as has began to occur in the ten year old Jack Pine stands.

The Tree Planting Subcommitee of VETAC is responsible for the selection of planting sites and tree allocation. Although a proposed tree planting plan has been drawn up (Regional Municipality of Sudbury, 1986) each site is visually inspected by the Sub-

## TABLE 1. LAND RECLAMATION PROGRAMME ACHIEVEMENTS 1978-89

| YEAR | AMOUNT LIMED (hectares) | AMOUNT FERTILIZED (hectares) | AMOUNT SEEDED (hectares) | AMOUNT SITE IMPROVED (hectares) | NUMBER OF TREES PLANTED | OTHER ACHIEVEMENTS |
|---|---|---|---|---|---|---|
| 1978 | 114.8 | 114.8 | 114.8 | 206.3 | | 30,000 pH and nutrient samples. 365 kg of native seed collected. 11,000 trees, shrubs and plants transplanted. 122 composting test plots. |
| 1979 | 478.6 | 466.6 | 420.2 | 295.9 | 4,250 | 420 hectares sampled for pH. 425 kilograms of native seed collected. 20,000 trees, shrubs and plants transplanted. Monitoring and assessment begun. |
| 1980 | 331.0 | 299.3 | 299.3 | 258.7 | 1,300 | Land reclamation data assembled and computer coded. 2,000 pH samples taken. 5 Year Land Reclamation Plan developed. |
| 1981 | 208.0 | 173.4 | 173.4 | 9.8 | 4,600 | 5 Year Plan updated. Monitoring and assessment records processed. 29 research plots established. |
| 1982 | 362.4 | 342.4 | 305.2 | 199.2 | - | Dismantled 2.4 km of abandoned trestle and improved tailings wildlife area. |
| 1983 | 1,084.0 | 934.6 | 935.4 | - | 228,080 | Established 10 wildflower experimental test plots. |
| 1984 | 57.7 | 188.4 | 215.9 | 7.5 | 149,350 | Timber cruised 3,213 hectares. Transplanted 400 trees. Updated all mapping records. Compiled second 5 year grassing plan. |
| 1985 | 112.0 | 106.0 | 106.0 | - | 154,600 | Native shrub seed sources identified. Land Reclamation Summary Report 1978-84 published. |
| 1986 | 24.0 | 30.0 | 30.0 | - | 80,300 | 5 Year Tree Planting Plan developed. Monitoring survey of trees planted 1978-1985 completed. Forest management report prepared. |
| 1987 | 59.5 | 64.7 | 64.7 | - | 264,880 | Seeds of 15 species of trees or shrubs collected for diversity and future outplanting. Seed collection report prepared. |
| 1988 | 34.8 | 34.8 | 34.8 | - | 118,885 | Planted 8,300 shrubs. Monitored and assessed past work. Wildlife habitat improvement report prepared. |
| 1989 | 7.5 | 12.2 | 12.2 | - | 65,740 | |
| Totals | 2,874.3 | 2,767.2 | 2711.9 | 977.4 | 1,071,985 | |

Table 2. Type and number of tree seedling planted (l979-89)

| Species | | Number | Symbol |
|---|---|---|---|
| conifers | | | |
| Jack Pine | *Pinus banksiana* | 345 000 | Pj |
| Red Pine | *Pinus resinosa* | 350 000 | Pr |
| White Pine | *Pinus strobus* | 100 000 | Pw |
| White Spruce | *Picea glauca* | 25 000 | Sw |
| Black Spruce | *Picea mariana* | 10 000 | Sb |
| Norway Spruce* | *Picea abies* | 5 000 | Sn |
| White Cedar | *Thuja occidentalis* | 15 000 | Ce |
| Japanese Larch* | *Larix leptolepis* | 2 000 | Lj |
| European Larch* | *Larix decidua* | 3 000 | Le |
| Tamarack | *Larix laricina* | 7 000 | Ta |
| hardwoods | | | |
| Black Locust* | *Robinia pseudoacacia* | 90 000 | Lb |
| Red Oak | *Quercus rubra* | 78 000 | Or |
| Silver Maple* | *Acer saccharinum* | 30 000 | Ms |
| Sugar Maple | *Acer saccharum* | 4 600 | Mh |
| White Ash | *Fraxinus americana* | 6 000 | Aw |

* Not native to the Sudbury area

committee and the numbers of each species deliminated on 1: 2000 maps for use by the planting crews.

In total, fifteen species have been used: ten conifer species and seven hardwood species (see Table 2). The majority of the stock has been bareroot, obtained through the Ontario Ministry of Natural Resources' tree nurseries. Trees are stored in a reefer van at 4 $^0$C until planted. All planting has occurred between early May and the beginning of June. Unfortunately hot and dry weather has been experienced in some years during or immediately after the planting period. In 1987, approximately 70 000 paper pot Red, White and Jack Pine were planted in addition to the bare root stock.

c) Evaluation of Tree Planting Programme. Monitoring of trees at selected sites is performed to evaluate survival, growth, performance and chemical composition. The results are used to refine the tree planting programme. Each monitoring site contains approximately 100 trees and each species planted is replicated at several locations across the Region of Sudbury. During the year of planting, trees are assessed for survival and general health only. In subsequent years trees are individually identified and survival, growth by height, diameter of trunk and general health are monitored. At a few sites tree parts are sampled for chemical analysis. Full details of monitoring sites, method and results may be found in a series of tree monitoring reports produced for the Vegetation Enhancement Technical Advisory Committee (1985 - 1989). The monitoring has been assisted through the use of students and their help is much appreciated.

## Results and Discussion

### Tree monitoring and assessment

a) Tree Survival Figure 1 reports the mean survival of the major tree species three years after planting. In general, the pines show the best survival rates. Black Locust is planted on difficult sites but showed good survival rates. The more sensitive hardwoods, maples and ash, are used in a few restricted sites with the most favourable conditions, but their survival is poor. The generally shallow and dry soils of the reclaimed lands are best suited to pines and this is reflected in their best overall survival rates. The high survival of Red and Jack Pine has led to the extensive use of these two species (see Table 2). The development of monocultures of these 2 species is not ideal and efforts are made to diversify the mixture of trees planted with or near the pines. Despite the low tolerance of White Pine to pollution stress,

Fig. 1. Percentage Survival Rate of Major Tree Species after 3 Years. (Legend for species is in Table 2.)

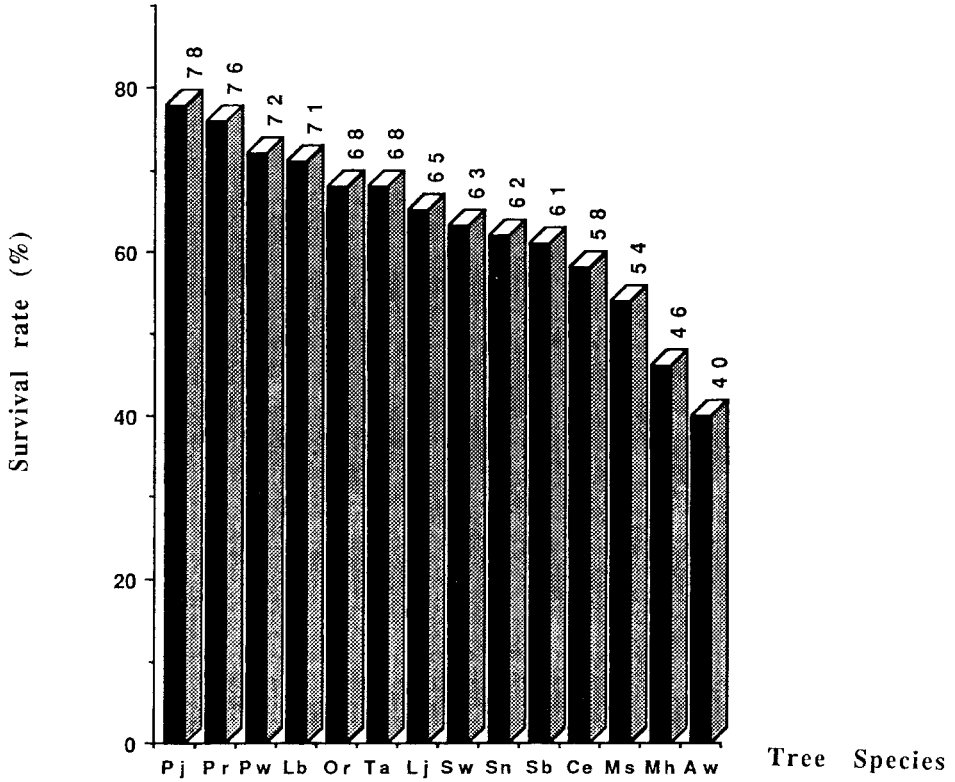
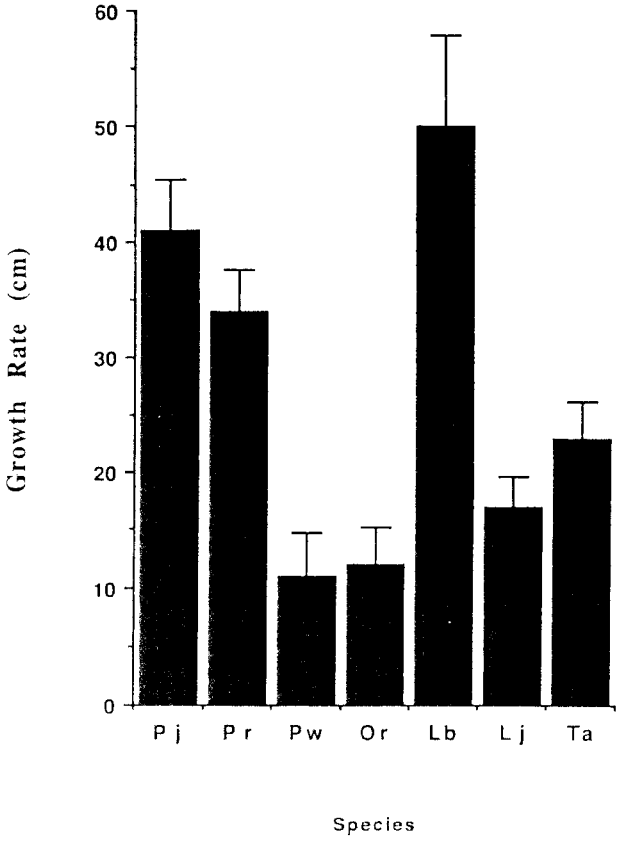



Fig. 2. Mean Annual Growth of Trees on Reclaimed Land (+ standard) error). Measurements were made 3 years after planting.



especially sulphur dioxide, the species has a respectable survival rate. White Cedar has a low rate of survival. In part this is due to its planting in wetland sites that are not always limed. On slopes where the reclamation treatment has been applied, the survival rate of cedar increases dramatically.

b. Tree Growth. The annual growth rates for a selection of the major species are reported in Figure 2. Jack Pine and Red Pine are the fastest growing conifers The growth of Black Locust in the Sudbury area is reduced each year by hard winter frosts killing the youngest extension growth

The experimental sites of Red and Jack Pine set up in 1979 have allowed growth patterns to be followed over the longer term. The cumulative growth of these species on reclaimed sites is compared to control sites (see Fig. 3 and Fig. 4). The control sites are outside Sudbury and have similar soil properties and soil acidity but not the elevated metal levels of the Sudbury area. There is no statistical difference ($p>0.05$ for slope comparison) between growth rates at the Sudbury reclaimed and control reforestation sites for either species. Jack pine has the faster growth rate and is therefore a useful tree for quickly providing cover and winter greenery. However, the visual appearance of Red Pine is more appealing. These results indicate that the trees in Sudbury are performing satisfactorily but need several years of growth before they can be visually appreciated.

c. Chemical composition. Metal content in the needles increases with age. Nickel and aluminium levels are high but do not appear toxic (Table 3), whereas, copper levels are about normal and zinc amounts are less than normal. Similar patterns have been observed in both Red and White Pine, although the metal levels are lower. Aerial deposition of metal particulates may contribute to the metal concentrations associated with the needles.

Amounts of macro-nutrients in the foliage are below normal. However, White Birch. Red Maple and Red Oak growing in the reclaimed areas have significantly higher levels of calcium and magnesium and do show the marginal chlorosis of leaves from untreated areas. Black Locust appears to be fixing nitrogen since the leaves are a dark green and contain 2-4% nitrogen; similar to nitrogen values for trees where nitrogen fixation has been demonstrated.

## Conclusions

1. The tree planting programme has stressed diversity, but uses a minimum number of exotic

Fig. 3. Cumulative Growth of Jack pine planted in 1979 compared with a non-polluted reforestation stand planted in the same year. (n = 100 at each site)

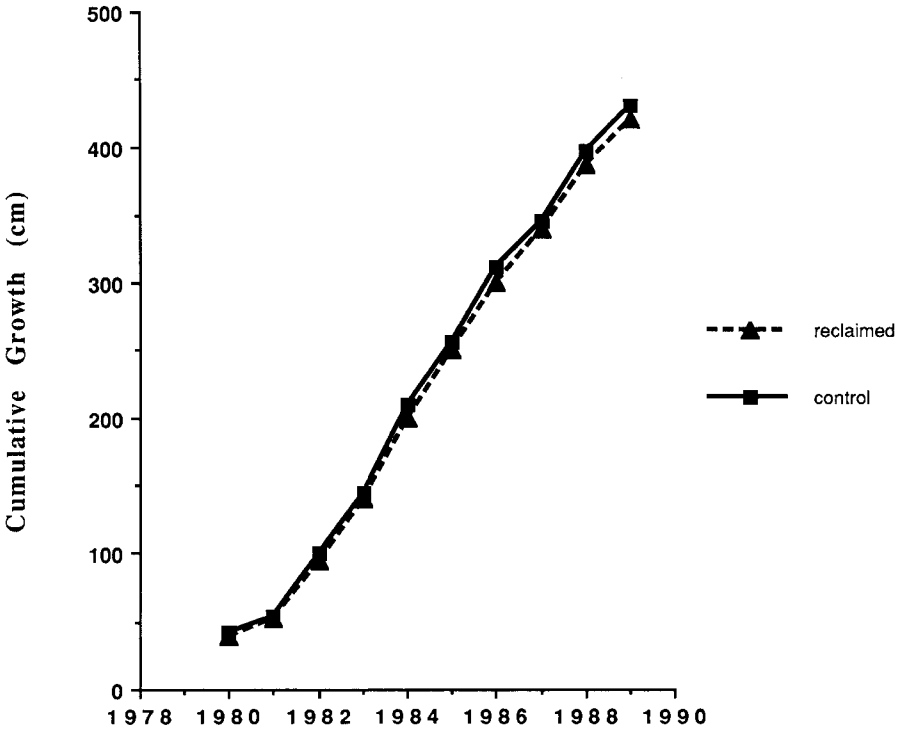



Fig. 4. Cumulative Growth of Red pine planted in 1979 compared with a non-polluted reforestation stand planted in the same year. (n = 100 at each site)

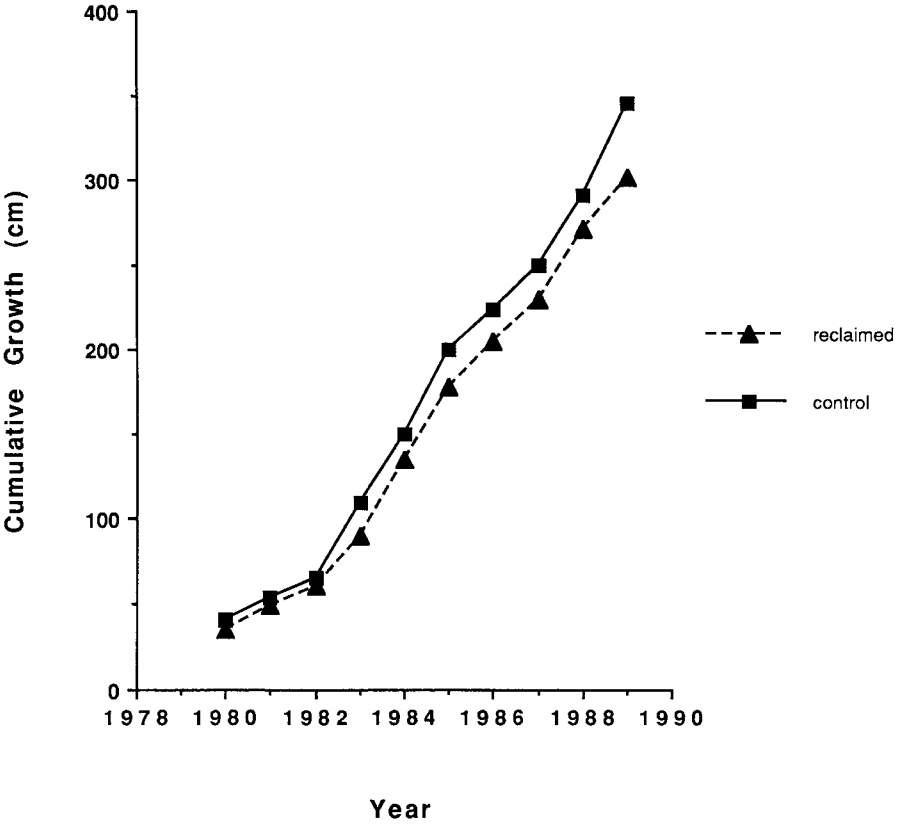


Table 3 Metal levels in Jack Pine Needles of 3 ages from a reclaimed site in Sudbury (planted l979) and a control, 50 km away. (Values are means +/- standard error based on a sample size of 20.)

| SITE | NEEDLE AGE (year) | METAL CONCENTRATION ($\mu g\ g^{-1}$) Al | Cu | Ni | Zn |
|---|---|---|---|---|---|
| Sudbury | 1 | 324 +/- 5 | 8.9 +/- 1.3 | 16.0 +/- 1.3 | 123 +/- 23 |
| Sudbury | 2 | 345 +/- 6 | 8.5 +/- 1.4 | 19.0 +/- 2.1 | 134 +/- 34 |
| Sudbury | 3 | 620 +/- 11 | 8.1 +/- 1.9 | 30.0 +/- 3.3 | 178 +/- 45 |
| Control | 1 | 234 +/- 6 | 2.0 +/- 0.9 | 3.5 +/- 0.8 | 235 +/- 34 |
| Control | 2 | 244 +/- 9 | 2.4 +/- 0.8 | 3.8 +/- 1.0 | 256 +/- 35 |
| Control | 3 | 439 +/- 7 | 2.7 +/- 1.3 | 6.3 +/- 1.3 | 310 +/- 32 |

species. However based on survival and growth information, a large number of Red and Jack Pine have been used.

2. The use of White Pine, despite its alleged sensitivity to sulphur dioxide, has allowed some diversity in the use of pines. Survival of White Pine indicates that pollution control measures in the Sudbury area have been beneficial.

3. The majority of the stock planted has been bare root. This has restricted the length of the planting season. Paper pot stock could be used with advantage to extend the planting period.

4. Other species of tree and shrubs have potential use in the Sudbury Programme. The present source of trees has been Ontario Ministry of Natural Resources' nurseries and thus only a restricted list of species is available. In addition, the plant materials may not have originated from NE Ontario. Other species of local origin could increase the diversity and be more characteristic of the original forest ecosystem of Sudbury. A programme to collect and grow material of local origin is in progress.

5. With respect to landscape creation, there has been an emphasis on making use of the natural topography. The micro-environmental pattern has imposed its mark upon the resulting vegetative cover. Grasses and other plants vary, with respect both to species and to density, in response to differences in soil depth, soil moisture and the shelter factor. The seeds of colonizing woody plants have been deposited and have germinated in a non-random fashion, presumably dependant on topographical factors such as soil moisture, aspect and turbulent wind-flow. Overall, the spatial heterogeneity of the vegetation, overlaid on the already diverse topography, has given rise to an attractive and natural-appearing landscape.

Superimposed on the topographic-vegetational pattern is one created by tree-planting. Apart from early trials, the plantation effect has been avoided, and trees are planted in groupings that conform to the topography. An extra dimension is added by the response of the trees themselves. Where even-aged tree seedlings are planted, the micro-environmental pattern is reflected in the varied stature and vigour of their growth, helping to blend them into the landscape.

6. At present the major land use of the reclaimed areas is rural passive, recreation and wildlife habitat. To futher wildlife enhancement, several species of shrubs have been

planted in and around areas with introduced trees. If the feasibility of growing shrubs from locally collected seed is economical, it is anticipated that more use of shrubs and other species useful for wildlife will be utilized.

7. The reclamation effort in Sudbury has played a role in improving the national image of Sudbury and has helped in economic diversification in the region. The less bleak landscape is contributing to the concept that Sudbury can be a "healthy place" for "healthy people".

## References


Amiro B.D. and G.M. Courtin. 1981. Patterns of Vegetation in the Vicinity of an Industrially Disturbed Ecosystem, Sudbury. Canadian Journal of Botany. 59: 1623-1639

Gignac L.D. and P.J.Beckett. 1986. The effect of Smelting operations on Peatlands near Sudbury, Ontario, Canada. Canadian Journal of Botany. 64: 1138-1147.

Beckett P.J. 1988. Tree PlantingActivities of the Sudbury Regional Land Reclamation Programme - An Evaluation. In Proc. 12th Annual Meeting, Canadian Land Reclamation Association, 32-41.

DeLestard, L.P.G 1967. A History of the Sudbury Forest District. Department of Lands and Forests, Ontario: District History Series No. 21. 90 pp.

Lautenbach, W.E. 1985. Land Reclamation Programme 1978-1984. Regional Municipality of Sudbury. 65 pp.

Lautenbach, W.E. 1987. The Greening of Sudbury. Journal of Soil and Water Conservation 42,228-231.

Lautenbach, W.E. 1988 The Sudbury Regional Land Reclamation Program. In Proc. 12th Annual Meeting, Canadian Land Reclamation Association, 22-31.

Negusanti, J.J. 1978. Revegetation of deteriorated Sudbury Sites through Tree Plant Trials. Proc. C.L.R.A. 3rd Annual Meeting, Sudbury: 14-24.

Regional Municipality of Sudbury. 1986. Proposed Tree Planting Plan. Regional Land Reclamation Program.

Rowe, J.S. 1959. Forest Regions of Canada. Bulletin 123, Canada Department of Northern Affairs and National Resources, Forestry Branch. 71 pp.

Vegetation Enhancement Technical Advisory Committee. 1985/86/87/88/89. Tree Monitoring Reports Regional Municipality of Sudbury.

Winterhalder, K. 1984. Environmental Degradation and Rehabilitation in the Sudbury Area. Laurentian University Review 16(2): 15-47.

Winterhalder, K. 1985. The Use of Manual Surface Seedling, Liming and Fertilization in the Reclamation of Acid Metal-contaminated Land in the Sudbury, Ontario Mining and Smelting Region of Canada. Proc. 1985 National Meeting, Amer. Soc. for Surface Mining and Reclamation, Denver: 196-204.

Winterhalder, K. 1988. The Sudbury Regional Land Reclamation Programme - An Ecologist's Perspective. Proc. C.L.R.A. 12th Annual Meeting, Sudbury: 81-92.

# Restoration of Forested Wetlands

This page intentionally left blank.

# DEVELOPMENT AND SUMMARY OF MiST: A CLASSIFICATION SYSTEM FOR PRE-PROJECT MITIGATION SITES AND CRITERIA FOR DETERMINING SUCCESSFUL REPLACEMENT OF FORESTED WETLANDS[1]

T.A. White, R. Lea, R.J. Haynes, W.L. Nutter, J.R. Nawrot, M.M. Brinson, and A.F. Clewell[2]

Abstract. Inconsistency in the evaluation of bottomland hardwood mitigation plans by state and federal agencies often leads to conflict. EPA Region IV and TVA requested the Hardwood Research Cooperative (HRC) of the College of Forest Resources at N.C. State University to conduct a workshop to address this issue. Subsequently, wetland experts representing industry, universities and regulatory agencies across the Southeast were assembled in a workshop to review and refine a mitigation site type classification system proposed by the HRC.

The result of the workshop is the MiST classification system. This system evaluates sites to be mitigated based upon the condition of key site factors controlling productivity. The degree of monitoring intensity required to assess project success depends upon the MiST classification of the site. The system promises to ease project evaluations and provide a conduit for communication between mitigators and commenting agencies. The MiST classification system is reviewed and examples are provided.

Additional keywords: creation, disturbance, enhancement, no-net-loss, reclamation, restoration, restoration ecology, Section 404, surface-mining.

## Introduction

The goals of the Clean Water Act (CWA) are to protect and maintain the chemical, physical, and biological integrity of the nation's waters. The Environmental Protection Agency (EPA) and Army Corps of Engineers are given regulatory authority for the discharge of dredged or fill material into waters of the United States. Section 404 of the CWA is the primary mechanism for the protection of wetland by Federal authorities. While "no net loss" of wetlands, including forested wetland ecosystems, has been identified by some groups as a top national priority (The Conservation Foundation 1988), the mechanisms by which such a goal can be achieved have not been developed. In addition to the regulatory and policy problems that a "no net loss" goal presents, there is the fundamental technical challenge of restoring or creating new wetland resources to offset inevitable or unavoidable losses that will continue to occur. This latter concern must be addressed through mitigation. Although the term "mitigation" has come to include avoidance of the wetland impact, for the purposes of this paper mitigation will refer to rectifying the wetland impact by repairing, rehabilitating, or restoring the affected environment or

[1] Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990. Paper No. 12538 of the Journal Series of the North Carolina Agricultural Research Service, Raleigh, NC.

[2] The authors and their working groups in parentheses are Restoration Ecologist (Workshop coordinator) and Director, Hardwood Research Cooperative, North Carolina State University, Raleigh, NC; Botanist, U.S. Fish and Wildlife Service, Atlanta, GA (Habitat); Professor, School of Forest Resources, University of Georgia at Athens, GA (Hydrology); Associate Scientist, Cooperative Wildlife Research Laboratory, Southern Illinois University, Carbondale, IL (Soils); Professor, Biology Department, East Carolina University, Greenville, NC (Water Quality); and President A.F. Clewell, Inc., Sarasota, FL (Vegetation), respectively.

compensating for the impact by replacing or providing substitute resources or environments (CEQ Mitigation Types, 40 CFR Part 1508.20).

Recent bottomland hardwood (BLH) workshops (Roelle et al. 1987a-c) outline the functional attributes of forested wetlands and lay the groundwork for the development of models to assess these functions (Adamus et al. 1987). Yet, regulatory agencies also need an approach for objective, standardized evaluation of forested wetland mitigation plans with respect to their ability to restore functional capacity in disturbed areas. In this document, terms regarding site functions refer to functional effectiveness (Adamus et al. 1987) unless otherwise stated.

Lack of standardization and proven methods has led to inconsistencies in evaluation of mitigation plans by state and federal agencies across EPA Region IV (Kentucky, Tennessee, North Carolina, South Carolina, Mississippi, Alabama, Georgia, and Florida). EPA and the Tennessee Valley Authority requested the N.C. State University Hardwood Research Cooperative (HRC) to conduct a workshop to address these problems. The Workshop on BLH Forest Mitigation of Disturbed Sites was held at the Fall Creek Falls State Resort Park near Pikeville, Tennessee on August 13-15, 1989. Individuals from federal and state regulatory agencies, industry, and universities participated.

HRC proposed that consistency in the evaluation of BLH mitigation plans could be reached by initially developing a framework to classify pre-construction mitigation sites[3]. Classification would be based upon the present condition of the mitigation site and the inputs likely to be required to restore the site to a pre-disturbance functional condition. A prototype system was presented to Workshop participants for their refinement.

The end result of the workshop is the Mitigation Site Type Classification System (MiST). Early in the Workshop discussions, participants decided to expand the system to include all freshwater forested wetland ecosystems commonly occurring in the Southeast and Southcentral United States. This classification system can be used to sort out the range of potential options available to monitor the success of forested wetland mitigation projects. MiST also can foster better communication regarding forested wetland mitigations throughout Region IV. This paper summarizes the draft MiST document generated from the Workshop (White et al. 1990) and provides examples of its potential utilization.

[3] In this document, a mitigation project is considered construction of an ecosystem. Terms such as "construction", "project site", etc. refer specifically to the mitigation and not to the proposed wetland fill activity per se.

## Background

Wetlands form the interface between terrestrial and aquatic environments and wetland hydrology is the driving force behind their creation (Federal Interagency Committee for Wetland Delineation 1989). Different combinations of wetland hydrologic and soil characteristics within a region contribute to the development of specific wetland plant communities. Yet, while vegetation, soils, and hydrology are the ingredients for wetland ecosystem development, ecosystem attributes derived from these resources are responsible for the ecosystem functions and societal values normally attributed to wetlands.

Wetland attributes of mitigation project sites encompass a wide range of conditions. The HRC proposed that classification of the condition of the vegetation, soils, and hydrology of the pre-project mitigation site can form a conceptual framework for assessment of mitigation plans and would facilitate communication among involved parties in forested wetland mitigation.

The classification system for proposed forested wetland mitigation sites is based upon the working hypothesis that the major process controlling functional effectiveness is carbon flow through the ecosystem. Specifically, ecosystem carbon input is initiated through the primary production of plants. One estimate of the quantity of vegetative production is net primary productivity (NPP), defined as the total amount of carbon fixed through photosynthesis minus that consumed by plants in respiration.

The allocation of NPP within the ecosystem forms the basis of numerous forested wetland functions (Figure 1).

## The MiST classification system

The MiST classification system is composed of three parts. Part I classifies the proposed forested wetland mitigation site with regard to the condition of its plant community, soils, and hydrology (Table 1) and provides the conceptual framework for classifying the sites with respect to their potential for attaining, returning to, or remaining in a high functionally effective state. Attributes included in the classification system were chosen with several considerations in mind:

1) Each attribute contributes directly to the quantity or quality of NPP or the manner in which it is allocated within the forested wetland ecosystem.

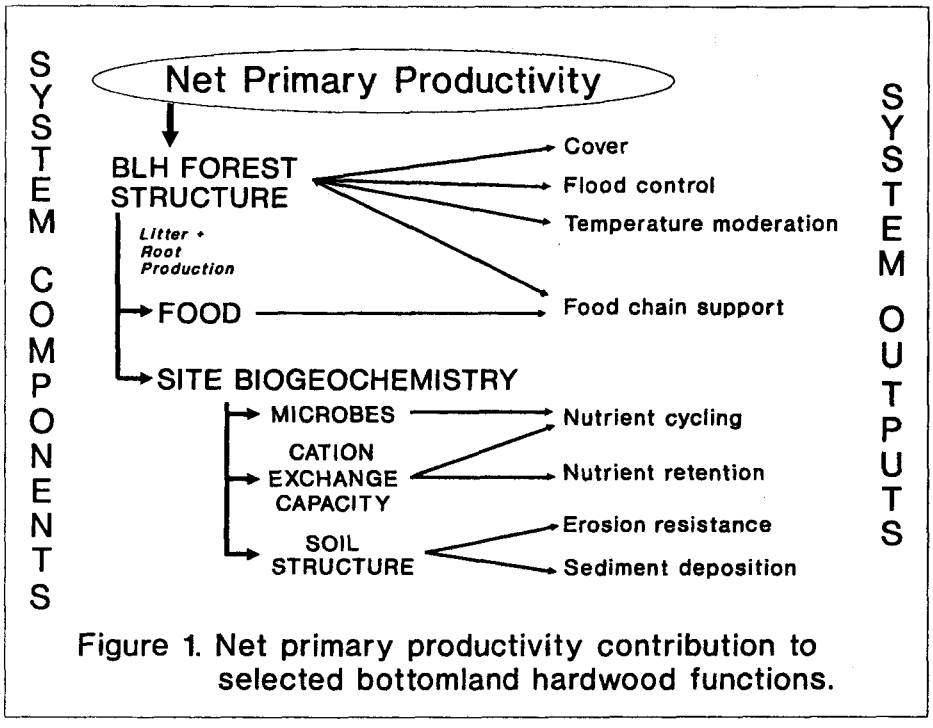



**Figure 1. Net primary productivity contribution to selected bottomland hardwood functions.**

Table 1. The Forested Wetland Mitigation Site Type (MiST) Classification System: Part I. Component Class Definitions.

| COMPONENT/CLASS | DEFINITION |
|---|---|
| VEGETATION | |
| 0 | Site has an overstory and understory species composition and physiognomy similar to the Reference Forest Ecosystem (RFE-see glossary). |
| 1 | Loss, relative to the RFE, of up to 50% of the:<br>a) tree canopy or species composition AND/OR<br>b) undergrowth cover or species composition. |
| 2 | Loss, relative to the RFE, of more than 50% of the<br>a) tree canopy or species composition AND/OR<br>b) undergrowth cover or species composition. |
| 3 | Originally not sufficiently populated with hydrophytic vegetation to be delineated as a wetland or an original forested wetland was removed prior to surface-mining. |
| SOIL | |
| 0 | Site is undisturbed by other than natural means. |
| 1 | Disturbance limited to the top 12 inches of the soil and/or loss of up to 50% of the top 12 inches of the existing soil. |
| 2 | 1) Disturbance within the top 12 inches of the soil with loss of greater than 50% of the top 12 inches of the existing soil, AND/OR,<br>2) Compaction that affects the rooting zone at a degree greater than the reference soil. The significance of the size of the affected area should be determined on an on-site basis. |
| 3 | Reconstructed soil: soil horizon replacement. |
| 4 | 1) Loss of soil profile to a depth greater than 12 inches, OR,<br>2) Loss of the original subsoil structure, OR,<br>3) New soils developed from materials other than original forested wetland soil. |
| HYDROLOGY | |
| 0 | Undisturbed hydrology based on comparison with hydrologic conditions in the RFE. |
| 1 | A deviation in the frequency and duration of not greater than 25% from Class 0 without deviation in the dominant season and source of inundation. |
| 2 | A deviation in the frequency and duration of not greater than 50% from Class 0 without deviation in the dominant season and source of inundation. |
| 3 | A deviation in the frequency and duration of greater than 50% from Class 0 without deviation in the dominant season and source of inundation. |
| 4 | A deviation in the frequency and duration of greater than 50% from Class 0 with deviation in the dominant season and source of inundation. |

Table 2. The Forested Wetland Mitigation Site Type (MiST) Classification System: Part II. Performance Standards for Sites Undergoing Mitigation.

| ATTRIBUTE | PERFORMANCE STANDARDS |
|---|---|
| **Vegetation** | Performance standards are attained when the mitigated forested wetland project sites contain:<br>1) An agency approved composition of canopy and undergrowth species typical of the RFE and represented by self-sustaining species populations.<br>2) An agency approved tree abundance in terms of overall density and spatial distribution throughout the project site.<br>3) Well established trees. A well established tree is one that has been rooted at the mitigation site long enough to survive the range of environmental conditions present on the site. |
| **Hydrology** | At a minimum, mitigated sites (both in-kind and out-of-kind) should:<br>1) Obtain the RFE hydrologic conditions dictated in Class 0 which emphasize the establishment of proper seasonality and source and/or;<br>2) If the vegetative, soil, and water quality conditions performance standards are satisfied within Class I hydrology criteria, hydrologic conditions will be considered successful. |
| **Soil** | A soil will be considered restored if it has the physical and chemical properties that are necessary for the successful re-establishment of the desired RFE. At a minimum, the soil has to be classified as a hydric soil as defined in the Federal Wetland Delineation Manual (1989). |
| **Water Quality** | The performance criteria for acceptable levels of water quality characteristics following mitigation activities is the same for all MiST Classes (vegetation, soils, and hydrology).<br>1. Water quality success will be achieved when the individual frequency of monitored parameter values for the constructed/restored site overlaps an agency approved percentage of the frequency distribution of the RFE when graphically represented. Methodology to determine this should be agency approved.<br>2. Minimally, measured levels of parameters should not violate State standards; it is recognized that applicable, State-established variances for certain wetlands and classes of naturally-deviating surface waters exist. |
| **Habitat** | Successful implementation of the specific mitigation measures for replacing vegetation, soils, and hydrology should provide reasonable and acceptable assurance within the monitoring time frames associated with most regulatory permits that a forested wetland similar to the RFE will occur given sufficient time. When this happens, the wildlife species that reside in or use the new forested wetland should correspond, assuming that the forest is not isolated from the surrounding landscape or other unrecognized limiting factors do not exist. |

2) Greater disturbance of an attribute should reflect larger:

   a) increments of degradation to forested wetland functions and;

   b) increments of complexity and/or magnitude in ameliorating the disturbance.

3) The attribute must be relatively easy to assess with field and/or historical data.

Part II of MiST defines mitigation performance standards to provide focus to the mitigation effort (Table 2). Performance standards describe the minimum thresholds of acceptable recovery (as defined at the workshop) at mitigation sites and are independent of mitigation site condition. They are a standardized list of forested wetland mitigation goals.

Finally, Part III lists the required monitoring and expected levels of performance of each attribute (Tables 3-7). These requirements are tied to the MiST classification. This section of MiST illustrates how severity of mitigation site condition corresponds to eventual

Table 3. The Forested Wetland Mitigation Site Type (MiST) Classification System: Part III. Measurements required for MiST Success Parameters: VEGETATION[1].

| | | PARAMETERS |
|---|---|---|
| **Overstory Criteria:** | | |
| 1) 400 TPA[2] overall >= 6 feet tall | | |
| 2) 400 TPA on every acre | | |
| 3) Approved species present at >= 10 TPA | | |
| Class 1: | | Determine if all criteria were met at end of the second year. |
| Class 2: | a) | Assess potential for natural recovery. If inadequate, prepare and implement plan for tree planting. |
| | b) | If applicable, assess survival of planted trees following first growing season. |
| | c) | Assess tree species density and height following subsequent growing seasons. |
| Class 3: | a) | Prepare and implement plan for tree planting. |
| | b) | Assess survival of planted trees following first growing season. |
| | c) | Assess tree species density and height following subsequent growing seasons. |
| **Understory/Herbs Criteria:** | | |
| 4) 10% of RFE[3] understory/herbs represented | | |
| 5) <10% nuisance species cover present | | |
| Class 1: | | Determine if understory/herbs and nuisance species criteria were met after 2 year period. |
| Class 2: | | Prepare annual lists of all preferred and nuisance understory/herbs species on a per acre basis. |
| Class 3: | | Prepare annual lists of all preferred and nuisance understory/herbs species on a per acre basis. Determine cover for each category. |

[1]It is recommended that a report be prepared and submitted to the permitting agency at each assessment. In addition, assessment methods should be agreed upon prior to mitigation activities.

[2]TPA = Trees per acre. Criterion #1 accounts for variation in productivity across the site; Criterion #2 refers to the distribution of stems across the site.

[3]RFE = Reference forest ecosystem (see glossary).

requirements for monitoring, viz, a higher MiST classification level (i.e., greater degradation) generally requires that monitoring be conducted more intensely.

MiST assumes the proposed mitigation project site will be mitigated. That is, permit sequencing procedures have determined the need for compensatory mitigation. MiST can be used for either on-site or off-site mitigation.

## Objectives and benefits of MiST

MiST establishes performance criteria for proposed forested wetland mitigations based upon attributes of the pre-project mitigation site. By providing a conceptual framework through which to objectively view the mitigation, MiST assists in standardizing the evaluation of mitigation plans by regulatory authorities.

It was recognized early in the workshop that the concept of a reference forest ecosystem (RFE - see glossary) was fundamental to the process. The RFE provides mitigation plans with specific targets or goals towards which to work (Brinson and Lee 1989). Concurrently, MiST supplies mitigators with an up-front understanding of the expectations of the restored ecosystem. Although MiST establishes standardized objectives, it allows mitigators the freedom to develop innovative mitigation designs.

## The relationship of MiST and WET

MiST differs from the Wetland Evaluation Technique (WET, Adamus et al. 1987) in several important ways. WET identifies and ranks current functional opportunity, social significance, and effectiveness of specific sites. In contrast, MiST assesses the potential for restoration of a proposed forested wetland mitigation site by defining the existing conditions of site attributes that contribute to functional effectiveness. MiST also assigns goals for site restoration and requirements for monitoring. MiST is site-

Table 4. The Forested Wetland Mitigation Site Type (MiST) Classification System: Part III. Measurements required for MiST Success Parameters: SOILS.

| | CLASS | | | |
|---|---|---|---|---|
| | 1 | 2 | 3 | 4 |
| REFERENCE FOREST ECOSYSTEM | ABC [1,4] | | | |
| INITIAL SITE CHARACTERIZATION[2] | ABC | ABC | ABC | ABC |
| RESTORATION/ RECLAMATION [3] | <---------- A as needed ----------> <---- B C ----> (min. 2 years) | | <----B C----> (min. 5 years) | |

1 For the reference forest ecosystem (Class 0): A factors measured in rooting zone; B and C factors measured at 0 - 9 inch depth.

2 Factor A to be measured in the rooting zone as defined in the RFE. Factors B and C to be measured at:
Class 1, 2 : 0 - 9 inch depth
Class 3 : by horizon
Class 4 : by horizons or depth as determined by backfill placement technique

3 All factors (as defined in footnote 2 above) measured on an annual basis.

4 Code for chemical and physical factors to be measured on the reference and mitigation sites:

A Physical: Bulk density, soil strength, texture, permeability (to assess the ability to establish hydric soil conditions)

B Chemical: Potential phytotoxic/micronutrient conditions: pH, pyritic sulphur, neutralization potential, Al Cu, Zn, B, Mn, base saturation, conductivity, redox potential

C Chemical: Macronutrients: N, P, K, organic C

specific; it does not address functional opportunity or social significance.

MiST assumes by default that the chosen forested wetland mitigation site ranks low in BLH-WET functional effectiveness. A higher WET ranking would reduce the need for or functional payback of restoration. The goals and monitoring requirements assigned to the condition of a mitigated forested wetland attribute are considered to be those needed to return the site to a WET medium or high functionally effective site.

## Rationale for the MiST classification hierarchy

For all attributes, Class 0 conditions are those present within a representative RFE possessing a species composition and functional opportunity (Adamus et al. 1987) similar to that present or desired on the site to be mitigated. Each increment in attribute class ranking represents an increase in dysfunction and/or magnitude or complexity of amelioration.

Once the vegetation, soils, and hydrology of a mitigation site are classified, the intensity of monitoring is defined for each attribute. Increasing monitoring intensity is reflected by an increase in: a) the number of variables to be evaluated; b) the frequency and periodicity of data collection and/or; c) the period of time needed for assessment to be completed.

In addition to vegetation (Table 3), soils (Table 4), and hydrology (Table 5), characteristics for water quality are also monitored (Table 6). This latter attribute is included in monitoring requirements because, along with vegetation, it represents measures of ecosystem functional development that the more basic driving attributes of soil and hydrology do not evaluate.

Water quality performance standards are the same regardless of the MiST attribute class. In practice, a highly degraded site may be expected to require more time to reach an acceptable level of

Table 5. The Forested Wetland Mitigation Site Type (MiST) Classification System: Part III. Measurements required for MiST Success Parameters: HYDROLOGY.

| | PARAMETERS |
|---|---|
| Class 0 | No monitoring is required. |
| Class 1 | Frequency and Duration<br>1) Semi-annual visual observation of site during dormant and early part of growing season.<br>2) Follow-up visits to determine duration plus visual observation of drift lines, sediment on leaves, silt lines on trees, etc. |
| Class 2 | Frequency and Duration<br>Quarterly monitoring visits coupled with a continuous recording device (combination piezometer/crest gage) with a frequency of recording not greater than seven days; couple recorded data with visual observations. |
| Class 3 | Frequency and Duration<br>Monthly monitoring visits coupled with a continuous recording device. |
| Class 4 | Frequency, Duration, Seasonality and Source.<br>Same as Class 3 |

Table 6. The Forested Wetland Mitigation Site Type (MiST) Classification System: Part III. Measurements required for MiST Success Parameters: WATER QUALITY.

| | PARAMETERS | |
|---|---|---|
| Analyses[1,2] | Surface water | Ground water |
| Field[3]: | | |
| Temperature | X | |
| pH | X | X |
| Conductivity | X | X |
| Dissolved $O_2$ | X | |
| Redox potential (Eh) | | X[4] |
| Lab: | | |
| Alkalinity | X | X |
| Suspended solids | X | |
| TOC\ TOC/TON TON/ | X | |

1 See Hem (1985) for more detailed descriptions. EPA quality assurance is implied. Minimum required analyses; additional analyses may be added for special cases such as sites formerly occupied by mines, industry, or other intensive land use.

2 The ecosystem parameters listed in this table are to be monitored for all MiST Classes. Where MiST Soils Classification 3 or 4 are determined, additional parameters judged appropriate may be added to this list of mandatory characteristics for ground water and surface water monitoring.

3 Mitigation site and RFE should be treated as a paired watershed comparison. Paired sites should be measured at nearly the same time of day because of anticipated diel fluctuations.

4 Precautions should be taken to assure that in situ values are not altered in the process of measurement.

Table 7. The Forested Wetland Mitigation Site Type (MiST) Classification System: Part III. Measurements required for MiST Success Parameters: HABITAT.

| PARAMETERS |
|---|
| PHASE I - REFERENCE FOREST ECOSYSTEM ANALYSIS |
| A. Determine if endangered/threatened species are present. |
| B. Develop species lists. |
| C. Select evaluation species based on perceived importance, indicator status, etc. |
| D. Evaluate habitat quality for selected species. |
| E. Determine relationship of reference site to surrounding landscape (interspersion among other habitat types, total area of reference type, etc.) |
| PHASE II - MONITORING DURING PERMIT REGULATORY PERIOD (Assumed to be at least 5 years, with maximum of 10) |
| A. Use MiST soils, hydrologic, and vegetation monitoring criteria as acceptable long-term habitat mitigation success; assumes acceptable habitat values for most indicator species will be met. |
| B. For MiST Class 2 Vegetation or Class 3 or 4 Hydrology, calculate habitat suitability index values (Schamberger et al. 1982) for selected evaluation species and community characteristics known to be important to wildlife (e.g., size of area, interspersion factors) during the following periods: 1) One year after mitigation plan is implemented. 2) Midway through regulatory period. 3) Immediately prior to bond release. |
| C. For MiST Class 2 Vegetation or Class 3 or 4 Hydrology, ensure that short term habitat improvement practices were implemented (means will vary according to practices employed). |
| PHASE III - LONG-TERM MONITORING (Optional) |
| A. Follow-up study by management entity (to be identified in mitigation plan) to compare baseline values with post mitigation values with goal of replacement of habitat type(s) and associated values. |

success. However, there is no a priori reason to alter the water quality performance standards because of initial site condition.

Monitoring of wildlife habitat is not specifically required by MiST (Table 7). Selection and use of meaningful criteria for measuring and evaluating the performance of replacing habitat factors for species that use a mature forested wetland are unclear given the typical regulatory time frames associated with wetland permitting. Consequently, the attainment of vegetative, soil, and hydrologic performance standards is assumed to result in the development of appropriate forested wetland habitat provided that habitat isolation problems or other limiting factors do not exist on the mitigation site. Wildlife quantity and diversity will be maximized when the mitigated site is not isolated from the surrounding landscape and has dimensional attributes such as edge length and shape, size, etc. for favored species. Other ecosystem functions should not be compromised in the enhancement of favored species, however.

In lieu of no specific habitat monitoring requirements, some site-specific habitat compensation measures may be requested in conjunction with state fish and wildlife agencies and the USFWS.

## Using MiST: An example

### Step 1: Identify Reference Forest Ecosystem

Of primary importance to the use of MiST is the selection of the RFE. The RFE forms the basis for comparison of many monitored characteristics. Selected by the applicant, it is approved by the regulatory agency preferably prior to classifying the site to be mitigated. While not specifically required, ideally the RFE should be located in the same or an adjacent watershed to the mitigation site. If possible, it should lie in a similar landscape position relative to upstream and downstream activities in its watershed and represent similar functional opportunity and significance (Adamus et al. 1987) as the site to be mitigated. However, the potential difficulty of identifying an ideal RFE for particular projects in specific locations render RFE selection as a negotiable item between the mitigator and the regulatory authority.

Selection of the RFE requires characterization of the vegetation, soils, and hydrology. While characteristics to be measured are identified in MiST, the methods used to assess each component are negotiable between the mitigator and agency personnel. Sources of information

concerning characterization of the RFE (as well as the mitigation site) can be found in the Federal Manual for Identifying and Delineating Jurisdictional Wetlands (Federal Interagency Committee for Wetland Delineation 1989).

For the purposes of evaluating water quality, the permit applicant may choose to select more than one RFE. This is recommended if there is a possibility that background conditions (i.e., water quality of source waters to the wetland) may change significantly within regulatory time frames due to changes in land use, point source discharges, or water flow alterations. Also, the wide variation among natural ecosystems and the potential difficulty in identifying an ideal RFE further argues for multiple RFEs.

Step 2: Classify the vegetation, soils, and hydrology of the mitigation site.

Once the RFE has been chosen, the site to be mitigated is classified using the definitions found in Table 1. Combinations of historical and/or field data may be required to classify hydrologic conditions. In the latter case, where historical data is not available, monitoring may be required to accurately classify the site.

**Example 1: Bean field conversion**

Suppose a forested wetland permit applicant suggests use of an off-site wetland formerly cropped to soybeans for mitigation. Good management practice yielded no significant erosion or compaction, but soil management (plowing, cultivating, etc.) has disturbed the surface layer of the forested wetland soil. Hydrology is undisturbed but complete removal of the forest cover has occurred.

**MiST classification:**
**Vegetation (V) = 2**
**Soils (S) = 1**
**Hydrology (H) = 0**

**Example 2: Surface-mined land**

Suppose an applicant wishes to perform on-site mitigation to compensate for a BLH to be surface-mined. Vegetation will be completely removed from the site prior to mining. The soil will be reconstructed with six inches of original A horizon material over graded cast overburden. The A horizon material will be stockpiled for two years prior to replacement. Based upon historical hydrologic data, it is determined that regrading to approximate original contour will yield a site with a flooding frequency that is 40% less than the RFE.

**MiST classification:**
**V = 3**
**S = 4**
**H = 2**

Although the site was originally BLH, V3 classification is applied since the area will be surface-mined. The rationale for this classification lies in the potential dilution of viable forested wetland propagules following soil handling, stockpiling and transport during the surface-mining operation. This results in a potential reduction of the rate of natural revegetation. Moreover, it is widely held that the substantial impact that surface-mining operations has on a site renders these sites as mitigation creations regardless of the original site status. S4 classification is applied to the rooting medium since complete soil horizon replacement (A + B) will not be exercised and, thus, the medium will not have the original subsoil structure. Moreover, subsoils will be developed from materials other than the original forested wetland soil. H2 classification was derived from historical data of adjacent water flows in conjunction with proposed post-grading contours.

Step 3: Execution of forested wetland mitigation plan

Within certain limits, freedom of mitigation designs is allowed all permittees. Table 2 sets the performance standards of all forested wetland mitigation plans. While innovation in design or approach is encouraged, minimum standards will be applied to all forested wetland mitigations, regardless of original MiST classification. In certain cases, site assessments will be made during the preparation of mitigation plans. For example, V2 sites require evaluation of natural regeneration potential of a site. This would be required prior to the execution of the mitigation project since if natural regeneration is not possible, tree planting will need to be incorporated in mitigation plans.

Step 4: Monitoring of forested wetland mitigation site

The original MiST classification manifests in the level of intensity required during the post-construction monitoring program (Tables 3-7). To illustrate, consider the classification examples given above.

**Example #1: Bean field, MiST classification V2, S1, H0**

Class 2 Vegetation requires an assessment of natural recovery. All sites require 400 trees per acre (TPA) on every acre and an overall average of 400 TPA that are at least six feet tall of an approved species composition (Table 3). Potential for natural regeneration of an approved species mix is evaluated. Assuming, in the case of a soybean field, that woody seedwalls are too distant and/or coppice stems are not available, tree planting would be required over at least part of this site. Following this assumption, then, survival tallies of planted

trees would be required after the first growing season (Table 3). Subsequent growing seasons would require species density and height data to be recorded. Positional records ensure adequate stem distributions across the site. Understory vegetation will be tallied and grouped annually according to negotiated "preferred" and "nuisance" species.

Soil restoration requires that physical and chemical properties necessary for successful re-establishment of the desired RFE be present and that, at a minimum, the soil should be hydric (Table 2).

Soil monitoring occurs in two phases. Phase I is the initial characterization of the site. This activity occurs following the impact to the site and establishes a baseline for project site soils prior to the execution of the mitigation plan. Comparison of the post-impact soil conditions with RFE soils serves as a guide for developing soil-specific mitigation procedures. Phase II monitors soil properties after the mitigation project has been completed. Class 1 soil classification requires the mitigator to monitor numerous physical characteristics as needed and chemical parameters for a minimum of 2 years following the completion of the project (Table 3).

Since hydrology is undisturbed, no hydrologic monitoring is necessary during construction or after the mitigation project has been completed.

A primary function of forested wetland ecosystems is the removal of sediments, nutrients, and toxins from floodwaters and overland flows thereby improving water quality. Water quality monitoring will always occur regardless of the MiST classification (Table 2). Measured parameters are listed in Table 6. Water quality standards will be considered achieved when the individual frequencies of each parameter overlap an accepted proportion of the frequency distribution of the RFE when graphically represented. At a minimum, all parameters must meet established state water quality standards. Methodology for determining the overlap criterion is negotiable, but data are to be acquired monthly for at least two years and must include at least four peak flows.

Habitat compensation measures would be site and project specific (Haynes et al. 1990). For example, mitigators might be expected to install and maintain nesting boxes for wood ducks, or establish brush piles for cover. Other compensation measures might include leaving buffer zones near streams, or establishing food plots within the mitigation site.

**Example #2: Surface-mined land: MiST classification**

**V3, S4, H2**

V3 classification assumes that tree planting will be required; the potential for adequate natural regeneration is considered minimal. Otherwise, vegetation monitoring would be identical to a V2 classification.

Following reconstruction, soil monitoring will consist of measurements of physical and chemical properties (Table 4) on an annual basis for a minimum of five years. These characteristics include physical properties that could be potentially limiting to such things as root elongation and water infiltration as well as toxic and nutrient chemical levels. Class S4 requires that these properties be measured throughout the reconstructed soil profile. The exact depths would be dictated by the backfill placement technique that was used during reconstruction.

Hydrologic monitoring of the site would require quarterly visits coupled with a continuous recording device such as a combination piezometer/crest gage with a frequency of data recording not greater than seven days. Monitoring schedules assume that the RFE is paired with the proposed mitigation site in a manner similar to a paired catchment study. Quantitative data should be coupled with visual observations to demonstrate the establishment of RFE hydrological conditions found in Class 0. These conditions emphasize re-establishment of proper seasonality and source. When local stream gaging data are available and correlation of the gaging data can be made with at least one year of reference and mitigation site monitoring data, gaging station data may be substituted for on-site monitoring.

Since water quality and habitat monitoring are not tied to the site conditions, monitoring of these site attributes following the execution of the mitigation plan are identical to those found in the requirements for Example #1.

## Ramifications of MiST

What impact will MiST have on forested wetland compensatory mitigation? Importantly, MiST does not attempt to dictate mitigation designs. Thus, the mitigator is free to choose the methods for re-establishment of the ecosystem. For example, while MiST does require 400 TPA, it does not set the protocol for obtaining this result. The mitigator could choose to plant seedlings or saplings, use direct-seeding, natural regeneration, or combinations of these techniques to produce the desired vegetation. Similarly, while MiST requires that chemical and physical conditions of the soil must agree with those present on the RFE, it does not establish how the soil must be

reconstructed to ensure that this occurs. Selective mixing or complete replacement of soil horizons would be two of several options available to meet soil requirements depending on the site and the materials found there.

While some may prefer strict standardization in MiST, the need to foster innovation in this new field of forested wetland mitigation technology requires flexibility in mitigation design in order to compare the suitability of different plans. An important caveat, however, is that it is incumbent upon the mitigator to ensure that the methodology is adequate to produce a fully functional forested wetland ecosystem. This is the only acceptable end result of forested wetland mitigation. Thus, the mitigator must carefully consider the advantages of specific designs with their risk of not attaining performance standards.

## Conclusions

This summary of the MiST classification system is the result of collaboration between industry, regulatory agencies and academic experts across the Southeast. It summarizes a more extensive draft document currently in review by EPA Region IV. The MiST system will require extensive testing prior to its adoption to ensure that the variety of conditions that exist on the natural landscape are properly represented and to streamline field procedures during its use.

Regardless of the refinements made to the system following field testing, a major strength of MiST is that it provides an avenue for communication between regulators and mitigators by establishing a standard approach for evaluation of mitigation plans. And, important to restoration ecologists, it provides a framework for illuminating the most pressing needs for restoration research relating to forested wetland ecosystems.

While MiST addresses concerns related specifically to Section 404 of the CWA, an additional benefit of MiST is the potential for its use outside of CWA regulatory programs. Classification of mitigation sites and development of performance criteria and monitored characteristics can lead to better restoration of lands such as those currently under consideration for conversion to wetland forest ecosystems through programs initiated in the Conservation Reserve Program and the "Swampbuster" provision of the Food Security Act of 1985 (Farm Bill). Thus, use of MiST may not only assist in mitigation of wetland disturbances but may also contribute to increasing the quality and quantity of the nation's wetland resource.

## Acknowledgements

The authors gratefully acknowledge the work of the participants of the Workshop on BLH Forest Mitigation of Disturbed Sites whose deliberations are summarized in this paper. Also, Region IV EPA is thanked for their support of this effort. Finally, the senior author thanks Drs. Doug Frederick and David Adams of the NCSU Department of Forestry for their insightful review of the draft manuscript.

## GLOSSARY

**CANOPY:** The uppermost stratum of trees in the reference forest ecosystem.

**COMPACTION:** Degree of firmness in the soil. When present at a high degree, it reduces water movement and limits plant root penetration. Relative degrees can be determined by comparing bulk density and/or soil strength (e.g., as measured with a constant rate penetrometer).

**DURATION:** The average length of time in months that inundation and/or saturation occurs each year.

**FREQUENCY:** The number of inundation and/or saturation events that occur on the average each year. At least one inundation / saturation event must occur on the average each year to meet Federal guidelines.

**HABITAT:** The total of environmental conditions of a specific place occupied by a wildlife species or a population of that species. It can be described in terms of food, water, cover, and any other recognized life requisites and their relative location (interspersion) within a given area.

**HABITAT SUITABILITY INDEX (HSI):** A habitat quality measure defined as a value between 0.0 and 1.0, with 1.0 representing the optimum habitat quality in a defined area for the evaluation species.

**PRINCIPAL HYDROLOGIC CONDITIONS:** Frequency, duration, seasonality and source of site inundation and/or saturation.

**NATURAL DISTURBANCE:** Physical processes (i.e., soil scouring, sediment deposition) normally associated with inundation of floodplain zones.

**NEW SOIL:** Recently deposited or drastically altered soil profiles atypical of undisturbed soils within the reference area (e.g., dredge spoil, mine tailings, mixed mine soil, overburden, construction backfill material).

**NO NET LOSS:** The suggested interim goal of a national wetlands protection policy emphasizing achievement of no overall net loss of the existing wetlands resource base. No net loss is achieved when wetland gains equal wetland losses (Conservation Foundation, 1988).

**NUISANCE SPECIES:** Plant species having the potential to threaten success prior to or following project release.

**PREFERRED SPECIES:** Plant species typical of the reference forest ecosystem to be mitigated. Those preferred species upon which project success is to be determined will generally exclude:

1) Exotics
2) Aggressive colonizing weeds of open environments
3) Non-persisting canopy gap herbs
4) Off-site species that are more typical of other ecosystems
5) Rhizomatous grasses with the propensity to form turfs.

**REFERENCE FOREST ECOSYSTEM:** The kind of forest selected for creation or restoration, as it is represented locally (same or nearby watershed) in terms of species composition and physiognomy. It is incumbent upon the applicant to characterize the reference forest type to the satisfaction of the regulatory authority.

**REFERENCE SOIL:** Soil type(s) associated with the reference forest ecosystem.

**SEASONALITY:** The season or seasons (growing and dormant) during which the dominant period of inundation and/or saturation occurs. The dominant season of inundation cannot be different from the reference forest ecosystem, otherwise a different forest ecosystem would develop over time.

**SOURCE:** The principal source of inundation and/or saturation such as riparian (upland) discharge, overbank flow and rising groundwater. The dominant source of inundation cannot be different from the reference forest ecosystem.

**UNDERGROWTH:** All vascular plants that do not contribute to the canopy of the reference forest ecosystem, except as vines or epiphytes, including herbs, vines, shrubs and small trees.

**UNDISTURBED NATURAL AREAS:** Forested wetland communities that do not exhibit evidence of an adverse impact by man-made activities (e.g., logging, grazing, agriculture, construction runoff and sedimentation).

**WETLAND HYDROLOGY:** The hydrologic factors such as frequency, duration, seasonality, and source of inundation and/or soil saturation resulting in maintenance of a reference forest ecosystem (as further defined in the Vegetation Criteria Section). By definition, the reference forest ecosystem must meet Federal criteria for jurisdictional delineation as a wetland.

## LITERATURE CITED


Adamus, P.R., E.J. Clairain, Jr., R.D. Smith, and R.E. Young. 1987. Wetland Evaluation Technique (WET); Volume II: Methodology. Operational Draft Technical Report Y-87-__, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Brinson, M.M. and L.C. Lee. 1989. In-kind mitigation for wetland loss: Statement of ecological issues and evaluation of examples. In: R.R. Sharitz and J.W. Gibbons (eds.). Freshwater Wetlands and Wildlife. USDOE CONF-8603101, Office of Scientific and Technical Information, US Department of Energy, Oak Ridge, TN.

Conservation Foundation, The. 1988. Protecting America's Wetlands: An Action Agenda. The Final Report of the National Wetlands Policy Forum. Harper Graphics, Waldorf, MD. 69 p.

Federal Interagency Committee for Wetland Delineation. 1989. Federal Manual for Identifying and Delineating Jurisdictional Wetlands. U.S. Army Corps of Engineers, U.S. Environmental Protection Agency, U.S. Fish and Wildlife Service, and U.S.D.A. Soil Conservation Service, Washington, D.C., Cooperative technical publication. 76 p. plus appendices.

Haynes, R.J., W. Davis, E. Clairain, R. Bay, J. Sandusky and J.A. Allen. MiST Classification Development: Habitat Working Group. In: T.A. White, J.A. Allen, S.F. Mader, D.L. Mengel, D.M. Perison and D.T. Tew (eds.). MiST: A Classification System for Pre-Project Mitigation Sites and Criteria for Determining Successful Replacement of Forested Wetlands. (DRAFT). EPA Region IV, Wetlands Planning Unit, Atlanta, GA.

Hem, J.D. 1985. Study and Interpretation of the Chemical Characteristics of Natural Water. Third edition. U.S. Geological Survey Water-Supply Paper 2254. U.S. Government Printing Office, Washington, D.C. 263 p.

Roelle, J.E., G.T. Auble, D.B. Hamilton, R.L. Johnson, and C.A. Segelquist (eds.). 1987a. Results of a workshop concerning ecological zonation in bottomland hardwoods. U.S.F.W.S., National Ecology Center, Ft. Collins, CO. NEC-87/14. 141 p.

Roelle, J.E., G.T. Auble, D.B. Hamilton, G.C. Horak, R.L. Johnson and C.A. Segelquist (eds.). 1987b. Results of a workshop concerning impacts of various activities on the functions of bottomland hardwoods. U.S.F.W.S., National Ecology Center, Ft. Collins, CO. NEC-87/15. 171 p.

Roelle, J.E., G.T. Auble, D.B. Hamilton, R.L. Johnson, and C.A. Segelquist (eds.). 1987c. Results of a workshop concerning assessment of the functions of bottomland hardwoods. U.S.F.W.S., National Ecology Center, Ft. Collins, CO. NEC-87/16. 173 p.

Schamberger, M.L., A.H. Farmer, and J.W. Terrell. 1982. Habitat suitability index models: Introduction. U.S. Fish and Wildlife Service. National Ecology Center, Ft.Collins, CO. FWS/OBS-82/10. 2 p.

White, T.A., J.A. Allen, S.F. Mader, D.L. Mengel, D.M. Perison, and D.T. Tew (eds). 1990. MiST: A Classification System for Pre-Project Mitigation Sites and Criteria for Determining Successful Replacement of Forested Wetlands. (DRAFT). EPA Region IV, Wetlands Planning Unit, Atlanta, GA.

This page intentionally left blank.

# Wetlands - General

This page intentionally left blank.

# FORESTED WETLAND RECLAMATION SUCCESS CRITERIA DEVELOPMENT IN NORTH FLORIDA[1]

by

J. H. Wester[2]

Abstract. Criteria for determining successful reclamation of forested wetlands in North Central Florida were established in 1987 by a Multi-Agency Group represented by federal and state environmental agencies, environmental action groups, and Occidental Chemical Company (OxyChem). OxyChem mines phosphate in North Central Florida and requires dredge and fill permits to mine in wetland areas. Three milestones were established for 4 demonstration sites for monitoring to be conducted quarterly and evaluated every two years. The 1989 data show the results of the first milestone monitoring period for ten criteria including tree density, tree diversity, growth rate, ground cover, seed production, hydrology, water quality, wildlife abundance, wildlife richness and soils. Vegetational monitoring employed belt transects (elongated line-strip quadrats), hydrology by water level recording, water quality by standard chemical methods, wildlife abundance and diversity by macroinvertebrate sampling and soils by percent organic matter. Results indicate the achievement of the Milestone 1 criteria as established by the Multi-Agency Group.

Additional key words: wooded wetland restoration, phosphate wetland reclamation.

## Introduction

Occidental Chemical Corporation *(OxyChem)* conducts surface mining of phosphate in North Central Florida. The two mines, Suwannee River and Swift Creek, can produce 3.0 and 2.5 million tons of phosphate rock per year. Suwannee River Mine began operation in 1965 and Swift Creek Mine in 1975. Approximately 6300 hectares (ha) (15,567 ac.) have been mined to date at the two sites. The operations are located in Southeastern Hamilton County, which is just below the Florida/Georgia border.

The area is generally classified as pine flatwoods interspersed with cypress domes and bayheads. Approximately 25 to 30 percent of the area would be classed as wetlands, mostly forested wetlands.

Land reclamation in Florida for phosphate mining has been mandatory since 1975. Companies mining phosphate are required to submit programs prior to mining and reclamation cannot begin until the programs are approved. The Florida Department of Natural Resources (FDNR) is the regulatory agency for mine reclamation activities.

In 1986, the U.S. Army Corps of Engineers completed an Environmental Impact Study *(EIS)* of the area within OxyChem's existing mining operation. As part of an agreement on the permit issued as a result of the *EIS*, a Memoranda of Understanding *(MOU)* was developed with success criteria for forested wetland establishment. These criteria were developed by members of the Multi-Agency Group including personnel from U.S. EPA (USEPA), U.S. Army Corps of Engineers (COE), Florida Department of Environmental Regulation (FDER), Florida Department of Natural Resources (FDNR), Florida Game and Fresh Water Fish Commission (FG&FWFC), U.S. Fish and Wildlife Service (USF&WS), Suwannee River Water Management District (SRWMD), Florida Defenders of the Environment (FDE) and OxyChem. The Multi-Agency Group was responsible for the final decisions on the success criteria. A subset of the Multi-Agency Group called the Success Criteria Working Group was composed of representatives of COE, FDER, USEPA, FG&FWFC, and OxyChem. The working group was responsible for working out the details on specific criteria and forwarding recommendations to the larger group. More than a dozen meetings of the two groups were held in Tallahassee and White Springs and many criteria were suggested, discussed, considered, sometimes initially rejected and later considered again prior to adoption of the final set of criteria.

Criteria that were considered had to be *(1) measurable, (2) reasonable, and (3) good indicators of wetlands.* The ability to make valid measurements of the criteria was important to all parties involved. Many suggested criteria were considered important characteristics of wetlands but had no quantiative measurement. Many aesthetic indicators of wetlands, while important to all, were difficult to measure to any reliable degree and were therefore rejected. Past experience of the mining industry was researched along with work done at the Center for Wetlands at the University of Florida in Gainesville. Advice was obtained

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]J. H. Wester is Reclamation Superintendent, Occidental Chemical Corporation, White Springs, Florida 32096

Table 1. Wetland Reclamation Success Criteria for Project Areas

| PARAMETER | MILESTONE I (No later than 11/30/89) | | | | MILESTONE II (No later than 11/30/91) | | | | MILESTONE III (No later than 11/30/93) | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | SP4 | SR8 | GA | SA1 | SP4 | SR8 | GA | SA1 | SP4 | SR8 | GA | SA1 |
| Density (trees/acre) | 200 | 200 | 300 | 500 | 200 | 200 | 200 | 450 | 200 | 200 | 200 | 400 |
| Diversity | | | | | | | | | | | | |
| % Taxodium | N/A | N/A | 75-80 | 50-60 | N/A | N/A | 75-80 | 50-60 | N/A | N/A | 75-80 | 50-60 |
| % Nyssa[1] | N/A | N/A | N/A | 20-25 | N/A | N/A | N/A | 20-25 | N/A | N/A | N/A | 20-25 |
| % other | N/A | N/A | 20-25 | 20-25 | N/A | N/A | 20-25 | 20-25 | N/A | N/A | 20-25 | 20-25 |
| Growth Rate[2] (% increase) | 100 | 200 | 50 | 50 | 200 | 300 | 100 | 100 | 300 | 400 | 200 | 200 |
| Ground Cover[3] (areal (extent) | 70 | N/A | 70 | 70 | 70 | N/A | 70 | 70 | 70 | N/A | 70 | 70 |
| Seed Production (% of Taxodium producing seed) | N/A | 5 | N/A | N/A | 5 | 10 | N/A | N/A | 10 | 10 | 5 | 5 |
| Hydrology[4] | N/A | N/A | N/A | note 4 | N/A | N/A | N/A | note 4 | N/A | N/A | N/A | note 4 |
| Water Quality[5] | N/A | N/A | --see note 5-- | | N/A | N/A | --see note 5-- | | N/A | N/A | --note 5-- | |
| Wildlife Abundance[6] | --- | N/A | --see note 6-- | | --- | N/A | --see note 6-- | | --- | N/A | --note 6-- | |
| Wildlife Richness[6] | --- | N/A | --see note 6-- | | --- | N/A | --see note 6-- | | --- | N/A | --note 6-- | |
| Soils (% organic matter in top 5 cm) | 0.5 | 0.5 | 0.5 | 0.5 | 0.5 | 0.5 | 0.5 | 0.5 | 1 | 1 | 1 | 1 |

1. N. sylvatica var. biflora and N. aquatica shall each comprise at least 5% of the trees in the SA-1 demonstration area. Together, Nyssa shall comprise at least 20% of the trees in the demonstration area.

2. Mean growth rate based on assumption that all plantings were 18 inches when planted for SP4 and SR8 projects. Green Area and SA1 projects will have initial seedling height measurements when planted. Growth rates will be averages across all species; however, Taxodium and Nyssa must each show at least 20% growth.

3. Cover is based on areal extent by species on the Section 17-4.022, FAC, wetland vegetation list. Permanent open water bodies will not be included in this calculation.

4. The entire wetland area in SA1 (approximately 10 acres) will be inundated for at least 90 days per year, of which at least 30 days will be consecutive. For one period of at least 30 consecutive days per year, at least 50% of the wetland area will not be inundated more than 12 consecutive hours.

5. TSS, TDS, turbidity, chlorophyll a, orthophosphate, total phosphate, ammonia nitrogen, nitrate/nitrite, and total nitrogen will be measured and evaluated based on applicable Chapter 17-3, FAC, Class III standards.

6. Macroinvertebrates will be used to measure faunal success based on the Shannon-Weaver diversity index value as defined in Section 17-3.021(24), FAC. The station will be sampled until at least 100 organisms are obtained or the sample will not count towards demonstrating success.

   The criteria will be deemed successful once the Shannon-Weaver diversity index values from the individual reclamation areas average at least 2.0, 2.25 and 2.5 for Milestones I, II, and III, respectively, for four consecutive sampling periods.

   Corps, EPA, and DER staff will be invited to participate in selection of sampling stations. Two stations will be sampled in SA1. Four stations will be sampled in the SP4 project, two in each portion of the wetland areas in the SP4 project. Six stations will be sampled in the Green Area.

   The raw data, diversity, equitability, total number of organisms, and total number of species will be provided to the Corps, EPA, and DER. Macroinvertebrate diversities will be used to assess the quality of the macroinvertebrate communities in the wetland demonstration areas. They will not be used to apply DER's biological integrity standard for class III waters. The generated diversity values will be utilized in determining restoration success and will not be used as a direct measure of water quality.

from university researchers as to expected growth characteristics of natural wetlands. Whether a criterion was a good indicator of wetland success was debated a great deal. Individuals have different opinions as to what makes a good indicator. If wildlife species, for example, are to be used, which one will be chosen? In many cases, a new wetland provides many benefits although a selected wildlife species may not be initially present. It may be present in later development of the wetland. All of these considerations made the exercise a challenge and provided some valuable insight into the concerns of those on all sides of the mining and reclamation issue. It certainly made everyone appreciate the difficulty in establishing realistic criteria.

In July 1987, the Regional Administrator for EPA and the Secretary of the Florida Department of Environmental Regulation accepted the criteria and the schedule for implementation of three milestones. *(See Table 1.)* Four demonstration plots were established on recently planted or soon to be planted areas. The areas were described and transects established. They are the wetland portions of:

1) Settling Area No. 1 (SA-1),
2) Special Project OCC-SR-SP(4) (SP-4),
3) Land and Lakes Project OCC-SR-8 (SR-8), and
4) The Green Area Project OCC-SR-83(2) (GA)

## Study Area

The total SA-1 area is approximately 40.5 ha (100 ac.). It has been used as a phosphatic clay settling area since 1965. As the area was exhausted (filled with clays) a portion of the area, approximately 36.4 ha (90 ac.), was capped with approximately 2.4 m to 3.1 m (8 to 10 ft.) of sand tailings, creating a higher well drained upland area. The wetland area of approximately 4.05 ha (10 ac.) had not been disturbed since 1980 and was dominated by large willows (Salix nigra) and red maple (Acer rubrum) some reaching 15 to 20 cm (6 to 8 in.) diameter at breast height (DBH) and 6 to 7.6 m (20 to 25 ft.) tall. In order to compare the affect of the established canopy on young planted seedling, half the area was cleared in strips across the wetland. Tree seedlings were then planted in both the cleared and uncleared portions. Each strip was approximately 30.5 m (100 ft.) wide and three fertilization treatments were used on the trees. Figure 1 reports the results for this area.

The SP-4 area is a sand tailings fill project consisting of a total of 200 ha (494 ac.) including approximately 40.5 ha (100 ac.) of wooded wetlands constructed in 1984 and 1985. Species planted in SP-4 in 1985 and 1986 include cypress (Taxodium sp), red maple, sweetgum (Liquidambar styraciflua), river birch (Betula nigra) and blackgum (Nyssa sylvatica). The wetland areas are split into two areas of approximately 20.2 ha (50 ac.) each on the east and west sides of the project. Figure 2 compares the growth rate on SP-4 between Bald cypress (Taxodium distichum) and Pond cypress (Taxodium distichum var. nutons) which are both native to the area.

The SR-8 project is 160 ha (396 ac.) in size and was constructed in 1981 and 1982. This is a land and lakes project with approximately 60 percent land and wetlands and 40 percent lake. The wetlands associated with SR-8 are within the *zone of fluctuation* of the lake. As the lake level fluctuates, the wetlands are at times inundated and at times dry mainly depending on rainfall. The wetland area totals approximately 4.05 ha (10 ac.) and was planted in 1982 with cypress.

The GA project is also a land and lakes project. Approximately 70 percent of the 117 ha (290 ac.) area is wetlands which was constructed from 1985 through 1988. Due to the planting date of 1987 only 8.1 ha (20 ac.) of the area are used for the demonstration area. The wetland area receives sheet flow from the lake and upland area to the west to maintain the wet and dry cycle.

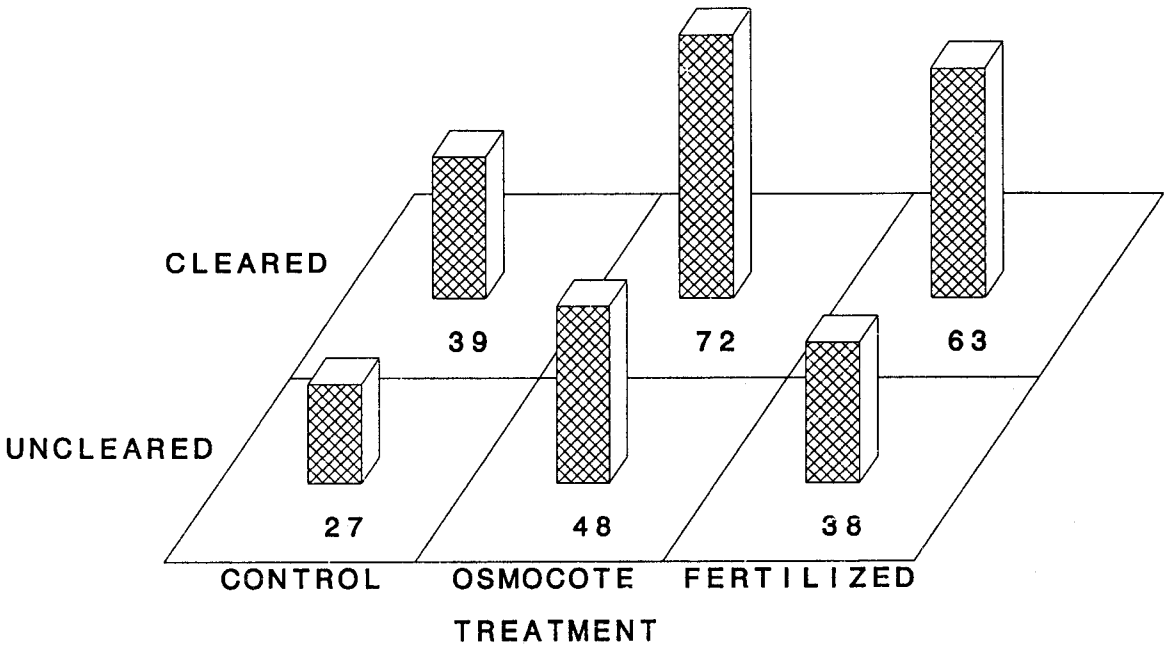



Figure 1. Comparison of tree growth in SA-1 in cleared and uncleared areas - growth in cm OSMOCOTE (50% 14-14-14 50% 18-6-12) Fertilized (22-8-2).

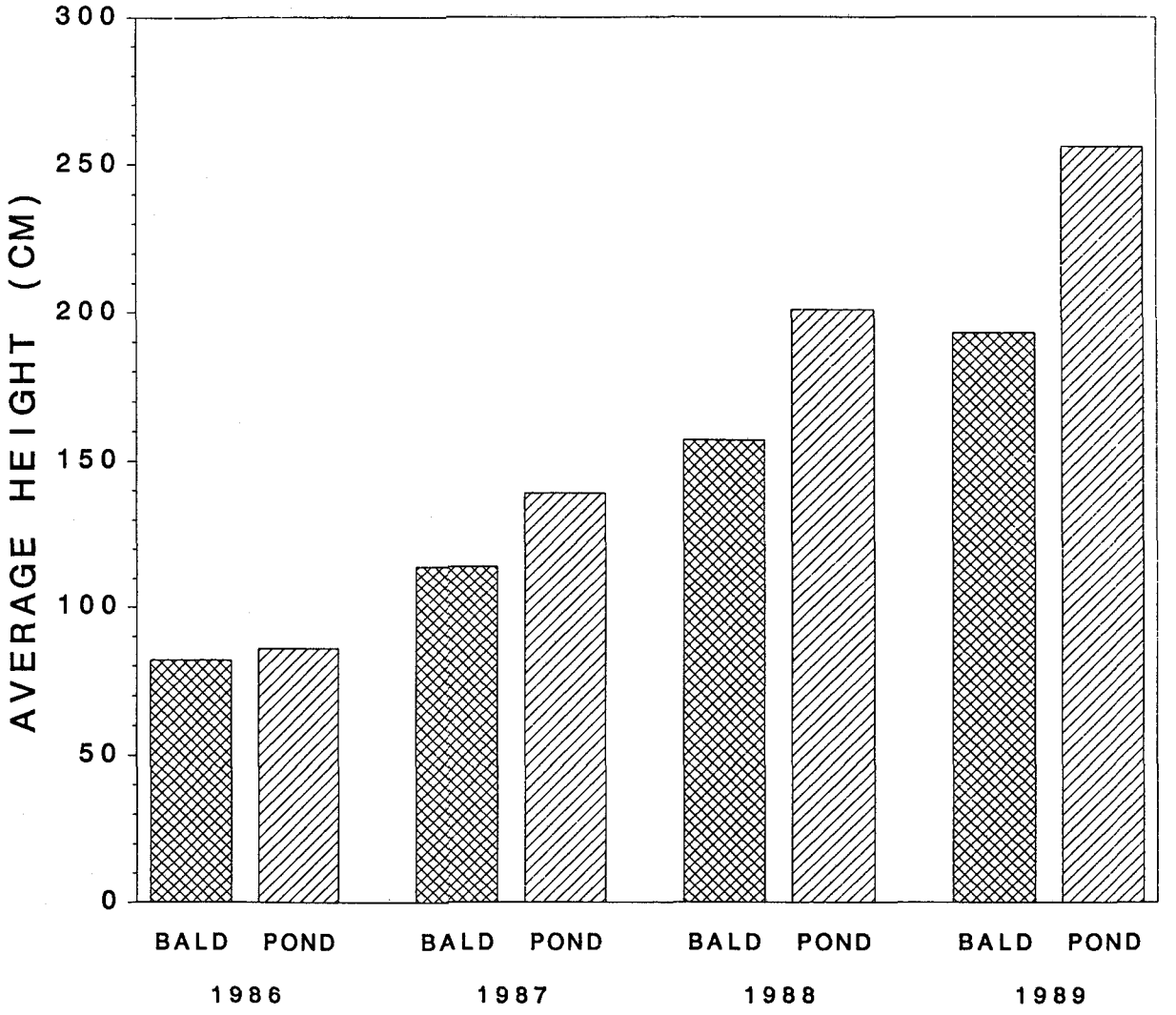



Figure 2. Comparison of growth rates of bald cypress and pond cypress in SP(4).

## Methods

### Tree Monitoring

Belt transects (*elongated line-strip quadrats*) were used to measure tree density, tree diversity and growth rates. The transects were laid out across the wetland areas along gradients from the upland edge through the wettest areas. Each transect was 10 m (32.8 ft.) wide. Sufficient numbers of transects were used to sample at least 5 percent of the demonstration areas. Trees within the belt transect were mapped using a grid system so that monitoring data could be established for a specific tree. The two categories in Table 2 were used to assess tree conditions. In addition, the following data were recorded:

1) tree species
2) tree height
3) water depth, if present.

Each demonstration area was divided into four quarters. Each quarter was surveyed to determine how many trees produced seed balls (*cypress only*). Once a tree was found to produce seed balls, the tree was flagged and labeled with a permanent aluminum tag and its approximate location mapped. Success was based on finding some evidence of cypress seed ball production in each quarter of the demonstration areas and finding the appropriate percentages listed in Table 1 within the entire demonstration area.

Table 2. Tree assessment categories for monitoring of success criteria.

| Category | Description |
|---|---|
| Live | Tree appears to be in a generally good condition. |
| Dead | A decision as to whether a tree is dead is generally made when a tree is in such poor condition that survival is unlikely. |

Ground Cover

Ground cover was measured using a modified line-strip quadrat method. The method consists of observations of plant species occurring along an elongated quadrat extending through the study area. The quadrat was divided into continuous 3.05 m (10 ft.) intervals, each of which was .61 m (2 ft.) wide. The 3.05 m (10 ft.) intervals were further divided into five .61 x .61 m (2 x 2 ft.) intervals. Species cover was determined on the basis of the percent cover occupied within each 3.05 x .61 m (10 x 2 ft.) cover interval. Seven cover categories were assigned to estimate ranges of percent cover that were usually determined (Table 3). In addition, frequency was determined on the basis of occurrence within each .61 m (2 ft.) interval. Therefore, a maximum value of 5 was possible for each 3.05 m (10 ft.) interval. Data were tabulated and summarized by species, as follows:

1. Total Frequency = the total number of .61 x .61 m (2 x 2 ft.) intervals where the species occurred.

2. Relative Frequency = the total number of occurrence intervals in relation to the total number of possible .61 x .61 m (2 x 2 ft.) intervals.

3. Average Occurrence Cover Value = the average cover category value assigned on all 3.05 x .61 m (10 x 2 ft.) intervals were species occurred.

4. Average Occurrence Percent Cover = the percent cover for each species calculated for only where it occurred.

5. Total Area Covered = the total square foot coverage exhibited by the species.

6. Total Percent Cover = the percent of the total transect area that was covered.

Table 3. Cover value categories and assigned ranges (%) for each classification.

| Category | Range (%) | Assigned Cover Values | % Range Category |
|---|---|---|---|
| 1 | <1 | 1 = 1% | > 1 |
| 2 | 1-10 | 2 = 10% | > 10 |
| 3 | 11-30 | 3 = 30% | > 20 |
| 4 | 31-50 | 4 = 50% | > 20 |
| 5 | 51-70 | 5 = 70% | > 20 |
| 6 | 71-90 | 6 = 90% | > 20 |
| 7 | 91-100 | 7 = 100% | > 10 |

Hydrology

Hydrology success criteria applied only to the SA-1 project. A standard water level recorder was placed in a deep pool area near the outfall to measure water depth in the demonstration area. The recorder was surveyed in to allow calculation of the amount and duration of inundation over the demonstration area. Any low areas >.04 ha (0.1 acre) within the demonstration area were connected to the pool in which the recorder is located, a connection was made via a small drainageway. Rainfall in the area was measured by an existing rainfall gauge at the Suwannee River Mine office approximately .8 km (.5 mi.) away.

After half the area was cleared, the demonstration area was resurveyed as necessary to document the existing elevations within the wetland area. This survey information, in conjunction with the water level recorders, was then used to determine inundation percentages and durations.

Water Quality

Water quality was measured in the GA and SA-1 demonstration areas. Samples were taken quarterly near the discharge from each of the areas. The samples consisted of grabs integrated over the water column. The samples were taken in accordance with the procedures outlined in Handbook for Analytical Quality Control in Water and Wastewater Laboratories (EPA 1979) or in Standard Methods for the Examination of Water and Wastewater (APHA et al. 1985).

Occidental's standard chain-of-custody and quality assurance procedures, which have been approved by EPA as part of an ongoing EPA study, were followed. The analyses were performed by either Occidental's environmental laboratory or a commercial laboratory with a DER-approved Quality Assurance Program that conformed to DER Guidelines for Preparing Quality Assurance Plans (DER 1986). The analyses were performed by an approved EPA method cited in 40 CFR Part 136. The sampling was concluded after four successive samples showed compliance with the applicable Section 17-3.121, Florida Administrative Code (FAC), water quality standards.

Faunal Criteria

Macroinvertebrate samples were used to measure faunal success. Quarterly, qualitative samples were collected in the SP-4, SA-1, and GA demonstration areas. The samples consisted of collecting three, at least one-man-hour replicates at each station, using various methods such as dip nets, screens, forceps, pipettes, and ~15 cm (5.9 in.) diameter cores ~5-10 cm (2.0 - 4.0 in.) deep. Specimens from all the sampling methods were composited to form a single sample for each replicate at each station. The samples were handled and in compliance with a FDER approved Quality Assurance Plan (DER) (DER 1986).

Diversity was calculated based on the definition in Section 17-3.021(24), FAC. Macroinvertebrate diversities were used to access the quality of the macroinvertebrate communities in the wetland demonstration areas. They were not used to apply DER's biological integrity standard for class III waters. The generated diversity values were utilized in determining restoration success and were not used as a direct measure of water quality.

The stations were located in permanently inundated areas to the extent practicable. Two stations were located in SA-1, four in SP-4, and six in the GA. The four stations in SP-4 were divided between the two wetland areas in the project.

Sampling began during the first growing season following tree planting. The sampling was discontinued once the diversity criteria outlined in Table 1 were met or exceeded for four consecutive sampling periods.

Soils Criteria

Soils criteria were based on the percentage of organic matter found in the top 5 cm (2.0 in.) (Table 1). At least 20 samples were collected in each demonstration area. The samples were composited or analyzed

individually at Occidental's option. If they were analyzed individually, the success criteria were compared to the average of the samples from each demonstration area.

The samples were taken by brushing away the litter accumulation on the surface and then taking the cores. The samples were handled in accordance with standard procedures and were subject to the same chain-of-custody procedures as the water quality samples. The samples were analyzed using the Walkley-Black method as described by Jackson (1958). Samples were taken annually, beginning at the end of the first growing season following tree planting. Sampling ceased once the criteria were met.

**Results**:

The following data represents the first milestone period which ended on November 30, 1989. Data are presented for those parameters which apply to each demonstration plot. (See Table 4.)

All criteria for Milestone I for the four demonstration areas were met with the exception of tree diversity in SA-1. (See Table 1.) Tree diversity numbers were not within the ranges specified but considering the stand density the removal of specific species would bring the diversity within the ranges without vioaltion of the tree density requirements. Survival rates on some of the hardwoods were greater than anticipated at planting time, causing the resultant diversity to be outside the range.

## Conclusions

These efforts to establish success criteria for forested wetlands in Florida have proven beneficial for the environmental agencies, environmental action groups and the mining company. Each party received information about the concerns of the other and yet were provided a forum to express their opinions and thoughts on the subject of forested wetland reclamation. It became a group effort working together to accomplish a common goal.

Table 4. Results of Success Criteria Monitoring - as of November 30, 1989.

**SP(4)**

| Parameter | Milestone I | Value |
|---|---|---|
| Density (trees/acres) | 200 | 425 |
| Growth Rate (% increase) | 100 | 346 |
| Ground Cover (area/extent) | 70 | 75 |
| Wildlife Abundance | 2.00 | 4.15 |
| Wildlife Richness | 2.00 | 4.15 |
| Soils (% organic matter in top 5 cm) | .5 | 1.64 |

**Green Area (GA)**

| Parameter | Milestone I | Value |
|---|---|---|
| Density (trees/acres) | 300 | 594 |
| Diversity | | |
| % Taxodium | 75-80 | 77 |
| % Other | 20-25 | 23 |
| Growth Rate (% increase) | 50 | 161 |
| Ground Cover (area/extent) | 70 | 77 |
| Water Quality | Chapter 17-3 Standards | Met All Applicable Standards |
| Wildlife Abundance | 2.00 | 3.88 |
| Wildlife Richness | 2.00 | 3.88 |
| Soils (% organic matter in top 5 cm) | 0.5 | 0.57 |

**SR(8)**

| Parameter | Milestone I | Value |
|---|---|---|
| Density (trees/acres) | 200 | 331 |
| Growth Rate (% increase) | 200 | 599 |
| Seed Production (% of Taxodium producing seed) | 5 | 22 |
| Soils (% origin matter in top 5 cm) | 0.5 | 1.59 |

**Settling Area 1(SA1)**

| Parameter | Milestone I | Value |
|---|---|---|
| Density (trees/acres) | 500 | 671 |
| Diversity | | |
| % Taxodium | 50-60 | 58 |
| % Nyssa | 20-25 | 25 |
| % Other | 20-25 | 17 |
| Growth Rate (% increase) | 50 | 114 |
| Ground Cover (area/extent) | 70 | 72 |
| Hydrology | 90 Days Inundate/Yr 30 Days Consecutive 30 Days 50% Not Inundated | Achieved |
| Water Quality Standards | Chapter 17-3 Applicable Standards | Met All |
| Wildlife Abundance | 2.00 | 3.40 |
| Wildlife Richness | 2.00 | 3.40 |
| Soils (% organic matter in top 5 cm) | 0.5 | 5.21 |

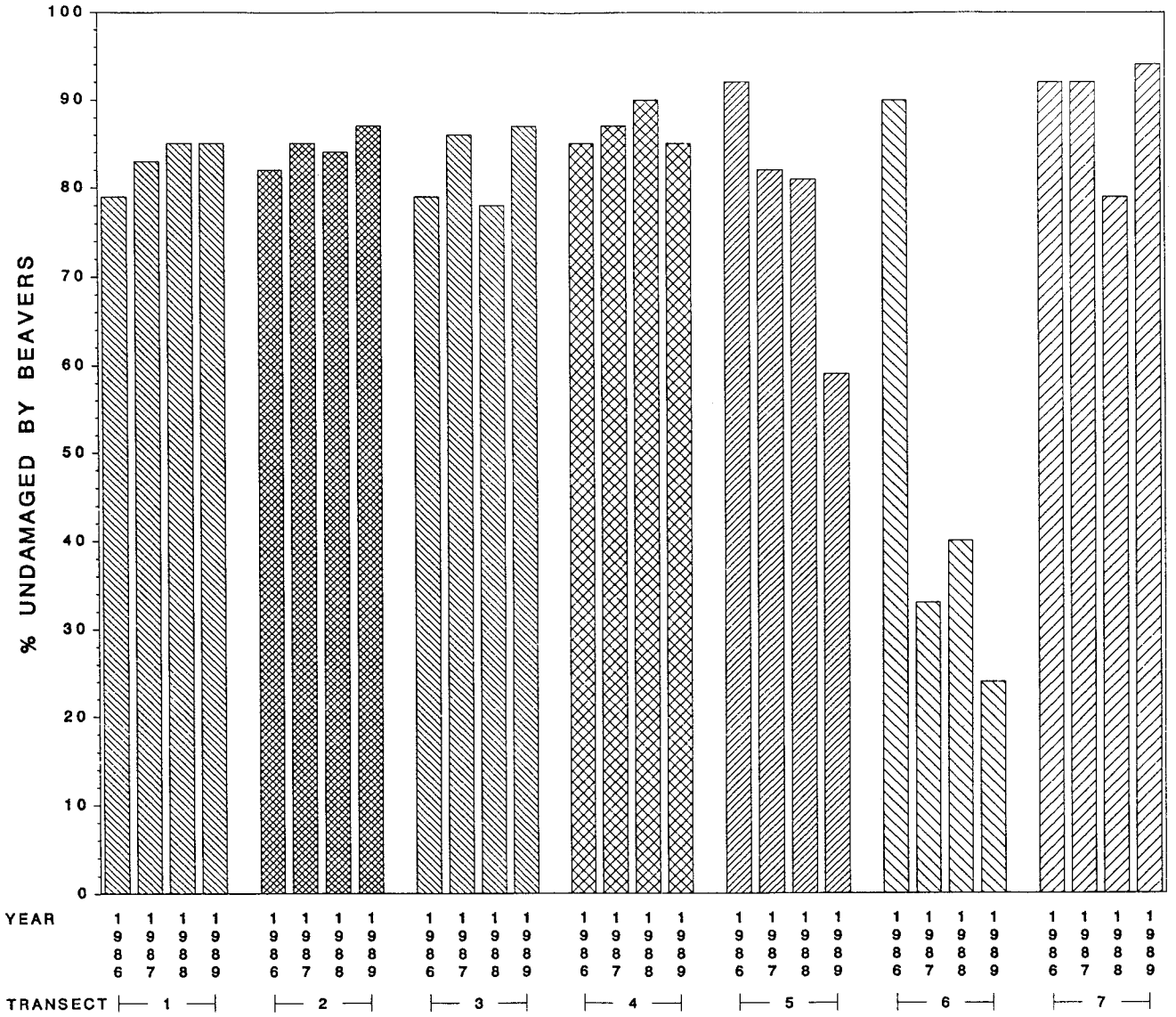



Figure 3. Effect of beaver damage on cypress in SP(4).

As a result of the monitoring performed, several things became noticeable that had not been considered at the time the success criteria were developed. Two of these are worth mentioning here, knowing that they may not be the only unforeseen things to occur in this type of activity. They are tree survival rates and animal damage.

Tree survival rates are less predictable for some species, especially hardwoods. A goal for a specific range of densities may not be met because of better than expected survival rates. Increased planting density will allow for adequate numbers of specific species if manipulation of the densities is really desired.

Animal damage both from rabbits and large rodents, including beavers, has also been an unforeseen problem. While the presence of wildlife species provides good evidence of the success of the reclamation, they also may dramatically affect the monitoring results, such as significant effect on growth rate determinations based on increase in height. Beavers have caused a major impact on one transect. While the presence of the beavers are positive indications of suitable habitat we experienced significant losses of cypress trees as a result of their activities. (See Figure 3.)

Future milestone reporting will occur in 1991 and 1993. Indications at this time are that the criteria will be met or exceeded. Results to date would verify that the criteria selected were measurable, reasonable and good indicators of wetland success.

## Literature Cited

American Public Health Association, American Water Works Association, and Water Pollution Control Federation. 1985. Standard methods for the examination of water and wastewater. American Public Health Association, Washington, DC. 1268 p.

Florida Department of Environmental Regulation. 1986. DER guidelines for preparing quality assurance plans - DER-QA-001/85. DER, Tallahassee, Florida.

Jackson, M.L. 1958. Soil chemical analysis. Prentice Hall, Inc., Englewood Cliffs, NJ. 219-220 p.

U.S. Environmental Protection Agency. 1979. Handbook for analytical quality control in water and wastewater laboratories - EPA-600/4-79-019. EPA, Cincinnati, OH.

# OBSERVATIONS ON THE GEOLOGIC HISTORY OF A WETLAND HAVE THE POTENTIAL OF ASSISTING EFFORTS TO CONSTRUCT WETLANDS ON MINED LAND

by

Robert E. Behling

Abstract. Disruption of surface water flow through a wetland along Pendleton Creek, Tucker County, WV due to the presence of an abandoned underground mine, led to a decision by the WV Department of Natural Resources to permit removal of a portion of the wetland to "daylight" the coal yet remaining, followed by reestablishment of a wetland. The wetland was examined as it was removed during mining and the data collected has resulted in a better understanding of the creation of this wetland over the past 30,000 years. It has also allowed us to observe both the "plumbing" of a host site for fixing iron as coatings on the fine-grained sediments that line the valley floor. It is anticipated that a better understanding of the natural system as it has operated during the past 30,000 years will aid in the effort to construct a wetland on this mined land.

Additional key words: "Plumbing" of a natural wetland, fixing iron.

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]Robert E. Behling is a Professor of Geology, Department of Geology and Geography, West Virginia University, Morgantown, WV 26506.

This page intentionally left blank.

Impacts of Volunteer Cattail Wetlands on Drainage Quality from Reclaimed Mined Land in Northern West Virginia. Edward Jamison, Hydrogeologist, GHR Hausler Company, Horshan, Pennsylvania, and Henry W. Rauch, Professor of Geology, West Virginia University, Morgantown, West Virginia.

A study was conducted of selected volunteer cattail wetlands located near reclaimed surface coal mines in northern West Virginia to test the effects of the wetlands on mine drainage throughout 1986 for both water entering and exiting each wetland. Eight wetlands were investigated in Monongalia County and five wetlands were studied in Preston County. The Monongalia County mine sites exhibited mildly adverse water, with mean values of 6.92, 4.91 mg/L, and 4.37 mg/L for pH, iron, and manganese respectively, reflecting drainage from Waynesburg coal mines. In contrast, the Preston County mine sites produced water with notably worse quality, with mean values of 4.87, 12.30 mg/L, and 10.04 mg/L for pH, iron, and manganese respectively, representing drainage from Upper or Lower Freeport coal mines. The Monongalia County wetlands on average reduced total iron concentration from mine drainage by 32.8 percent, decreased total manganese concentration by 11.4 percent, and increased the pH by 5.2 percent. Thus these wetlands improved mine drainage water quality significantly. The Preston County wetlands on average reduced total iron concentration by 65.7 percent, reduced total manganese concentration by just 2.6 percent, and decreased pH by 2.9 percent. Some wetlands had additional mine drainage seepage originating within them, which served to complicate the study of wetland effectiveness in mitigating mine drainage quality. In summary, these wetlands were significantly effective in reducing iron concentration, but not in reducing manganese or sulfate concentration or in raising pH.

ADDITIONAL KEY WORDS: Preston County, Monongalia County; iron, manganese, pH, sulfate.

This page intentionally left blank.

**Simulated Typha Wetlands Applied to Removal of Iron and Manganese from Acid Mine Drainage.** J.P. Calabrese, D.K. Bhumbla, A.J. Sexstone, G.K. Bissonnette, and J.C. Sencindiver. *Division of Plant and Soil Sciences, West Virginia University, Morgantown, WV. 26506*

Biological nuetralization of acid mine drainage (AMD) in natural and man-made cattail (Typha) wetlands offers an alternative to chemical treatments. Mechanisms within wetlands which remove iron and manganese from AMD are not completely understood. Such information is vital for predicting impacts of long term use of Typha wetlands for treatment of AMD. In this study, simulated Typha wetlands were constructed in a controlled greenhouse environment. Four wooden troughs measuring 480 x 60 x 60 cm (L x W x D) were lined with 40 mil thick plastic liners. Substrate consisted of a bottom layer of limestone (15 cm) covered by a peat moss/soil mixture (45 cm). Two ecotypes of *Typha latifolia*, designated A and B, were collected from non-AMD impacted field sites. Troughs contained 32 plants arranged in 16 rows of alternating ecotypes. Within 60 days, cattial root systems were fully established both on surface and subsurface and plants averaged 200 to 250 cm in hieght. Control systems were constructed in which no *T. latifolia* was planted. All wetlands received inoculation with a mixed sediment slurry collected from a natural Typha wetland currently treating AMD in order to introduce both aerobic and anaerobic microorganisms. Preliminary addition of tap water to all troughs (300 ml/min) allowed establishment of microflora prior to addition of AMD. Interstitial water samplers were installed prior to AMD addition at the input and output points of the simulated wetland and allowed to equilibrate for 14 days. Pore water collected from the samplers show a pronounced decrease in Eh and increase in pH with depth, indicating oxygen consumption by microorganisms and buffering by the added limestone, respectively. Sulfate concentrations decreased with depth, probably due to establishment of active sulfate-reducing bacterial populations. Chemical analyses were performed on all troughs containing *T. latifolia* to obtain base line data on Fe, Mn, and $SO_4^{-2}$ levels at various depths. Total Fe levels showed an overall increase with increasing depth and consisted primarily of ferric ($Fe^{+3}$) iron. Similar patterns were observed with Mn levels. Data indicate that the greenhouse systems are sufficient to study microbial biotransformations of Fe, Mn, and S. Several isolation procedures are currently being evaluated for their ability to recover Fe and Mn-oxidizing and -reducing, as well as S-oxidizing and $SO_4^{-2}$ - reducing bacteria. Data will be presented on bacterial populations and activities before and after AMD additions to simulated Typha wetlands.

This page intentionally left blank.

# BIOCHEMICAL TREATMENT OF MINE DRAINAGE THROUGH A REEDGRASS WETLAND

by

Jack R. Nawrot and William B. Klimstra[1]

**Abstract.** Treatment of pre-law acid mine drainage with lime neutralization facilities is an expensive and short-term solution for a chronic problem. Processes associated with microbiological sulfate reduction have long been recognized as a potential technique for acid water treatment. The value of biochemical treatment of mine drainage was demonstrated through the management and monitoring of a wetland treatment unit established at the Peabody Will Scarlet Mine in southeastern Williamson County, Illinois. More than 60 million gallons of acid water was directed through an existing 27-acre non-acid wetland during April 1988 through February 1989, to evaluate biochemical treatment capabilities of the receiving wetland. Monitoring of surface and groundwater quality documented chemical, and hydrological characteristics within the oxidized surface zone, and the wetland anaerobic substrate of a reedgrass (*Phragmites australis*) wetland subjected to chronic acid input.

## Introduction

Mine associated wetlands in the midwest have long been recognized as valuable assets rather than liabilities (Klimstra and Nawrot 1982). Water quality of the majority of mine impoundments in Illinois is adequate to support many industrial, agricultural, recreational, and biological uses (Klimstra and Nawrot 1986). However, in some regions pre-law mining practices have contributed to phytotoxic spoil areas and acid runoff. Although mine drainage treatment plants are effective in addressing the problems of acid generation, treatment of symptoms can become a perpetual and expensive commitment (Nawrot et al. 1988). Abatement of acid sources rather than perpetual treatment is the only long term solution; however, in areas of extensive acid generation, complete abatement is not always possible. Chronic acid seeps may persist and water quality of inadequately buffered impoundments may remain acidic.

Many pre-law acid impoundments have recovered through natural processes. Recovery of poorly buffered acid mine impoundments is contingent upon development of physical and chemical conditions conducive to sulfate reduction (King et al. 1974). The capacity of an acid impoundment to attain neutral conditions depends upon organic matter input from the surrounding watershed as vegetation density increases; thereby, providing an organic substrate for sulfate reducing bacteria (Brugam et al. 1983). Microbial sulfate reduction has long been recognized as a potential treatment for mine drainage; it was suggested that with addition of organic matter through sewage sludge, waste paper, algae, aquatic weeds, etc., conditions for sulfate reduction could be greatly enhanced (Tuttle et al. 1969a). Hedin (1988) suggested sulfate reduction mechanisms are an important design consideration for current AMD treatment wetlands.

Introduction of organic matter can enhance the sulfate reduction process in AMD treatment. Organic matter decomposition promotes the establishment of anaerobic conditions conducive to heterotrophic bacteria such as *Desulfovibrio* which convert sulfuric acid to hydrogen sulfide (Rice and Rabolini 1972). Organic carbon associated with decomposing wetland plants or sewage sludge provides an electron source (energy) for sulfate

[1]J.R. Nawrot is an Associate Scientist for the Cooperative Wildlife Research Laboratory at Southern Illinois University, Carbondale, IL 62901; W.B. Klimstra is a Researcher for the Cooperative Wildlife Research Laboratory at Southern Illinois University, Carbondale, IL 62901.

reduction. Sulfate reduction is most effective when a pH of $\geq$5.5 is maintained and temperatures are above 10°C (a reaction rate increase of 4x occurs at 37°C) (Rice and Rabolini 1972). However, sulfate reduction has been recorded at a pH of 3.0 (Tuttle et al. 1969b). As hydrogen sulfide forms, acidity decreases, and pH increases as the hydrogen sulfide gas is lost to the atmosphere (King et al. 1974). For every equivalent of sulfate reduced and precipitated (as iron sulfide), one equivalent of alkalinity should be produced (Brugham et al. 1983). As sulfate reduction proceeds, ferrous and other metallic ions form metallic sulfides. Although large amounts of hydrogen sulfide are produced in the anaerobic zone of submerged wetland soils, the concentration of water soluble hydrogen sulfide should remain negligible due to removal of sulfide in insoluble forms (Ponnamperuma 1972). The role of sulfate reduction in AMD amelioration, is a beneficial geochemical process that has been well documented in the rice growing regions of Asia where seasonal and planned inundation of acid sulfate soils promotes sulfate reduction and alkalinity generation (Moorman and van Breeman 1978).

Algae blooms resulting from nutrient additions such as sewage sludge can also promote bicarbonate alkalinity and anaerobic substrate conditions (Stowell et al. 1981). As pH increases, metallic ions precipitate, forming black deposits of FeS. Additional metallic ions can be removed by absorption (physical-chemical, and bio-floculation) to extracellular polymers produced by sulfate reducing bacteria (slime layers) (Dugan 1970).

Applicability of microbiological processes for acid water treatment through sulfate reduction has been recognized for almost 20 years; however, only recently have these principles been applied in the eastern coal fields. Initial laboratory studies and plot demonstrations have emphasized the wetland treatment capability of Sphagnum bogs in the eastern United States (Weider and Lange 1982, Kleinmann et al. 1983). Currently, wetland treatment systems emphasize cattails (Typha) established in organic substrates using a compartmentalized cell design (Hedin 1989).

Constructed wetlands have provided valuable data regarding specific hydrologic physical, chemical, and vegetation relations of man-made systems receiving mine drainage (Brodie et al. 1988). However, performance of all constructed wetlands has not been consistent. Sizing and loading factors are variables that are currently being investigated by many state and federal organizations.

To determine the acid loading capacity and treatment capability of a pre-law wetland, a field demonstration was conducted at the Peabody Will Scarlet Mine in southeastern Illinois. This paper briefly summarizes flow, acid loading data, and receiving water quality data which were collected during low volume and high volume discharge trials. These data were derived from a portion of a wetland treatment study supported by the Illinois Mines and Minerals Resource Institute (USBM Grant G118-4117). Data used in this paper were part of a field and laboratory study conducted by Mark Guetersloh, Research Assistant.

## Methods

To demonstrate the potential value of wetlands as treatment units for acid mine drainage, monitoring and documentation of water quality and substrate characteristics were conducted on two wetlands located at the Peabody Will Scarlet Mine "Old Works" area, southeastern Williamson County, Illinois. The treatment unit (IC1) is a 27-acre reedgrass wetland with a large length to width ratio (1,750 feet long, 600 feet wide) (Figure 1). A rectangular notch weir was installed between IC1 and an adjacent 18-acre acid (pH 3.3 to 3.8) impoundment (IC2) to regulate the introduction of acid water into the wetland.

Monitoring of IC1, IC2 was conducted at 21 sampling stations (1a, 1b in IC2, and 2-20 in IC1) (Figure 1). Surface water sampling was conducted monthly (April 1988-February 1989) at each sampling station (1a/1b-20). Surface water analyses included pH, eH (redox potential), dissolved oxygen, and conductivity. Interstitial water sampling was conducted monthly at sample stations 1a, 1b, 3, 9, 15, and 20; analyses included pH, eH (redox potential), dissolved oxygen, and conductivity.

Six groundwater wells, constructed of 2 inch Brainard-Kilman Triloc slotted (.010 inch slot size) PVC monitor pipe were installed in IC1 and IC2 (sample stations 1a, 1b, 3, 9, 15, and 20). Slotted wellpipe extended from the sediment surface to a depth of 30 inches with brown pea-gravel installed around the slotted section to prevent excessive sedimentation. Dissolved oxygen, pH, eH and conductivity within wells were determined in-field with portable meters (probes lowered to a depth of 4 inches below sediment-water interface within each well). Water samples were evacuated from each well (from a depth of 4 inches below sediment water interface) with a hand operated peristaltic pump. Acidity of surface and interstitial water was determined by hydrogen peroxide boiling procedure (ASTM D1067-70, Method B), and alkalinity by electrometric sodium hydroxide procedure (ASTM D1067-70, Method A) (American Society for Tests and Materials 1979).

During April 1988 through February 1989 more than 60 million gallons of acid water (pH 3.3) were introduced into IC1 from IC2 through a rectangular notch weir. Flow measurement was determined by standard methods (ASTM D2305, American Society for Testing and Materials 1979).

## Results and Discussion

Managing and monitoring of discharge events were conducted to identify the biochemical threshold of a naturally established reedgrass wetland receiving chronic acid mine drainage. The goal of this field demonstration was simply to identify the operational limits (i.e. maximum flow and/or allowable duration of acid input) of an established ($\sim$ 15 years old) wetland that had previously served ($\sim$ 1984-1987) as a natural "treatment" system for polishing chronic acid runoff prior to off-site discharge. Observation and monitoring by mine personnel of outflow quality below the reedgrass wetland during the mid-1980's

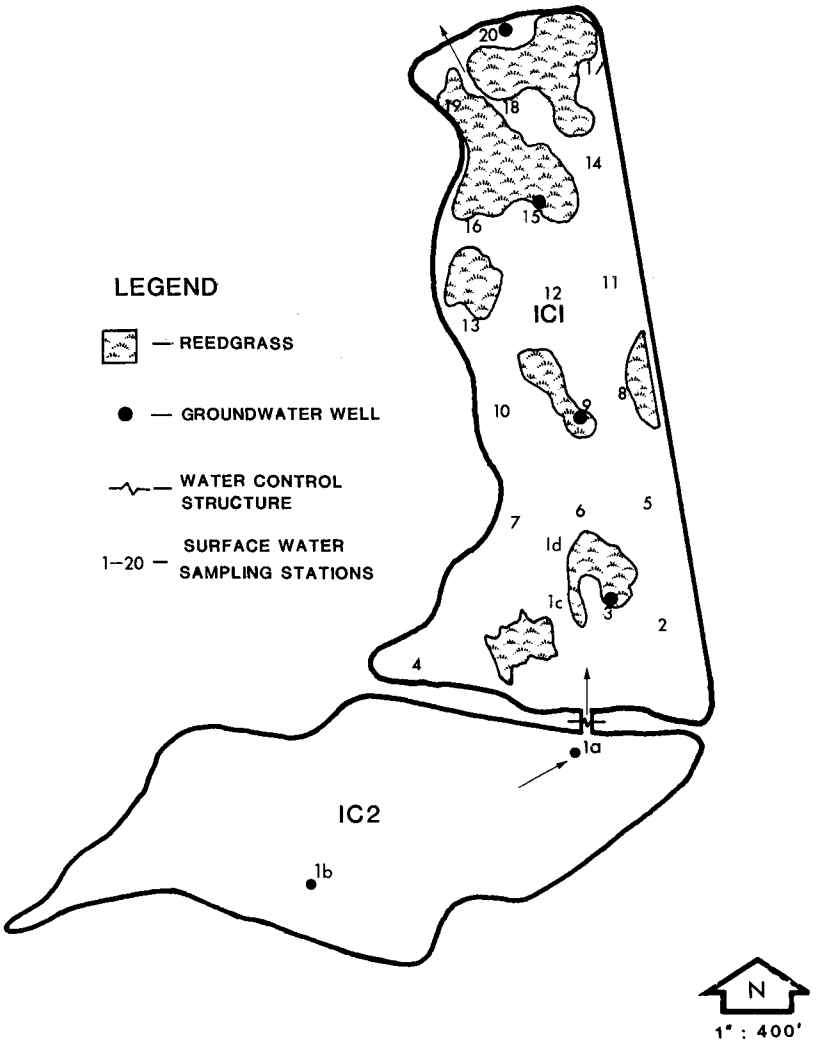



Figure 1. Location of groundwater wells and sampling sites in wetland areas IC1 and IC2 at the Peabody Will Scarlet Mine, Williamson County, Illinois.

established the ability of the reedgrass wetland to function as a supplemental treatment system if operational limits could be better defined.

However, prior to initiation of pre-law acid soils reclamation in the upslope watershed in 1983, water quality within the reedgrass wetland was unacceptable for discharge without chemical (i.e. hydrated lime) treatment. Water quality within IC1 was generally acidic (pH $\leq$ 3.2; acidity @ 480 to 1,300ppm $CaCO_3$). Following the initiation in 1983, of reclamation of approximately 175 acres of upslope acid slurry disposal areas, and alkaline flushing of the reedgrass wetland (IC1), water quality within IC1 returned to acceptable levels. By fall 1987, surface and interstitial water within IC1 had returned to an acceptable equilibrium characterized by excess alkalinity within both the oxidized surface zone and the anaerobic interstitial zone (Table 1).

Following three more months (February through April 1988) of baseline monitoring, acid discharges (from IC2) into the reedgrass wetland (IC1) were initiated. A series of 11 discharge events were monitored between 12 April 1988 through 9 February 1989 (Table 2). Discharge events included high flow (>13,000 gpm) and short duration (5 hours) acid discharges, as well as moderate to high flow (500 to 2,500 gpm) in association with moderate (1 day) to prolonged durations (greater than 30 days). Monitoring of surface sample locations documented pre- and post-discharge water quality changes during the 11 treatment events (Table 2).

More than 60 million gallons of acid water (pH @ 3.3, 262ppm acidity) were discharged into the reedgrass wetland during the 10-month period of acid loading. Volume of water discharged ranged from slightly more than 1 million gallons during a 20-hour period in August 1988 to more than 30 million gallons during a 1-month period in late winter 1989. These various loadings reflected an attempt to determine the biochemical threshold of the receiving wetlands' natural buffering system. At some point simple dilution by the chemical buffering capacity of the receiving wetland should have been exceeded as based upon physical factors of flow rates, and retention time.

Determination of chemical buffering capacity (based on total volume and average alkalinity) of the receiving wetland indicated the ability of IC1 to dilute and 'treat' approximately 22 million gallons of acid water (average acidity @ 70ppm $CaCO_3$) assuming sufficient retention time to allow complete mixing. As input quality of IC2 acid discharge averaged 250ppm acidity, 5 to 6 million gallons would be the maximum acid loading before exhaustion of alkalinity would be expected. Depletion of alkalinity within the receiving wetland should have occurred during the December 1988 or January 1989 discharge when more than 10 million gallons of acid water were introduced in less than a 20-day period. During this mid-winter discharge, acid loading exceeded the "dilution" ability of the receiving wetland by more than 10 million gallons. During this high loading period (500,000 gallons/day) of 17 December to 6 January 1989, significant depletion of alkalinity was noted in the inlet monitoring station (Station #3) compared to the outlet station (Station #20) within the receiving wetland (Table 2).

Following temporary cessation of acid input to allow sampling of surface and interstitial zones, a final discharge event introduced approximately 1 million gallons of acid water per day for 34 days. Acid discharge was terminated on 9 February when alkalinity had been depleted in both the inlet and outlet stations of the receiving wetland (Table 2). During this discharge event dilution capabilities were exceeded by more than 65 million gallons. As recovery of surface alkalinity values were documented after this and all previous acid discharge events (Figure 2) it can be assumed that biochemical amelioration contributed to alkalinity restoration.

Maintenance and restoration of surface water alkalinity in the 'treatment' wetland (IC1) can be seen in the graphic summarizations of monthly water quality monitoring data (Figure 2, 3, and 4). Water pH values for the acid source (IC2) during October 1987 through July 1989, illustrated the ability of the reedgrass wetland to maintain acceptable water quality despite acid input totalling more than 60 millions gallons (Figure 3). Alkaline buffering capacity of the reedgrass wetland throughout the acid loading trials was severely stressed only during the final high volume input event of December 1988 through January 1989 (Figure 2). Only during this extreme attempt at "overloading" (and the April 1988 7.4 million gallon loading) was alkalinity significantly reduced within the anaerobic interstitial zones. Subsequent sampling showed a recovery of the wetland treatment surface water to pre-discharge alkalinity levels of 70ppm (Table 3, and Figure 4). As suggested by other wetland research (Hedin 1988), restoration of surface water alkalinity could be occuring due to interaction with alkaline ($\sim$ 300ppm @ $CaCO_3$) water in the anaerobic sediment zone. Restoration of alkalinity within previously stressed (i.e. depleted) anaerobic wetland sediment zones (of IC1) following acid discharge events, can be attributed to naturally occuring biochemical processes such as sulfate reduction.

This monitoring project emphasized documentation of effects associated with a large-scale field demonstration of acid loading. Monitoring of trial discharge events documented the response of a natural wetland that had been subjected to similar accidental acid loadings for the previous 5 to 7 years. Many hydrologic variables, such as mixing, flow paths, retention time, etc. are acknowledged as having an influence on wetland treatment performance of the receiving wetland. However as the acid loading factor greatly exceeded the dilution capability of the treatment wetland, this demonstration empirically demonstrated wetland biochemical treatment as a potentially effective method for continued amelioration of chronic acid drainage at this site.

The naturally established (>15 year-old) mined-land wetland was able to process chronic

Table 1. Pre-treatment water quality[1] for surface and interstitial zones of a 27-acre naturally established reedgrass mine drainage treatment wetland (IC1) at the Peabody Will Scarlet Mine, Williamson County, Illinois.

| Sample Location | Depth | pH | Acidity (ppm $CaCO_3$) | Alkalinity (ppm $CaCO_3$) | Conductivity (μmhos/cm) | Dissolved Oxygen (ppm) | Redox. Potential (MV) |
|---|---|---|---|---|---|---|---|
| 3 | Surface | 7.4 | 0.0 | 72.0 | 3,700 | 2.7 | -- |
| | Interstitial | 6.6 | -- | 340 | 2,600 | 0.0 | -175 |
| 9 | Surface | 6.9 | 0.0 | 70.5 | 2,100 | 7.9 | -- |
| | Interstitial | 6.6 | -- | 228 | 3,100 | 0.0 | -52 |
| 15 | Surface | 8.0 | 0.0 | 96.0 | 2,200 | 9.2 | -- |
| | Interstitial | 6.2 | -- | 204 | 3,950 | 0.0 | -52 |
| 20 | Surface | 7.6 | 0.0 | 76.0 | 2,200 | 8.4 | -- |
| | Interstitial | 6.2 | -- | 116 | 2,200 | 0.0 | -5 |
| MEAN | Surface | 7.5 | 0.0 | 71.9 | 2,550 | 7.1 | -- |
| | Interstitial | 6.4 | -- | 220 | 2,962 | 0.0 | -71 |

1. Surface water quality samples collected 7 October 1987. Interstitial water quality samples collected 12 January 1988.

Table 2. Water quality summary data[1] for discharge of acid water (IC2) into non-acid wetland (IC1) at the Peabody Will Scarlet Old Works area, for the period 12 April 1988 through 9 February 1989.

| Date(s) | Duration (hours) | Flow | | Acid Wetland (IC2) (Station 1a) | | Non-Acid Wetland (IC1) Inlet-Station 3 (after discharge) | | Non-Acid Wetland (IC1) Outlet-Station 20 (after discharge) | |
|---|---|---|---|---|---|---|---|---|---|
| | | Rate (GPM) | Total (Gal x $10^6$) | pH[2] | Acidity[2] (ppm $CaCO_3$) | pH | Alkalinity (ppm $CaCO_3$) | pH | Alkalinity (ppm $CaCO_3$) |
| 12 April 1988 | 5 | 13,124 | 3.9 | 3.5 | 268 | 7.0 | 64 | 7.2 | 72 |
| 21-22 April | 25 | 2,747 | 4.1 | 3.5 | 260 | 6.8 | 68 | 6.8 | 90 |
| 28-29 April | 20 | 2,747 | 3.3 | 3.6 | 256 | 6.8 | 52 | 6.2 | 132 |
| 26-27 Aug. | 25 | 1,087 | 1.6 | 3.3 | 300 | 7.3 | 80 | 7.3 | 210 |
| 29-30 Aug. | 20 | 898 | 1.1 | 3.2 | 272 | 7.0 | 76 | 7.2 | 208 |
| 5-7 Sept. | 42 | 700 | 1.8 | 3.2 | 232 | 7.4 | 108 | 7.0 | 120 |
| 7-9 Sept. | 50 | 557 | 1.7 | 3.2 | 244 | 7.0 | 100 | 7.0 | 212 |
| 25-28 Sept. | 72 | 492 | 2.1 | 3.1 | 264 | 7.2 | 68 | 7.2 | 160 |
| 25 Oct.-5 Nov. | 192 | 180 | 2.0 | 3.2 | 272 | 6.6 | 48 | 7.2 | 152 |
| 17 Dec.-6 Jan. 1989 | 458 | 377 | 11.0 | 3.5 | 248 | 5.6 | 16 | 6.6 | 56 |
| 6 Jan.-9 Feb. | 815 | 607 | 30.0 | 3.6 | 256 | 5.8 | 12 | 5.5 | 16 |

[1]Summary: 62.6 million gallons (documented; actual >90 million gallons) @ pH ~3.3 and 262 (ppm $CaCO_3$) acidity.
[2]Average value based on initial and final water quality.

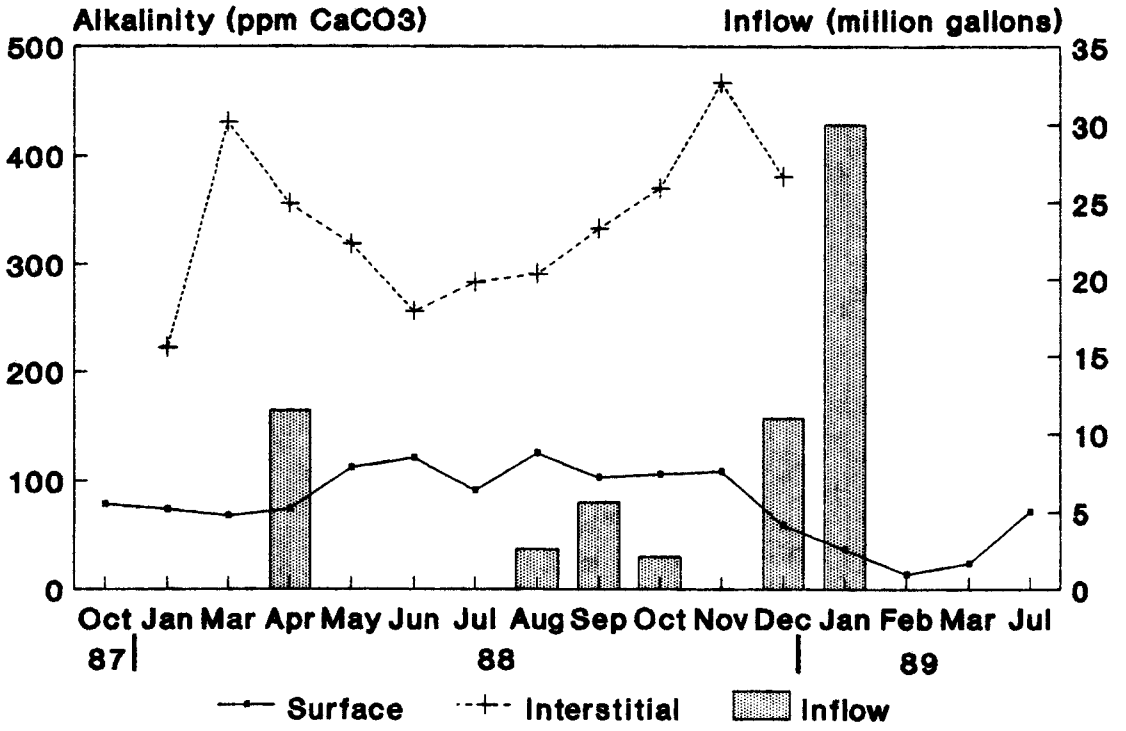



Figure 2. Acid discharge loading volumes and alkalinity response of surface and interstitial water in a 27-acre naturally established reedgrass mine drainage treatment wetland at the Peabody Will Scarlet Mine, Williamson County, Illinois.

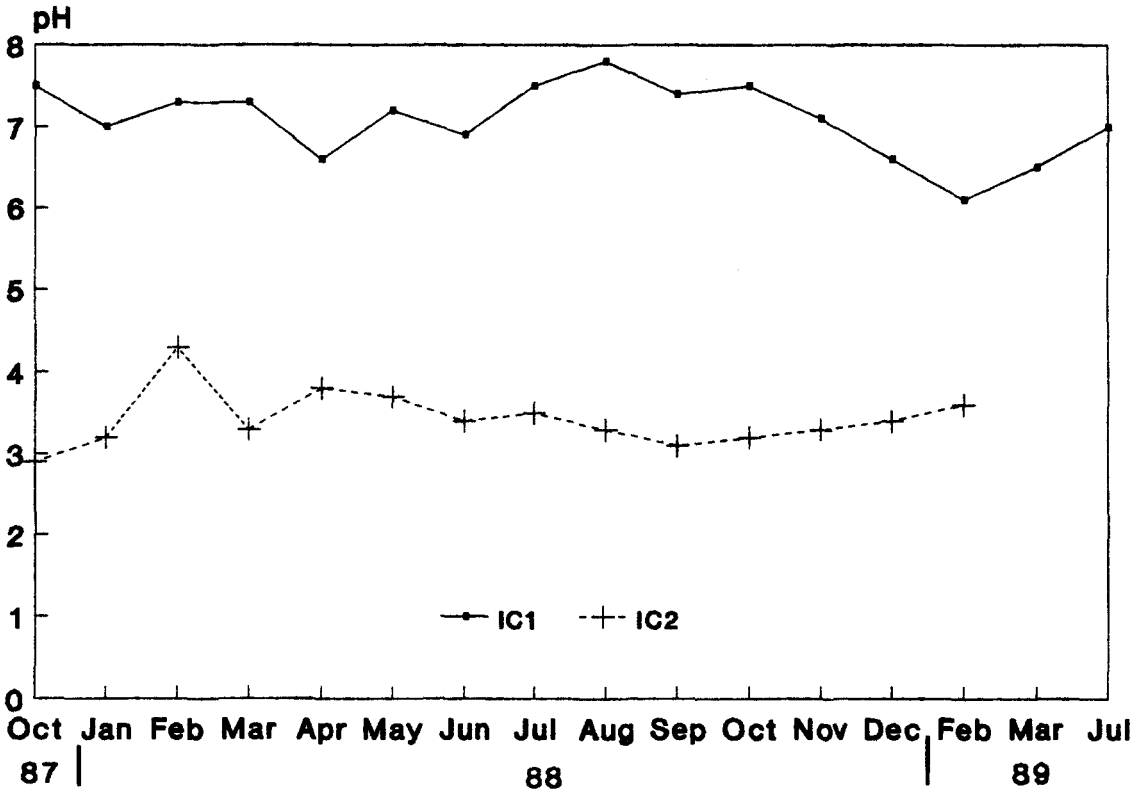



Figure 3. Surface water pH values for a 27-acre naturally established reedgrass wetland (IC1) receiving input from a pre-law acid wetland (IC2). Acid discharge (pH 3.3, acidity @ 262 ppm) exceeded 60 million gallons during April 1988 through February 1989.

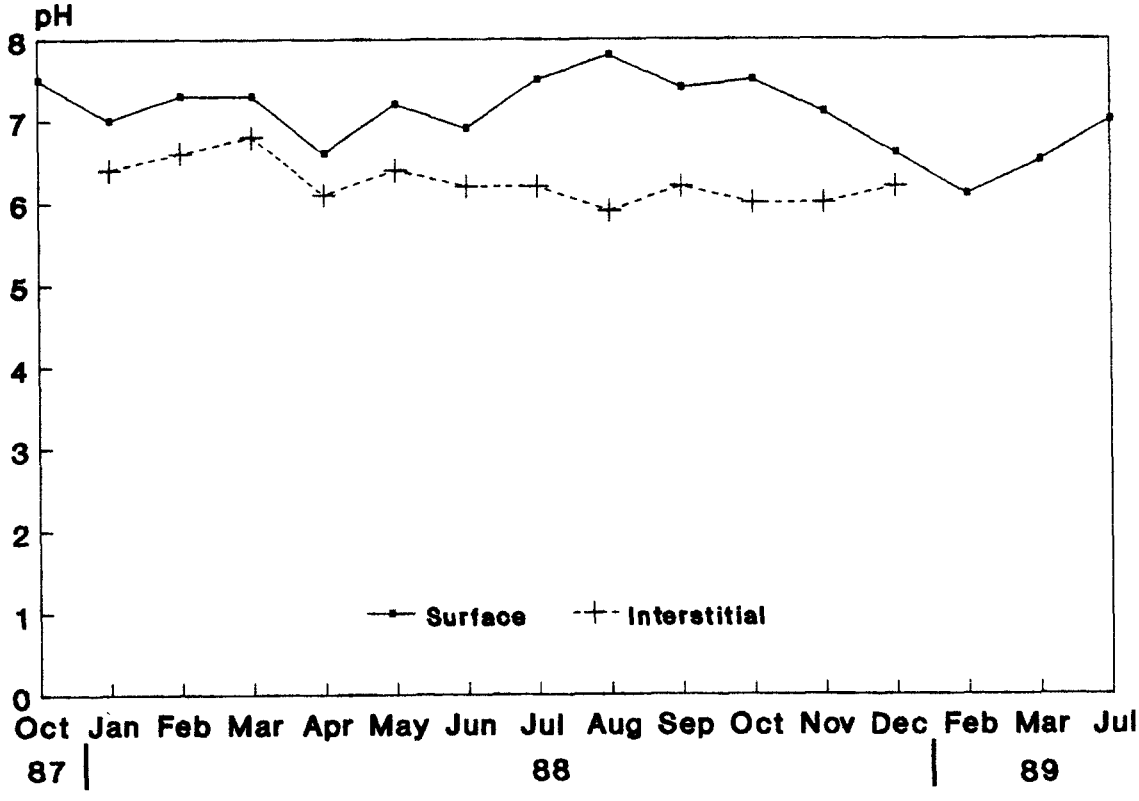



Figure 4. Surface and interstitial zone pH values for a 27-acre naturally established reedgrass wetland subjected to acid loading (62.6 million gallons; pH 3.3, acidity 262 ppm) during April 1988 through February 1989.

Table 3. Water quality summary data for surface and interstitial zones of a 27-acre naturally established reedgrass wetland (IC1) at the Peabody Will Scarlet Mine, Williamson County, Illinois.

| Sample Date | Depth | pH | Acidity[1] (ppm $CaCO_3$) | Alkalinity[1] (ppm $CaCO_3$) | Conductivity (µmhos/cm) | Dissolved Oxygen (ppm) | Redox Potential (MV) |
|---|---|---|---|---|---|---|---|
| 10/87 | Surface (n=4) | 7.5 | 0 | 79 | 2,150 | 7.1 | -- |
| 11/87 | Interstitial (n=4) | 6.4 | -- | -- | 2,675 | 0.0 | -48 |
| 1/88 | Surface (n=19) | 7.0 | -16 | 74 | 2,142 | 9.6 | -- |
| | Interstitial (n=4) | 6.4 | 27 | 222 | 2,962 | 0.0 | -71 |
| 2/88 | Surface (n=19) | 7.3 | -- | -- | 1,821 | -- | -- |
| | Interstitial (n=4) | 6.6 | -- | -- | 2,675 | 0.0 | -62 |
| 3/88 | Surface (n=19) | 7.3 | -15 | 68 | 1,505 | 10.0 | +109 |
| | Interstitial (n=4) | 6.8 | 368 | 431 | 2,675 | 0.0 | -70 |
| 4/88 | Surface (n=19) | 6.6 | -34 | 74 | 2,137 | 7.1 | +46 |
| | Interstitial (n=4) | 6.1 | 328 | 356 | 3,075 | 0.0 | -93 |
| 5/88 | Surface (n=19) | 7.2 | -79 | 112 | 2,016 | 5.7 | +25 |
| | Interstitial (n=4) | 6.4 | 235 | 319 | 2,800 | 0.0 | -126 |
| 6/88 | Surface (n=19) | 6.9 | -60 | 121 | 2,279 | 4.7 | -31 |
| | Interstitial (n=4) | 6.2 | 246 | 256 | 2,825 | 0.0 | -128 |
| 7/88 | Surface (n=19) | 7.5 | -40 | 91 | 2,216 | 12.9 | +43 |
| | Interstitial (n=4) | 6.2 | 414 | 283 | 2,925 | 0.0 | -151 |
| 8/88 | Surface (n=19) | 7.8 | -73 | 125 | 2,300 | 10.8 | +16 |
| | Interstitial (n=4) | 5.9 | 334 | 291 | 2,975 | 0.0 | -142 |
| 9/88 | Surface (n=19) | 7.4 | -78 | 103 | 2,316 | 10.7 | +44 |
| | Interstitial (n=4) | 6.2 | 328 | 333 | 2,875 | 0.0 | -118 |
| 10/88 | Surface (n=19) | 7.5 | -78 | 106 | 2,245 | 10.4 | +113 |
| | Interstitial (n=4) | 6.0 | 319 | 370 | 3,000 | 0.0 | -92 |
| 11/88 | Surface (n=19) | 7.1 | -49 | 109 | 2,032 | 10.2 | +129 |
| | Interstitial (n=4) | 6.0 | 6 | 467 | 2,775 | 0.0 | -78 |
| 12/88 | Surface (n=19) | 6.6 | -41 | 59 | 1,874 | 11.9 | +114 |
| | Interstitial (n=4) | 6.2 | 285 | 381 | 2,625 | 0.0 | -67 |
| 1/89 | Surface (n=5) | 6.3 | -11 | 30 | 1,870 | -- | -- |
| 2/89 | Surface (n=5) | 5.0 | 76 | 14 | 2,080 | -- | -- |
| 3/89 | Surface (n=1) | 6.5 | 21 | 24 | -- | -- | -- |
| 4/89 | Surface (n=1) | 6.5 | 22 | 30 | -- | -- | -- |
| 7/89 | Surface (n=1) | 7.0 | -32 | 72 | -- | -- | -- |

1. n=4 for all depths except the 3, 4, and 7/89 samplings where n=1.

acid inputs as it had during the past 5 to 7 years. Factors contributing to the ability of this wetland to tolerate acid loading without permanent degredation of its alkaline buffering capacity suggested natural processes such as sulfate reduction can maintain an acceptable acidity-alkalinity equilibrium under field conditions if biochemical thresholds of the wetland systems are not exceeded.

Wetland treatment system can be viewed as a "biological shock absorber" capable of absorbing some environmental shocks and handling normal loads. However, subjecting an environmental system to constant abuse or even a short series of acute stressful events can sufficiently exceed the system's loading capacity and its ability to recover. Once a biological system has been degraded, time is required for natural recovery processes to restore a stable and healthy equilibrium. In the case of acid treatment wetlands, sizing and flow design must accomodate a reasonable constant and even distribution of acid flows (both quanitity and quality) to prevent deterioration of biochemical processing ability at the input point. Biological thresholds must be determined and upper limits of tolerance to acid loading can not be exceeded.

The wetland system will need to be well established and diverse to prevent disruption of fragile equilibriums characteristic of recently constructed wetlands. Natural productivity of above-ground biomass is required for natural recovery processes to restore a stable and healthy equilibrium. Natural productivity of above-ground biomass or introduction of other organic matter sources should be maximized to facilitate sulfate reduction in anaerobic substrates.

The previous recommendations represent a general description of the Will Scarlet treatment wetland (IC1) and other reclamation practices currently being implemented at wetland sites at Will Scarlet. Sewage sludge is being used as an organic substrate for alkaline capped (AMD sludge) slurry impoundments. Sewage sludge capped slurry wetlands are being designed to function as additional biochemical buffering systems for chronic acid runoff which originates from pre-law acid seeps. The overall reclamation goal of the Will Scarlet site will be to eventually (3 to 5 years) provide sufficient wetland buffering capacity to eliminate the need for chemical water treatment.

The acid-loading trials summarized in this paper served both as an empirical evaluation of wetland treatment potential of a naturally established wetland, and as a guideline for "not-to-exceed" loading rates for future design and operation of wetland treatment units at Will Scarlet. This demonstration did not control or accurately measure effects or functions of the many physical, vegetational, or chemical components operating within the treatment wetland. Wetlands will always be the most diverse and dynamic natural system ever established on mined lands, or ever used in the treatment process of mine drainage. Understanding or documenting the complexity of wetlands does not affect their intrinsic ability to perform their many functions. From waterfowl habitat enhancement to water quality improvement, wetlands will continue to serve a vital function in the reclamation of the Peabody Will Scarlet mine.

## Literature Cited


Brodie, G.A., D.A. Hammer, and D.A. Tomljanovich. 1988. An evaluation of substrate types in constructed wetlands acid drainage treatment systems. Pages 389-398 in 1988 Mine Drainage and Surface Mine Reclamation. Conf. Vol I: Mine Water and Mine Waste. USDI Bur. of Mines, Infor. Circ. 9183.

Brugam, R.B., M.A. Carlson, S. Chakraverty, and M. Lusk. 1983. Postmining neutralization of acidic surface mine lakes. Pages 79-88 in Univ. of Illinois Water Resource Center, Urbana-Champaign. UILV-WRC-83-0178, Res. Rep. 178.

Dugan, P.R. 1970. Removal of mine water ions by microbial polymers. Pages 279-283 in Third Symp. on Coal Mine Drainage Research, Mellon Inst., Pittsburgh, PA.

Hedin, R.S. 1989. Treatment of coal mine drainage with constructed wetlands. Wetl. Ecol. and Cons. 28:349-362.

Hedin, R.S., D.M. Hyman, and R.W. Hammack. 1988. Implications of sulfate-reduction and pyrite formation processes for water quality in a constructed wetland: preliminary observations. Pages 383-388 in 1988 Mine Drainage and Surface Mine Reclamation Conf. Vol. I: Mine Water and Mine Waste. USDI Bur. of Mines, Infor. Circ. 9183.

King, D.L., J.J. Simmler, C.S. Decker, and C.W. Ogg. 1974. Acid strip mine lake recovery. J. Water Pollution Control Fed. 46(10):2301-2316.

Kleinmann, R.L.P., T.O. Tiernan, J.G. Solch, and R.L. Harris. 1983. A low-cost, low-maintenance treatment system for acid mine drainage using sphagnum moss and limestone. Pages 241-245 in Symp. on Surface Mining, Hydrology, Sedimentology, and Reclamation. Univ. of Kentucky, Lexington.

Klimstra, W.D. and J.R. Nawrot. 1986. Wetlands/ wildlife habitat development. Pages 187-222 in The National Mined Land Reclamation Conf., St. Louis, MO.

Klimstra, W.D. and J.R. Nawrot. 1982. Water as a reclamation alternative: an assessment of values. Pages 39-44 in Symp. on Surface Mining, Hydrology, Sedimentology, and Reclamation. Univ. of Kentucky, Lexington.

Mooreman, F.R. and W. Van Breeman. 1978. Rice: soil, water, land. Int. Rice Res. Inst., Los Banos, Phillippines. 185pp.

Nawrot, J.R., J. Sandusky, and W.B. Klimstra. 1988. Acid soils reclamation: applying the principles. Pages 93-103 in 1988 Mine Drainage and Surface Mine Reclamation Conf. Vol. II: Mine Reclamation, Abandoned Mine Lands and Policy Issues. USDI Bur. of Mines, Infor. Circ. 9183.

Ponnamperuma, F.M. 1972. The chemistry of submerged soils. Adv. Agron. 24:29-96.

Rice, P.A. and F. Rabolini. 1972. Biological treatment of acid mine water. Pages 293-306 in Fourth Symp. on Coal Mine Drainage Research, Mellon Inst., Pittsburgh, PA.

Stowell, R., R. Ludwig, J. Colt, and G. Tchobanoglous. 1981. Concepts in aquatic treatment system design. Pags 919-940 in J. Environ. Eng. Div., Am. Soc. Civil Eng., Vol. 107, No. EE5, Proc. Paper 16555.

Tuttle, J.H., P.R. Dugan, and C.I. Randles. 1969a. Microbial sulfate reduction and its potential utility as an acid mine water pollution abatement procedure. Applied Microbiol. 17(2):297-302.

_____, et al. 1969b. Microbial dissimilatory sulfur cycle in acid mine water. J. Bacteriol. 97(2):594.

Weider, R.K. and G.E. Lang. 1982. Modification of acid mine drainage in a freshwater wetland. Pages 43-53 in Symp. on Wetlands of the Unglaciated Appalachian Region, West Virginia Univ., Morgantown.

This page intentionally left blank.

# DIVERSITY PATTERNS OF INVERTEBRATE FAUNA IN CATTAIL WETLANDS RECEIVING ACID MINE DRAINAGE[1]

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

Abstract. Invertebrate diversity patterns at a three-celled, 0.3 ha, cattail wetland, constructed in 1985 to receive mine drainage, were surveyed during May and June in 1988 and 1989. Three nearby sites, each comprised of three volunteer cattail cells, were also examined. Benthic invertebrates were collected using an Ekman dredge and volant insects were obtained with sticky traps covered with tangle-trap. Dredge samples were examined at the family level and sticky trap samples at the ordinal level.

Substrates from the constructed wetland contained significantly fewer taxonomic families and total numbers of invertebrates than the other sites. Shannon-Weaver diversity indices (H') also demonstrated that the constructed wetland supported a simple community relative to the other sites; however, the number of invertebrate families in the constructed wetland increased from three in 1988 to eight in 1989. Dipteran larvae were the most abundant invertebrates in the sediments from the constructed wetland.

The constructed wetland supported significantly higher numbers of volant insects than the other sites, but no difference in the number of taxonomic orders was found across sites. Shannon-Weaver diversity indices showed the constructed wetland to be intermediate in ordinal diversity relative to the other sites, with these values influenced by a site by year interaction effect. The number of orders of insects recorded in sticky trap samples at the constructed wetland increased from seven in 1988 to nine in 1989. The implications of these findings for assessing the overall value of surface mine wetlands are discussed.

Additional key words: Aquatic invertebrates, diversity indices, surface mine wetlands.

## Introduction

The construction of wetlands to treat acid mine drainage is a practice that is receiving increased acceptance (Girts and Kleinmann 1986), as these systems require limited maintenance and are more cost effective than the continued application of chemicals. Research examining the functional efficiency of constructed wetlands for removal of iron and manganese, along with the elevation of water pH level, has been conducted (Stark et al. 1988, Stillings et al. 1988), but a complete evaluation of the environmental integrity of these wetlands requires an examination of the biotic components that are contained within these systems.

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]M. J. Lacki is an Assistant Professor, Department of Forestry, University of Kentucky, Lexington, KY; J. W. Hummer is an Instructor, Wildlife Technology Program, The Pennsylvania State University, DuBois, PA; and H. J. Webster is an Assistant Professor of Biology, The Pennsylvania State University, DuBois, PA.

The use of food chain support in assessing a wetland's potential for sustaining fish and wildlife populations is arguably a valid approach (Harris 1988), as a wetland will function depending upon its proximity to other wetlands and terrestrial input sources in the surrounding landscape (Klopatek 1988). However, if constructed wetlands are to function as ecological equivalents of natural wetlands, then all system processes, including food chain support, should be established. Klopatek (1988) indicated that wetland food chains begin with autotrophs or detritus. A plant base (autotrophs) is normally provided in constructed wetlands, with the detrital component left to develop over time. The implications of this approach for aquatic invertebrates inhabiting the benthic zone of constructed wetlands have not been resolved. Benthic invertebrate communities are dependent on the supply of detritus, and thus represent an integral component of detritus-based food chains in wetland systems.

Studies examining invertebrate taxa in surface-mine waters have emphasized lakes and ponds that were mostly alkaline in pH and generally supportive of diverse biotic communities (Hepp 1987, Fowler et al. 1985, Jones et al. 1985). Wetlands constructed in the headwaters of stream drainages (lotic systems), designed primarily to filter water containing acid mine runoff, have not been examined for benthic invertebrates. Bosserman and Hill (1985) showed that populations of benthic invertebrates in natural stream headwaters were negatively impacted when exposed to drainage inputs from surface mines, suggesting that benthic invertebrates inhabiting wetlands constructed to treat surface mine runoff may exhibit similar stress responses.

A thorough evaluation of any constructed wetland requires an examination of the biotic communities contained therein, and an assessment of impacts upon the biota downstream. Pinder and Farr (1987) indicated, with some exceptions, that diversity of aquatic organisms declined under exposure to environmental stress. We present results from an ongoing study examining diversity patterns of benthic invertebrates and volant insects occupying a constructed, cattail (Typha latifolia) wetland in relation to natural cattail wetlands both downstream and within the surrounding watershed. This paper represents one phase of a long-term study evaluating the diversity patterns of organisms at all levels of the food chain in the constructed wetland.

## Materials and Methods

Four sites were selected for study. All sites were located within the Wills Creek drainage system, in Coshocton and Muskingum Counties, Ohio. A history and description of the constructed wetland (Simco #4) is presented in Stark et al. (1988). The remaining three sites were each composed of three cattail cells to allow direct comparison with the design of the Simco #4 wetland. One site was located directly downstream (Downstream) from the constructed wetland. This site had been slightly disturbed by the presence of a power line right-of-way neighboring the output end of the third cattail cell, and by historical mining activity throughout the surrounding hillsides. Mining activity ceased in the vicinity of the Downstream site as of 1961 (T. Romanoski, pers. commun.). The third site (Volunteer) represented a drainage system with minimal disturbance from mining activity, whereas the final site (Reclamation) represented a reclaimed strip mine with volunteer cattail wetlands present in the drainage bottom.

Samples for benthic invertebrates and volant insects were collected during four sampling periods: 19 to 25 May, 1988; 10 to 15 June, 1988; 22 to 26 May, 1989; and 12 to 16 June, 1989. Benthic invertebrates were obtained using an Ekman dredge sampler, with five samples collected randomly from each cattail cell by site combination during each sampling period. Macroinvertebrates were extracted from each substrate sample and identified to family using a binocular microscope. Volant insects were captured using 0.18 square meter-sticky traps suspended approximately 0.75 m aboveground. White (color) traps were covered with tangle-trap adhesive on each side. Two traps were established at the midpoint of each cattail cell and run for two nights during each sampling period. Insects were identified to order using a binocular microscope and were enumerated. The Shannon-Weaver diversity index (H') was derived for all faunal samples, with data grouped by cattail cell. A discussion of this computational method is provided in Hair (1980). The value in using a diversity index over biotic indices to assess stress responses of invertebrates in wetland systems is that they do not assume any tolerance level for a particular species group to a given pollutant (Pinder and Farr 1987).

Water quality was assessed by collecting replicate samples from the inflow to each cattail cell, and the outflow of the cattail cell furthest downstream at a site. One sample was acidified for metal analysis with nitric acid, while the other sample remained untreated and was analyzed for remaining properties. Samples were iced, delivered to the Central Environmental Services testing facility (Clearfield, PA 16830),

Table 1.--Numbers of macroinvertebrates recorded in benthic samples across wetlands sampled, 1988 and 1989 combined.

| Taxonomic order | Common name | No. Families | No.Collected Simco | Downs | Volun | Recla |
|---|---|---|---|---|---|---|
| Molluscs | | | | | | |
| Heterodonta | bivalves | 1 | - | 306 | 472 | 758 |
| Basommatophora | pulmonate snails | 3 | - | 432 | 87 | 46 |
| Stylommatophora | gilled snails | 1 | - | - | 8 | 1 |
| Crustaceans | | | | | | |
| Cyclopoida | copepods | 1 | 1 | 347 | 179 | 111 |
| Decapoda | crayfish, shrimp | 2 | - | - | 1 | 109 |
| Isopoda | sowbugs | 1 | - | - | 2 | - |
| Insects | | | | | | |
| Diptera | flies | 9 | 59 | 153 | 118 | 122 |
| Neuroptera | alderflies | 1 | - | 10 | 56 | 12 |
| Coleoptera | beetles | 5 | 2 | 6 | 24 | 10 |
| Trichoptera | caddisflies | 2 | - | 29 | - | 6 |
| Odonata | dragonflies | 1 | 2 | 1 | - | - |
| Collembola | springtails | 1 | - | 1 | - | - |
| Homoptera | aphids | 1 | 1 | - | - | - |

and analyzed for iron, manganese, pH value, acidity, and alkalinity. Samples were collected on 10 August in both 1988 and 1989.

All data were analyzed using a balanced, nested analysis of variance (SAS Institute 1982). ANOVA's were treated as mixed effects models, with sites as fixed, main effects. The random effect varied among tests depending on the data collection procedure. For all tests of faunal data cattail cells were used as the random effect. Sample location was used as the random effect for tests of water quality because samples were acquired between, and not within, cattail cells. Subsequently, cattail cell mean square and sample location mean square were used as the error terms for testing site effects. Where appropriate, the month and year of sampling were used as block effects and evaluated for possible interactions with site effects. For all models where significant site effects were obtained, site means were tested using Tukey's studentized range test. Test outcomes were considered significantly different when a $P < 0.05$ was achieved. Data for number of benthic invertebrates and iron content were both log transformed prior to analysis to surmount the problem of unequal variances across site (main effect) means.

## Results and Discussion

Insect larvae represented the majority of taxonomic families of macroinvertebrates found in the dredge samples (Table 1). Dipterans (flies) were the most abundant group in the benthos of the constructed wetland with six families present. Of these, the Chironomidae were most common representing 76.3% (n = 45) of the number of dipteran larvae found. All

Table 2.--Total numbers per sample (N), number of families per sample (F) and diversity patterns (H') of benthic invertebrates across wetlands sampled, 1988 and 1989 combined.

| Sites | No. Samples | N Mean | N s.d. | F Mean | F s.d. | H' Mean | H' s.d. |
|---|---|---|---|---|---|---|---|
| Simco #4 | 60 | $5.42^{b}$ | 7.15 | $1.33^{b}$ | 1.37 | $0.31^{b}$ | 0.50 |
| Downstream | 60 | $107.9^{a}$ | 86.3 | $6.17^{a}$ | 1.95 | $1.11^{a}$ | 0.42 |
| Volunteer | 60 | $80.2^{a}$ | 59.6 | $6.67^{a}$ | 2.10 | $1.13^{a}$ | 0.38 |
| Reclamation | 60 | $98.1^{a}$ | 96.4 | $7.08^{a}$ | 1.08 | $1.13^{a}$ | 0.47 |

$^{a,b}$Within columns, means without common letters are significantly different at $P = 0.05$. Data for N were log transformed prior to analysis.

Table 3.--Numbers of invertebrates recorded in sticky trap samples across wetlands sampled, 1988 and 1989 combined.

| Taxonomic order | Common name | No.Collected | | | | % |
|---|---|---|---|---|---|---|
| | | Simco | Downs | Volun | Recla | Total |
| Diptera | flies | 3537 | 2521 | 2377 | 1658 | 82 |
| Coleoptera | beetles | 385 | 332 | 250 | 323 | 11 |
| Homoptera | aphids, leaf hoppers | 69 | 41 | 51 | 78 | 3 |
| Hymenoptera | bees, wasps | 63 | 57 | 21 | 29 | 1 |
| Hemiptera | bugs | 18 | 62 | 14 | 25 | 1 |
| Lepidoptera | butterflies, moths | 20 | 15 | 18 | 30 | * |
| Araneida | spiders | 16 | 16 | 13 | 13 | * |
| Neuroptera | alderflies | 3 | 6 | 7 | 20 | * |
| Orthoptera | crickets, grasshoppers | 2 | 1 | 2 | 2 | * |
| Odonata | dragon & damselflies | 1 | 2 | 1 | - | * |
| Trichoptera | caddisflies | - | 1 | - | 1 | * |
| Ephemeroptera | mayflies | - | - | 1 | - | * |

* = less than 1%.

of these were recorded in the 1988 sampling effort. The constructed wetland was completely lacking in bivalves (clams) and snails, and exhibited a paucity of crustaceans, with only one copepod present across the two years sampled.

The remaining three wetlands were much richer in both numbers and families of benthic invertebrates than Simco #4 (Tables 1 and 2), yielding significantly greater values for diversity (H'). No significant month or year effects were obtained for numbers, families, or diversity of benthic invertebrates. The Downstream, Volunteer, and Reclamation sites appeared to support benthic communities that were more stable than those for the Simco #4 site. In the former case, the number of taxonomic families found at a site during a particular year ranged from 14 to 16, whereas at Simco #4 the number increased from three in 1988 to eight in 1989. Further, the number of families found to occur in common to both years of collecting at a site ranged from eight to 11 for the natural wetlands, while only one family (Libellulidae - dragonfly larvae) was found to occur in Simco #4 during both years sampled.

Dipterans were the most frequently captured invertebrate in sticky trap samples across all four sites (Table 3). With the exception of a small number of spiders (Araneida), volant insects made up all of the specimens collected on sticky traps. The abundance of insect orders did not differ across sites and remained stable between sampling years. Simco #4 increased in the abundance of insect orders from seven to nine, whereas the Downstream site decreased from nine to seven. Simco #4 exhibited significantly higher numbers of volant insects in sticky

Table 4.--Total numbers per sample (N), number of orders per sample (O), and diversity patterns (H') of volant insects across wetlands sampled, 1988 and 1989 combined.

| Sites | No. Samples | N | | O | | H' | |
|---|---|---|---|---|---|---|---|
| | | Mean | s.d. | Mean | s.d. | Mean | s.d. |
| Simco #4 | 24 | 343[a] | 140.8 | 5.92 | 1.31 | 0.58[a,b] | 0.20 |
| Downstream | 24 | 254[a,b] | 89.0 | 6.58 | 0.90 | 0.66[a,b] | 0.20 |
| Volunteer | 24 | 230[b] | 69.6 | 6.25 | 1.06 | 0.54[b] | 0.17 |
| Reclamation | 24 | 182[b] | 45.4 | 6.25 | 1.48 | 0.78[a] | 0.21 |

[a,b]within columns, means without common letters are significantly different at P = 0.05.

trap samples than either the Volunteer or Reclamation site and was intermediate in ordinal diversity (Table 4). The significantly higher numbers were due primarily to the abundance of dipterans collected (Table 3). This significantly higher abundance of adult flies at Simco #4 may at first be partially explained by the dominance of fly larvae in the dredge samples for this site (Table 1); however, given the overall lower abundance of benthic invertebrates at Simco #4 relative to the natural wetlands, this explanation is unlikely.

A significant year by site interaction effect ($P < 0.03$) was obtained for the analyses of ordinal diversity (H'). This outcome was difficult to interpret as mean diversity levels for the Simco #4 and Downstream sites did not vary between years, while the Volunteer and Reclamation sites declined in mean value from 1988 to 1989 (Table 5). Why such a difference would occur among the sites examined is unclear, but it may be tied to the substantially different rainfall patterns that resulted in 1988 and 1989, as a severe drought took place during the growing season of 1988.

Metal analysis showed no differences for manganese across all four sites examined (Table 6). Levels of iron in the water at the Downstream site were not significantly different from the remaining natural wetlands, indicating that the constructed wetland was removing substantial amounts of iron because iron levels in Simco #4 were significantly higher than all other sites (Table 6).

Mean pH values increased significantly ($P < 0.0001$) across all sites from 1988 to 1989. Simco #4 had significantly lower pH values than the three natural wetlands (Table 6), with this pattern paralleled by mean acidity levels, as Simco #4 had significantly higher levels of acidity than the remaining sites. A significant year by site interaction effect ($P < 0.004$) was obtained for mean acidity levels, but an examination of the means across years revealed no interpretable pattern. Levels of alkalinity in Simco #4 were intermediate to those for the other sites, with the Reclamation site exhibiting significantly higher levels than the remaining three sites (Table 6). A significant year by site interaction effect ($P < 0.0001$) was also found for alkalinity, with the Downstream, Volunteer, and Reclamation sites all showing large declines from 1988 to 1989.

The constructed wetland (Simco #4) supported a much simpler and relatively unstable benthic community than natural wetlands in the surrounding watershed. Whether these differences were due to higher acidity levels, higher levels of iron, synergistic effects of the two aforementioned factors, or to an insufficient time for benthic communities to develop is uncertain. Eilers et al. (1984) compiled a literature review that demonstrated tolerance of pH levels by a variety of macroinvertebrates equal to and below pH levels recorded at Simco #4. The dominance by dipterans, particularly chironomids, in samples from Simco #4 coincided with a study examining impacts of acid mine drainage on stream biota that showed the most heavily impacted streams to have simpler benthic communities comprised primarily of chironomids (Bosserman and Hill 1985). Hepp (1987) examined newly created surface-mine wetlands not impacted by acid drainage and also found chironomids to be the most frequent colonizer in the benthos of these systems.

In terms of diversity (H') and overall richness of macroinvertebrates, the Downstream site appeared to be functioning in a manner compatible with the other natural wetlands examined. This suggested that the Simco #4 wetland was ameliorating probable negative impacts from acid drainage on the benthic fauna directly downstream from the point source. Water chemistry analyses supported this conclusion, indicating that water quality at the Downstream site was comparable to that for the Volunteer and Reclamation sites.

Larson (1988) has pointed out that achieving acceptable standards for wetland construction might be contingent on allowing sufficient time for some essential ecosystem components to develop naturally after a wetland is established

Table 5.--Mean ordinal diversity levels (H') for volant insects by year across wetlands sampled.

| | Simco #4 | | Downstream | | Volunteer | | Reclamation | |
|---|---|---|---|---|---|---|---|---|
| Year | Mean | s.d. | Mean | s.d. | Mean | s.d. | Mean | s.d. |
| 1988 | 0.55 | 0.21 | 0.64 | 0.16 | 0.65 | 0.12 | 0.92 | 0.17 |
| 1989 | 0.60 | 0.21 | 0.68 | 0.25 | 0.44 | 0.15 | 0.64 | 0.14 |

Table 6.--Water quality parameters for wetlands examined. Except for pH value, all units are in mg/liter. Sample size equals 4 in all cases.

| | | pH | | Fe | | Mn | | Acidity | | Alkalinity | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Site | Year | Mean | s.d. | Mean | s.d. | Mean | s.d. | Mean | s.d. | Mean | s.d. |
| Simco | 1988 | 5.92[b] | 0.06 | 75.3[a] | 46.9 | 1.93 | 0.08 | 28.8[a] | 27.4 | 102[b] | 42.2 |
| | 1989 | 6.26 | 0.05 | 52.8 | 24.7 | 1.48 | 0.06 | 18.6 | 24.8 | 104 | 36.2 |
| Downs | 1988 | 6.59[a] | 0.25 | 0.84[b] | 0.95 | 0.52 | 0.27 | -22.4[b] | 6.06 | 62.5[b] | 7.11 |
| | 1989 | 6.76 | 0.12 | 0.56 | 0.08 | 0.64 | 0.05 | -26.4 | 2.01 | 46.1 | 1.89 |
| Volum | 1988 | 6.88[a] | 0.15 | 2.74[b] | 2.98 | 3.83 | 2.83 | -52.2[b] | 8.19 | 106[b] | 2.96 |
| | 1989 | 7.12 | 0.10 | 2.53 | 3.15 | 2.57 | 1.48 | -44.6 | 3.35 | 73.8 | 4.80 |
| Recla | 1988 | 6.96[a] | 0.42 | 0.97[b] | 0.76 | 3.54 | 2.20 | -127[c] | 32.4 | 200[a] | 28.2 |
| | 1989 | 7.18 | 0.54 | 5.71 | 10.5 | 1.90 | 1.70 | -121 | 35.4 | 156 | 35.9 |

[a,b,c]Within columns, means without common letters are significantly different at P = 0.05. Tests based on the overall means for a site with years combined. Data for Fe were log transformed prior to analysis.

(i.e., a time lag effect). Data for Simco #4 indicate that aquatic macroinvertebrates were poorly represented in the benthos, but adult, volant insect communities were comparable to nearby natural wetlands. Adult insects are more mobile than aquatic macroinvertebrates and were able to colonize the available habitat at Simco #4 more quickly. The rate at which a constructed wetland achieves a complete faunal composition is likely to be a function of its juxtaposition to other wetlands nearby (Klopatek 1988). However, where wetlands are implemented to ameliorate environmental disturbance, achievement of the equivalent to naturally functioning systems may take a longer period of time or may not be possible at all. In the latter case, prevention of negative impacts downstream from constructed wetlands would still justify their implementation, especially if partial wetland benefits were accrued in the vicinity of the wetland without creating an additional environmental hazard through bioaccumulation of metals in the tissues of vertebrate animals at higher levels in the food chain. Iron and manganese levels are presently being examined in tissues of vertebrates inhabiting the Simco #4 wetland and vicinity.

No data exist to indicate whether modifications to wetland design could hasten the development of a complex, benthic invertebrate community. In the particular case of riparian, headwater wetlands, modifications in shoreline terrestrial vegetation to provide shading effects and allocthonous inputs of woody debris may facilitate the establishment of some benthic species through creation of more suitable habitat conditions (Vannote et al. 1980, Minshall et al. 1985).

Acknowledgements

Support for this research was provided by a grant from American Electric Power Service Corporation, Division of Environmental Engineering - Fuel Supply, Lancaster, OH. We thank T. Romanoski for helpful advice and logistic support. We are grateful to R. Baumgart, L. Bearer, B. Burhans, M. Magee, and K. Roser for field and laboratory assistance.

Literature Cited

Bosserman, R. W., and P. L. Hill, Jr. 1985. Community ecology of three wetland ecosystems impacted by acid mine drainage. p. 287-303. In Brooks, R. P., D. E. Samuel, and J. B. Hill, eds. Wetlands and Water Management on Mined Lands: Proceedings of a conference. (The Pennsylvania State University, University Park, PA, October 23-24, 1985).

Eilers, J. M., G. J. Lien, and R. G. Berg. 1984. Aquatic organisms in acid environments: A literature review. Wisconsin Dept. of Natural Resour., Tech. Bull. No. 150. 18 p.

Fowler, D. K., D. M. Hill, and L. J. Fowler. 1985. Colonization of coal surface mine sediment ponds in southern Appalachia by aquatic organisms and breeding amphibians. p. 261-285. In Brooks, R. P., D. E. Samuel, and J. B. Hill, eds. Wetlands and Water Management on Mined Lands: Proceedings of a conference. (The Pennsylvania State University, University Park, PA, October 23-24, 1985).

Girts, M. A., and R. L. P. Kleinmann. 1986. Constructed wetlands for treatment of acid mine drainage: A preliminary review. p. 165-171. In 1986 National Symposium on Mining, Hydrology, Sedimentology, and Reclamation, Proceedings. University of Kentucky, Lexington.

Hair, J. D. 1980. Measurement of ecological diversity. p. 269-275. In Schemnitz, S. D., ed. Wildlife Management Techniques Manual, 4th ed., The Wildlife Society. Washington, D.C.

Harris, L. D. 1988. The nature of cumulative impacts on biotic diversity of wetland vertebrates. Environ. Manage. 12(5):675-693.

Hepp, J. P. 1987. An ecological survey of four newly created surface-mine wetlands in central Pennsylvania. M. Sc. Thesis, The Pennsylvania State University, University Park. 186 p.

Jones, D. W., M. J. McElligott, and R. H. Mannz. 1985. Summary of biological, chemical, and morphological characterizations of 33 surface-mine lakes in Illinois and Missouri. p. 211-238. In Brooks, R. P., D. E. Samuel, and J. B. Hill, eds. Wetlands and Water Management on Mined Lands: Proceedings of a conference. (The Pennsylvania State University, University Park, PA, October 23-24, 1985).

Klopatek, J. M. 1988. Some thoughts on using a landscape framework to address cumulative impacts on wetland food chain support. Environ. Manage. 12(5):703-711.

Larson, J. S. 1988. Wetland creation and restoration: An outline of the scientific perspective. p. 73-79. In Zelazny, J., and J. S. Feierabend, eds. Increasing Our Wetland Resources: Proceedings of a conference. (National Wildlife Federation, Washington, D.C., October 4-7, 1987).

Minshall, G. W., K. W. Cummins, R. C. Petersen, C. E. Cushing, D. A. Bruns, J. R. Sedell, and R. L. Vannote. 1985. Developments in stream ecosystem theory. Can. J. Fish. Aquat. Sci. 42:1045-1055.

Pinder, L. C. V., and I. S. Farr. 1987. Biological surveillance of water quality - 3. The influence of organic enrichment on the macroinvertebrate fauna of small chalk streams. Arch. Hydrobiol. 109(4):619-637.

SAS Institute. 1982. SAS user's guide: Statistics. 1982 ed., SAS Institute Inc., Cary, NC. 584 p.

Stark, L. R., R. L. Kolbash, H. J. Webster, S. E. Stevens, Jr., K. A. Dionis, and E. R. Murphy. 1988. The Simco #4 wetland: Biological patterns and performance of a wetland receiving mine drainage. p. 332-344. In Mine Drainage and Surface Mine Reclamation, Volume I: Proceedings of a conference. (Pittsburgh, PA, April 19-21, 1988).

Stillings, L. L., J. J. Gryta, and T. A. Ronning. 1988. Iron and manganese removal in a Typha -dominated wetland during ten months following its construction. p. 317-322. In Mine Drainage and Surface Reclamation, Volume I: Mine Water and Mine Waste., U.S. Bureau of Mines Information Circular 9183.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. 1980. The river continuum concept. Can. J. Fish. Aquat. Sci. 37:130-137.

This page intentionally left blank.

# WILDLIFE USE OF MITIGATED WETLANDS ON SURFACE MINED LANDS IN WESTERN PENNSYLVANIA [1]

by

Fred J. Brenner and Daniel L. Hofius[2]

Abstract. Twelve wetlands totaling 22.4 ha were constructed on 2 mine sites in western Pennsylvania to mitigate 6.6 ha destroyed by mining. A total of 24250 wetland plants and 14,000 trees and shrubs were planted on the 2 sites at a total cost of $17,068. Sixty species of birds representing 23 families and 18 feeding guilds, including 7 species of special concern in Pennsylvania, were observed using these wetlands during the first 2 years. Broods of wood ducks (Aix sponsa), Canada geese (Branta canadensis) and mallards (Anas platyrhynchos) were observed on both mine sites. In addition, 17 species of mammals and 8 and 7 species of amphibians and reptiles, respectively, were observed using these wetlands. Based on these results, functional wetlands may be established on mined lands without added cost to the mine operator and their construction should be given prime consideration by the mine industry and regulatory agencies.

Additional key words: Wetland construction, cost involved, vegetation establishment wildlife use.

## Introduction

Wetlands are a declining resource in the middle Atlantic region of the United States (Tiner 1987) as well as elsewhere in North America. The reclamation of surface coal mines provide an opportunity to establish wetland communities without additional costs to the mine operator (Brenner 1987). For over 4 decades, wetlands have been developed on mine lands, especially in the eastern (Brenner and Steiner 1987) and midwestern (Jones et al. 1985) regions of the United States. These wetlands provide habitat for a variety of wildlife species, including amphibians (brooks et al. 1985, Fowler et al, 1985), reptiles (Brooks et al. 1985), birds and mammals (Brenner and Mondok 1979, Sandusky 1978, Thompson 1984, Brenner and Sterner (1988), as well as their use for resting and feeding during migration (Brenner and Steiner 1987). Although wetlands and mine lands have been shown to be used extensively by wildlife, they are often left with little if any planning for their integration into the overall reclamation of the mine site.

Avian fauna has been suggested an an index of habitat quality in natural wetland ecosystems (Cable et al. 1989, Glooschenko et al. 1988) and would also be applicable to man-made wetland habitats. The primary objective of this paper is to describe and discuss the evaluation of wetlands established on 2 mine sites in western Pennsylvania as to their value as wildlife habitats using birds as the primary indicator of habitat quality. For over a decade, I have advocated the development of wetlands on mine lands (Brenner 1973, 1983), including special areas designed for rare and endangered species (Brenner 1985, 1986). In this paper, the construction and establishment of wetlands on mine lands and their ultimate use by wildlife will be discussed.

[1] Paper presented at the 1990 Mining Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2] Fred J. Brenner is a professor and Daniel L. Hofius is a student, Biology Department Grove City College, Grove City, PA 16127.

## Materials and Methods

Prior to the actual wetland construction, a 10-month planning and evaluation process was necessary to obtain the required permits and to complete the pre-mining surveys on the wetlands that would be impacted by mining. As an example of cooperation between the public and private sector, the project involved the Pennsylvania Department of Environmental Resources, the Pennsylvania Fish Commission, the Pennsylvania Game Commission, the U.S. Fish and Wildlife Service, the U.S. Soil Conservation Service, the U.S. Army Corps of Engineers, Lawrence County Conservation District, the Lawrence County Planning Commission, and the Western Pennsylvania Conservancy.

Approximately 22.4 ha (56 acres) of wetlands were established on 2 mine sites in northwestern Pennsylvania in 1987 as part of a mitigation and reclamation plan developed by Adobe Mining, Inc. On the site, hereafter referred to as the Guarnieri Mine, 7 palustrine wetlands and a deep water habitat totaling 19.2 ha were established on a 73.6 ha mine site to replace approximately 4.8 ha (4:1 ratio) of palustrine wetlands on an adjacent active mine. The palustrine wetlands consisted of a sedimentation pond converted to a wetland, a deep water habitat, and 6 other areas constructed during site reclamation. Water level control devices were installed on 64 of the palustrine wetlands, the sediment pond, and the deep water habitat for the management of aquatic vegetation. All water courses were diverted into the wetlands complex. Four of the 6 wetlands were designed with emergency spillways to allow flood waters to flow into the deep water habitat, which could then overflow into the sediment pond. This drained into a natural wetlands complex owned and managed by the Pennsylvania Game Commission. A 0.2 ha island was constructed in the center of the deep water habitat to provide additional waterfowl nesting habitat.

On a second mine, the Edwards Mine, 3 palustrine wetlands totaling 3.2 ha were constructed to replace a 1.8 ha palustrine wetland (1.8:1 ratio) destroyed during mining of the site. As for the Guarnieri Mine, 1 wetland was a converted sedimentation basin and 2 were constructed during reclamation. All water courses were diverted into these wetlands so that adequate water levels could be maintained. One 1.4 ha palustrine wetland was constructed with irregular shorelines along with 5 islands to increase the amount of edge, thereby providing additional feeding and nesting areas for waterfowl. On both mines, non-wetland areas were planted with trees and shrubs interspersed among the grassland (grasses and legumes) to provide additional habitats for upland wildlife species.

Two different procedures were used for the construction and establishment of wetlands on these sites. On the Guarnieri Mine, 2 wetland sites received soils and sediment material salvaged from natural palustrine wetlands that were destroyed by mining. Once the upper 60 cm of material was removed and stockpiled at the site, the lower clay and muck deposits were removed, transported, and spread in the newly constructed wetland. Within 5 working days, the construction and transportation phase of the project was completed and water was released into the sites. These areas did not receive supplemental planting of wetland species. This project was scheduled to be completed either in the late fall of 1985 or early spring 1986, but due to the excessive delays in the permitting process, it was not completed until late April 1986.

The remaining 6 wetlands received a total of 18,150 individual plants, comprising 10 different species, plus 23 kg of coontail (Ceratophyllum demersum) and 218 kg of grass and legume seeds. In addition, a total of 6,950 trees and shrubs representing 17 species were planted either adjacent to the wetlands or in upland areas, depending on site conditions.

Prior to mining on the Edwards Mine, soils and sediment were removed from a natural palustrine wetland and transported to a 1.4 ha wetland constructed on the same site. The same procedure was followed as at the Guarnieri Mine for the removal, storage and transportation of material between the natural and man-made system, except that scrapers rather than trucks were used to transport the material. Due to intermittent rainfall in the middle of August when this project was initiated, water from the active pit (pH 7.0-7.2) was pumped into the site as soon as all material was in place. The time lapse from the start of the project until there was sufficient water to maintain the system was approximately 10 days. A total of 6,100 individual wetland plants, 11 kg of coontail, plus 7,450 trees and shrubs and 314 kg of seeds representing 37 different species have been planted on the site. The total cost of the establishment of vegetation at this entire 3.2 ha wetland complex was $5,488 for material and $1,738 for labor. On both mines, the cost of construction, removal and transportation of wetland material were considered part of the overall mining and reclamation plan of these sites. The company indicated that the total cost of these wetland development was by current regulations.

To provide additional nesting sites for wildlife, 6 and 4 wood duck (Aix sponsa) and 17 and 7 blue bird (Sialia sialis) nest boxes were placed on the Guarnieri and Edwards Mines, respectively.

Vegetation and Wildlife Assessments

To assess the response of natural revegetation of wetland species on the Guarnieri Mine, $22m^2$ quadrats were established on 5 wetlands, 3 received supplemental planting and 2 which received wetland soils and plant material from the adjacent mine site and $12m^2$ quadrats were established on the Edwards Mine. Each mine was visited at least once a week and the numbers of different species of wildlife observed were recorded. In addition, 15m small mammal snap trap transects with traps set at 3m intervals were established on each site. Trapping was conducted for 3 nights and animal capture rate was expressed as number of animals/100 trap nights.

## Results and Discussions

Response of Wetland Vegetation

On both mines, there was good survival and growth of vegetation on both the relocated wetlands as well as those receiving supplemental plantings of wetland species. Of the 833 individual plants found on the quadrats 752 or 90% were volunteer species. In open water areas Nitella spp. and Lemna minor had a density of approximately $15/cm^2$, respectively. Nineteen of the 24 species identified on these wetlands were the result of volunteer invasions and/or from seeds or root stocks present in the material transported from the adjacent mine site. The number of species/$m^2$ and the density of plants/$m^2$ on those wetlands that received the transplanted wetland material was similar to those receiving supplemental plantings. Between 8 and 13 different species occurred on the individual quadrats receiving supplemental plantings compared to the 8 or 9 species that occurred on the transplanted sites. The average number of plants was 36 (32-41) and 38 (23-53) individuals/$m^2$ for areas that received supplemental plantings and those that received wetland soils, respectively. The similar number of species and density of vegetation on wetlands with supplemental plantings compared with those that received only wetland soils indicate that both procedures may be used to establish productive wetlands of mine sites. The principal species occurring on all wetlands on the Guarnieri Mine included Echinocloa muricata, Eleocharis obtusa, Bidens frondosa, Pancium capillare and Polygonum sagittatum (Table 3). Sensitive fern (Onoclea sensibilis), skunk cabbage (Symplocarpus foetidus), along with rushes (Juncus ssp.) and sedges (Carex ssp.) were observed on the transplanted wetlands, but did not occur on the randomly selected quadrats. Throughout the transplanted wetlands, numerous dogwoods (Cornus spp.), black willow (Salix nigra), and buttonbush (Cephalanthus occidentalis) were observed sprouting from root stocks present in the soil. On the wetlands that received supplemental plantings of wetland species, the species in the shallow and intermediate depth zone became established within 2 months of planting and those in the deep zone became established the following spring.

On the Edwards Mine, the response of vegetation on the site that received the transplanted material was not as rapid as it was on the Guarnieri Mine. Natural revegetation of wetland plants did not occur until the following spring and summer months. However, the response of those species that did regenerate naturally as well as the supplemental species in the shallow water zone showed excellent survival and growth.

In contrast to the Guarnier Mine, only 9 species were present on the 5 quadrats and of those 2 species, Trifoloim pratense and Trifolium repens, probably invaded from adjacent upland areas. A total of 127 individual plants were found on this wetland, and of these, 40% were volunteers. Between 4 and 6 species were on the individual quadrats and the average density was 25 (22-29) individuals/$m^2$.

This wetland has a dense grass and legume cover immediately adjacent to the relocated wetland. Dense cover along with elevated water levels over the transplanted wetland materials could be factors in the reduction in the number of volunteer species that invaded this wetland.

Table 2. Comparison of total number of species, feeding guilds, population status, breeding status and abundance of avian fauna observed on constructed wetlands on two mine sites compared with those observed on natural wetland prior to their removal.

| LOCATION | TOTAL SPECIES | TOTAL FEEDING GUILDS | POPULATION STATUS C | V | T | E | TOTAL POINTS | BREEDING STATUS * | ** | M | ABUNDANCE 1 | 2 | 3 | 4 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Guarnieri | 57 | 18 | 50 | 3 | 1 | 3 | 1550 | 47 | 3 | 7 | 17 | 9 | 15 | 16 |
| Edwards | 37 | 14 | 34 | 3 | - | - | 640 | 35 | - | 2 | 11 | 6 | 10 | 10 |
| Total | 60 | 19 | 53 | 3 | 1 | 3 | 2190 | 48 | 3 | 9 | - | - | - | - |
| Private Mining | 19 | 11 | 18 | 1 | - | - | 280 | 19 | - | - | 1 | 1 | 2 | 15 |

Classification same as that used in Table 1. C = 10 points, V = 100 points, T = 150 points, E = 200 points.

To provide additional nesting sites for wildlife, 6 and 4 wood duck (Aix sponsa) and 17 and 7 blue bird (Sialia sialis) nest boxes were placed on the Guarnieri and Edwards Mines, respectively.

Vegetation and Wildlife Assessments

To assess the response of natural revegetation of wetland species on the Guarnieri Mine, 22 $m^2$ quadrats were established on 5 wetlands, 3 received supplemental planting and 2 which received wetland soils and plant material from the adjacent mine site and 12 $m^2$ quadrats were established on the Edwards Mine. Each mine was visited at least once a week and the numbers of different species of wildlife observed were recorded. In addition, 15m small mammal snap trap transects with traps set at 3m intervals were established on each site. Trapping was conducted for 3 nights and animal capture rate was expressed as number of animals/100 trap nights.

## Results and Discussions

### Response of Wetland Vegetation

On both mines, there was good survival and growth of vegetation on both the relocated wetlands as well as those receiving supplemental plantings of wetland species. Of the 833 individual plants found on the quadrats 752 or 90% were volunteer species. In open water areas Nitella spp. and Lemna minor had a density of approximately 15/$cm^2$, respectively. Nineteen of the 24 species identified on these wetlands were the result of volunteer invasions and/or from seeds or root stocks present in the material transported from the adjacent mine site. The number of species/$m^2$ and the density of plants/$m^2$ on those wetlands that received the transplanted wetland material was similar to those receiving supplemental plantings. Between 8 and 13 different species occurred on the individual quadrats receiving supplemental plantings compared to the 8 or 9 species that occurred on the transplanted sites. The average number of plants was 36 (32-41) and 38 (23-53) individuals/$m^2$ for areas that received supplemental plantings and those that received wetland soils, respectively. The similar number of species and density of vegetation on wetlands with supplemental plantings compared with those that received only wetland soils indicate that both procedures may be used to establish productive wetlands of mine sites. The principal species occurring on all wetlands on the Guarnieri Mine included Echinocloa muricata, Eleocharis obtusa, Bidens frondosa, Pancium capillare and Polygonum sagittatum (Table 3). Sensitive fern (Onoclea sensibilis), skunk cabbage (Symplocarpus foetidus), along with rushes (Juncus ssp.) and sedges (Carex ssp.) were observed on the transplanted wetlands, but did not occur on the randomly selected quadrats. Throughout the transplanted wetlands, numerous dogwoods (Cornus spp.), black willow (Salix nigra), and buttonbush (Cephalanthus occidentalis) were observed sprouting from root stocks present in the soil. On the wetlands that received supplemental plantings of wetland species, the species in the shallow and intermediate depth zones became established within 2 months of planting and those in the deep zone become established the following spring.

On the Edwards Mine, the response of vegetation on the site that received the transplanted material was not as rapid as it was on the Guarnieri Mine. Natural revegetation of wetland plants did not occur until the following spring and summer months. However, the response of those species that did regenerate naturally as well as the supplemental species in the shallow water zone showed excellent survival and growth.

Table 1. Species of birds, feeding guilds, populations, breeding status, and abundance of avian species observed on constructed wetlands on two surface coal mine sites in western Pennsylvania.

| SPECIES | A Guilds | B State Population Status | C Regional Breeding Status | D Abundance Guarnieri | Edwards |
|---|---|---|---|---|---|
| Ciconiiformes | | | | | |
| Great Blue Heron (Ardea herodias) | 249 | V | * | 4 | 3 |
| Green-backed Heron (Butoides striatus) | 249 | C | * | 4 | 3 |
| Anseriformes | | | | | |
| Tundra Swan (Cygnus colombianus) | 340 | C | M | 1 | |
| Canada Goose (Branta canadensis) | 340 | C | * | 4 | 4 |
| Mallard (Anas platyrhynchos) | 340 | C | * | 4 | 4 |
| Black Duck (Anas rubripes) | 340 | C | * | 3 | |
| Blue-winged Teal (Anas discors) | 340 | C | * | 3 | 2 |
| Green-winged Teal (Anas crecca) | 340 | C | * | 3 | 1 |
| Gadwall (Anas strepera) | 340 | C | * | 1 | |
| Wood duck (Aix sponsa) | 340 | C | * | 3 | 3 |
| Ring-necked Duck (Athya collaris) | 348 | C | M | 2 | |
| Greater Scaup (Athya marila) | 348 | C | M | 1 | |
| Bufflehead (Bucephala albeola) | 348 | C | M | 1 | |
| Ruddy Duck (Oxyura jamaicensis) | 348 | C | M | 1 | |
| Hooded Merganser (Lophodytes cucullatus) | 248 | C | * | 1 | |
| Podicioediformes | | | | | |
| Pied-billed Grebe (Podilymbus podiceps) | 248 | C | * | 2 | |
| Falconiformes | | | | | |
| Red-tailed Hawk (Buteo jamaicensis) | 257 | C | * | 4 | |
| American Kestrel (Falco sparverius) | 257 | C | * | 4 | |
| Northern Harrier (Circus syaneus) | 257 | V | * | 4 | 2 |
| Osprey (Pandion haliaetus) | 247 | Exp. | ** | 1 | |
| Galliformes | | | | | |
| Ringnecked Pheasant (Phasianus colchicus) | 123 | C | * | 1 | 1 |
| Gruiformes | | | | | |
| American Coot (Fulica americana) | 248 | C | * | 2 | |
| King Rail (Rallus elegans) | 261 | E | ** | 2 | |
| Charadriiformes | | | | | |
| Herring Gull (Larus argentatus) | 349 | C | M | | 3 |
| Killdeer (Charadrius vociferus) | 223 | C | * | 4 | 4 |
| Spotted Sandpiper (Actitus macularia) | 261 | C | * | 3 | 2 |
| Upland Sandpiper (Bartramia longicauda) | 223 | T | * | 1 | |
| American Woodcock (Philohela minor) | 221 | C | * | 1 | 1 |
| Willet (Catoptrophorus semipalmatus) | 261 | C | * | 1 | |
| Greater Yellowlegs (Tringa melanoleuca) | 261 | C | M | 3 | 3 |
| Lesser Yellowlegs (Tronga flavipes) | 261 | C | M | | 1 |
| Columbiformes | | | | | |
| Morning Dove (Zenaida macroura) | 123 | C | * | 3 | 3 |
| Strigiformes | | | | | |
| Great Horned Owl (Bubo virginianus) | 237 | C | * | 4 | |

| | | | | | |
|---|---|---|---|---|---|
| Short-eared Owl (Asio flammeus) | 237 | E | ** | 2 | |
| **Coraciiformes** | | | | | |
| Belted Kingfisher (Ceryle alcyon) | 248 | C | * | 3 | |
| **Piciformes** | | | | | |
| Yellow Shafted Flicker (Colaptes auratus) | 211 | C | * | 3 | 1 |
| Downy Woodpecker (Picoideus pubescens) | 211 | C | * | 2 | |
| Hairy Woodpecker (Picoideus villosus) | 211 | C | * | 3 | 1 |
| **Passeriformes** | | | | | |
| Horned Lark (Eremophila aipestris) | 223 | C | * | | 3 |
| Tree Swallow (Tridoprocne bicolor) | 237 | C | * | 4 | 4 |
| Barn Swallow (Hirundo rustica) | 237 | C | * | 1 | 1 |
| American Crow (Corvus brachyrhynchos) | 334 | C | * | 4 | 4 |
| Blue Jay (Cyanocitta cristata) | 334 | C | * | 4 | 3 |
| Black Capped Chickadee (Parvus atricapillus) | 323 | C | * | 2 | 2 |
| American Robin (Turdus migratorius) | 323 | C | * | 4 | 4 |
| Chestnut-sided Warbler (Dendroica pensylvanica) | 234 | C | * | 1 | |
| Ruby-crowned Kinglet (Regulis calendula) | 232 | C | M | 1 | |
| Red-winged Blackbird (Agelaius phoeniceus) | 334 | C | * | 4 | 4 |
| Common Grackle (Quiscalus quiscula) | 323 | C | * | 3 | 3 |
| Eastern Meadow Lark (Sturnella magna) | 323 | C | * | 4 | 4 |
| Bobolink (Dolichonyx oryzivorus) | 234 | C | * | 3 | 1 |
| Slate-colored Junco (Junco hyemalis) | 334 | C | * | 3 | 4 |
| Cardinal (Cardinalis cardinalis) | 134 | C | * | 1 | 4 |
| Rufous Sided Towhee (Pipilo erythrophthalmus) | 323 | C | * | 1 | |
| Chipping Sparrow (Spizella passerina) | 334 | C | * | 2 | 2 |
| Tree Sparrow (Spizella arborea) | 237 | C | * | 3 | 1 |
| Vesper Sparrow (Pooecetes gramineus) | 334 | V | * | 2 | 1 |
| Song Sparrow (Melospiza melodia) | 334 | C | * | 3 | 4 |
| Swamp Sparrow (Melospiza georgiana) | 334 | C | * | 4 | 3 |
| Savannah Sparrow (Passerculus sandwichensis) | 334 | C | * | 1 | 1 |

Species names after American Orinthologist's Union (1983).

A Feeding guilds are characterized by a 3-digit (ABC) number where; A = primary food habits (1) granivore, (2) carnivore (e.g., insectivore, piscivore), or (3) omnivore; B = foraging stratus (1) bark, (2) ground, (3) foliage, (4) water, (5) ground and air, or (6) mud; and C = foraging behavior (0) dip, (1) probe, (2) bark glean, (3) ground glean, (4) foliage glean, (5) sally, (6) friut or buds, (7) sustained flight, (8) dive, or (9) hunt (modified from Willson 1974).

B Classification based on Gill (1985), V = vulnerable; T = threatened; E = endangered; C = common (unclassified).

C Based on Leberman (1988), *Breeding species; ** Probable breeder, M = Migrant.

D 4 = observed 75-100% visits; 3 = observed 50-75% of visits; 2 = observed 25-30% of visits; 1 = observed 25% or less of visits.

Wildlife Use

Sixty species of birds (Table 1) representing 23 families, including 7 species of birds of special concern in Pennsylvania (Genoways and Brenner 1985) were observed using these wetlands and adjacent uplands on both mine sites during the first 2 years. Of these, 57 species representing 18 feeding guilds were observed on the Edwards Mine (Table 2). Three of the 7 species of special concern were listed as endangered species and included the King Rail (Rallus elegans), the osprey (Pandion haliaetus), and the short-eared owl (Asia flammeus). According to Gill (1985), the osprey is considered to be an extirpated species in western Pennsylvania. Leberman (1988) indicated that it might be an occasional breeder, but confirmed osprey nesting had not been documented in western Pennsylvania and only reintroduced individuals have bred in eastern Pennsylvania. Five broods of Canada geese (Branta canadensis), and 3 mallard (Anas platyrhnchos)broods were observed in both years using the Guarnieri Mine. Two and 4 wood duck broods (Aix sponsa) were observed using this wetland complex the first and second year, respectively. On separate visits during the summer months between 100 and 300 Canada geese and flocks of over 100 ducks, including mallards, black ducks (Anas rubripes), blue-winged teal (Anas discors), green-winged teal (Anas crecca), and wood ducks were observed feeding on these wetlands and adjacent upland areas. In addition, approximately 20 spotted sandpipers (Actitic macularia), 3 great blue herons (Ardea herodias), and 2 green-backed herons (Butorides striatus) were observed on each visit feeding along the shoreline and the shallow areas of this wetland complex. Two and 4 of the 6 wood duck nesting boxes were used by nesting females producing 16 and 30 fledglings the first and second year, respectively. Thirteen of the 17 bluebird boxes were used by tree swallows (Tridoprocne bicolor) the first year and all boxes were used by nesting tree swallows the second year, producing 66 and 88 fledglings, respectively. In addition 2 of the wood duck nesting boxes were occupied by nesting American Kestrels (Falco sparverius).

The use of wetlands on the Edwards Mine by avian fauna was not as great as the Guarnieri site. A total of 37 different species representing 14 different guilds were observed using this wetland during the year. Of these 3 were listed as vulnerable species by Gill (1985) and are therefore considered as species of special concern in Pennsylvania. The first year, a Canada goose nesting on one of the islands and the second year, 3 broods of Canada geese along with 3 mallard broods were produced on these wetlands. In the first year, 4 of the 7 bluebird boxes were used by nesting tree swallows. In the second year all 7 of these boxes were occupied by nesting tree swallows producing 23 and 35 fledglings, respectively. In addition to the wetland species, 3 breeding pairs of bobolinks (Dolichonyx oryzivorus), a species of special concern in Pennsylvania, were observed nesting on surrounding grasslands on both sites.

Based on the total number of species and the amount of points assigned as to whether the species were common in Pennsylvania or a species of special concern, the avian fauna on the Guarnieri Mine (Table 2). The avian fauna used at both mines, however, was greater than the 19 species observed using the wetlands that were destroyed during mining and the northern harrier (Circus syaneus)was the only species of special concern observed on these sites. The number of total points accumulated by the avian fauna use of the wetlands destroyed by mining was 240 which was considerably less than that accumulated by the avian fauna using either of the 2 mitigated wetland systems.

These wetlands were also used by diverse mammalian species (17) used the wetlands on the Guarnieri Mine site and 10 used the Edwards Mine site (Table 3). Based on a snap trap transects on both mine sites, the total number of small mammals/100 trap nights was 19 on the Guarnieri site and 45 per 100 trap nights on the Edwards site. The 3 species trapped on both mine sites included the meadow mole (Microtus pennsylvanicus), the white-footed deer mouse (Peromyscus leucopus), and the short-tailed shrew (Blarina brevicauda) (Table 4). The greater number of captures on the Edwards Mine was probably due to the dense grass and legume cover which surrounded the wetland complex. The number of captures on both wetland complexes, however, either equalled or exceeded that present on the wetlands that were removed during mining (Table 4).

A total of 8 and 5 species of amphibians were observed using the Edwards and Guarnieri sites, respectively (Table 5). Of these, the most common species included the green frog (Rana clamitans) and the red spotted newt (Notophthalmus viridescens). On the Guarnieri mine, the leopard frog (Rana pipiens)and the pickerl frog (Rana palustris) were found in large numbers on those wetlands that were completely surrounded by grasslands. Snakes were the most prevalent reptiles on both mine sites with the eastern garter snake (Thamnophis sirtalis), black racer (Coluber constrictor), and water snake (Nerodia sipedon)being the most common species encountered during site visits.

Table 3. Species of mammals observed using constructed wetlands on two surface coal mine sites in western Pennsylvania.

| MAMMALS | GUARNIERI | EDWARDS |
|---|---|---|
| Virginia Opossum (*Didelphis virginianus*) | X | X |
| Star-nosed Mole (*Condylura cristata*) | X | X |
| Masked Shrew (*Sorex cinereus*) | X | X |
| Short-tailed Shrew (*Blarina brevicauda*) | X | X |
| White-footed Deer Mouse (*Peromyscus leucopus*) | X | X |
| Eastern Chipmunk (*Tamias striatus*) | X | |
| Meadow Jumping Mouse (*Zapus hudsonius*) | X | X |
| Meadow Vole (*Microtus pennsylvanicus*) | X | X |
| Muskrat (*Ondatra zibethicus*) | X | |
| Woodchuck (*Marmota monax*) | X | |
| Eastern Cottontail Rabbit (*Sylvilagus floridanus*) | X | X |
| Raccoon (*Procyon lotor*) | X | |
| Long-tailed Weasel (*Mustela frenata*) | X | |
| Mink (*Mustela vison*) | X | |
| Red Fox (*Vulpes vulpes*) | X | X |
| Grey Fox (*Urocyon cinereoargenteus*) | X | X |
| White-tailed Deer (*Odocoileus virginianus*) | X | X |
| TOTAL | 17 | 10 |

Classification based on Jones et al. 1982.

Table 4. Comparison of the number of small mammals trapped on constructed wetlands on two mine sites in western Pennsylvania compared with those removed during mining.

| LOCATION | *Microtus pennsylvanicus* N/100 Trap Nights | *Peromyscus leucopus* | *Blarina brevicauda* |
|---|---|---|---|
| Guarnieri | 11.3± | 2.5± | 5.0- |
| Edwards | 25.0± | 10.0± | 10.0- |
| Wetlands removed during mining | 4.0± | 3.0± | 5.0- |

Table 5. Species of amphibians and reptiles observed using constructed wetlands on two surface coal mine sites in western Pennsylvania.

| | GUARNIERI | EDWARDS |
|---|---|---|
| AMPHIBIANS | | |
| American Toad (*Bufo americanus*) | X | X |
| Fowler's Toad (*Bufo woodhousei fowleri*) | X | X |
| Green Frog (*Rana clamitans*) | X | X |
| Leopard Frog (*Rana pipiens*) | X | |
| Pickerel Frog (*Rana palustris*) | X | |
| Wood Frog (*Rana sylvatica*) | X | |
| Spring Peeper (*Hyla crucifer*) | X | X |
| Red-spotted Newt (*Notophthalmus viridescens*) | X | X |
| TOTAL | 8 | 5 |
| REPTILES | | |
| Black Rat Snake (*Elophe obsoleta*) | X | X |
| Eastern Garter Snake (*Thamnophilis sirtalis*) | X | X |
| Eastern Milk Snake (*Lampropeltis doliata*) | X | X |
| Black Racer (*Coluber constrictor*) | X | X |
| WaterSnake (*Nerodia sipedon*) | X | X |

| | | |
|---|---|---|
| Painted Turtle (Chrysemys picta) | X | X |
| Snapping Turtle (Chelydra serpentina) | X | |
| TOTAL | 7 | 6 |

Classification after McCoy (1982).

## Summary

Based on these results, it is apparent that wetlands can be constructed and mitigated on mine lands which will have value for wildlife as well as providing other functions including sediment and erosion control (Brenner 1987).

The similarity in the number of species and the density of vegetation on wetlands with supplemental planting compared with those that received only wetland soils indicate that both procedures may be used to establish productive wetlands on mine sites. But if the objectives is to develop a wetland with identical species composition as the one being mined, then the removal and transportation of wetland soils to the constructed wetland would be the preferred procedure. Regardless of the procedure, the cost of wetland establishment is approximately the same as if the land is returned to AOC as required by current regulations. The rapid establishment of vegetation on these sites as well as the multiple use by wildlife indicates that the construction of wetlands on mine sites should be given increased consideration by both the coal industry and the regulatory agencies alike.

## Literature Cited


American Orinthologists' Union. 1983. Checklist of North American Birds. 6th Edition. Allen Press, Lawrence, Kansas 877 pp.

Brenner, F.J. 1973. Evaluation of abandoned strip mines as fish and wildlife habitats. Trans. N.E. Wildlife Conf. 30:205-229.

Brenner, F.J. 1983. Environmental aspects of coal production in Pennsylvania: A guide to reclamation. pp. 405-422. IN S.K. Majumdar and E.W. Miller (eds.). Pennsylvania Coal: Resources, Technology and Utilization. PA Acad. Sci.

Brenner, F.J. 1984. Restoration of natural ecosystems on surface coal mines in the northeastern United States. pp. 211-225. IN T.N. Vezroglu (ed.). The Bioshpere: Problems and Solutions. Elsevier, Amsterdam.

Brenner, F.J. 1985. Aquatic and terrestrial habitats in Pennsylvania. pp. 7-18. IN H.H. Genoways and F.J. Brenner (eds.). Species of Special Concern in Pennsylvania. Spec. Publ. No. 14, Carnegie Mus. Nat. Hist. Pittsburgh, PA.

Brenner, F.J. 1986. Habitat preservation and development for rare and endangered species management. pp. 37-52. IN S.K. Majumdar, F.J. Brenner and A.F. Rhoads (eds.). Endangered and Threatened Species Programs in Pennsylvania and Other States: Causes, Issues and Management. PA Acad. Sci.

Brenner, F.J. 1987. Surface Coal Mine Reclamation: An opportunity for wetland development. pp. 338-344. Increasing Our Wetland Resources.

Brenner, F.J. and J.J. Mondok. 1978. Waterfowl nesting rafts designed for fluctuating water levels. J. Wildl. Manage. 43:978-982.

Brenner, F.J. and J. Kelly. 1981. Characteristics of bird communities on surface mined lands in Pennsylvania. Environmental Manage. 5:441-449.

Brenner, F.J. and R.P. Steiner. 1987. Alternative reclamation strategies for mined lands. pp. 112-129. IN Environmental Consequences of Energy Production: Problems and Prospects. PA Acad. Sci.

Brooks, R.P., J.B. Hill, F.J. Brenner, S. Capets. 1985. Wildlife use of coal mines in western Pennsylvania. pp.327-352. IN R.P. Brooks, D.E. Samuel, J.B. Hill (eds.). Wetlands and Water Management on Mined Lands. Pennsylvania State University, University Park.

Cable, T.T., V. Black, Jr., V.R. Holmes, 1989. Simplified method for wetland habitat assessment. Environmental Manage. 13: 207-215.

Fowler, D.K., D.M. Hill , and L.J. Fowler. 1985. Colonization of coal surface mines in western Pennsylvania. pp. 327-352. IN R.P. Brooks, D.E. Samuel, J.B. Hill (eds.). Wetlands and Water Management on Mined Lands. Pennsylvania State University. 393 pp.

Genoways, H.H. and F.J. Brenner. 1985. Species of Special Concern in Pennsylvania. Spec. Publ. No. 14., Carnegie Mus. Nat. Hist. Pittsburgh, PA. 430 pp.

Gill, F.B. 1985. Birds IN H.H. Genoways and F.J. Brenner (eds.). Species of Special Concern in Pennsylvania. Spec. Publ. No. 14. Carnegie Mus. Nat. Hist. 430 pp.

Glooschenko, V., J.H. Archbold, and D. Herman. 1988. The Ontario wetland evaluation system: Replacability and bird habitat selection. pp. 115-127. IN The Ecology and Management of Wetlands. D.D. Hook et al. (eds.). Vol. II. Timber Press. Portland, Oregon. 394 pp.

Jones, D.W., M.J. McElliogott, and R.H. Mannz. 1985. Biological, chemical and morphological characterization of 33 surface mine lakes in Illinois and Missouri. Peabody Coal Company. St. Louis, Mo. 243 pp.

Jones, J.K., D.C. Carter, H.H. Genoways, R.S. Hoffman, and D.W. Rice. 1982. Revised checklist of North American mammals north of Mexico. 1982. Occasional Pap. Mus. Texas Tech. Univ. No. 80 22 pp.

Leberman, R.C. 1988. A fieldlist of the birds of western Pennsylvania and adjacent regions. Carnegie Mus. Nat. Hist. Spec. Publ. 13. 51 pp.

McCoy, C.J. 1982. Amphibians and Reptiles in Pennsylvania. Spec. Publ. No. 6. Carnegie Mus. Nat. Hist. 91 pp.

Sandusky, J.E. 1978. The potential for management of waterfowl nesting habitats on reclaimed lands. pp. 325-327. IN D.E. Samuel, J.R. Stauffer, C.H. Hocutt, and W.H. Masons (Eds.). Surface Mining and Fish/Wildlife Needs in the Eastern United States. Fish and Wildlife Service/Office of Biological Service. 78/81 386 pp.

Thompson, C.S. 1984. Experimental practices in surface coal mining-creating wetland habitat. Natural Wetlands Newsletter. Mar-April, 1984. pp. 15-16.

Tiner, R.W., Jr. 1987. Mid-Atlantic wetlands - a disappearing natural treasure. U.S. Department of the Interior Fish and Wildlife Service, New Cormer, Mass. and U.S. Environmental Protection Agency, Philadelphia, PA. 27pp.

Vogel, W.G. 1981. A guide for revegetating coal mine soils in the eastern United States. U.S. Department of Agriculture Forest Service General Technical Report. NE-66 198 pp.

Willson, M.F. 1974. Avian community organization and habitat structure. Ecology. 55:1017-1028-9.

# Wetlands - U.S. Bureau of Mines

This page intentionally left blank.

# SIZING AND PERFORMANCE OF CONSTRUCTED WETLANDS: CASE STUDIES[1]

Robert S. Hedin and Robert W. Nairn[2]

Abstract: Iron removal in three Pennsylvania wetlands constructed to treat acid mine drainage was evaluated. All wetlands were constructed with a mushroom compost substrate and were planted with Typha spp. Performance was evaluated by calculating area-adjusted iron loadings and removals as Fe g $day^{-1}m^{-2}$ (gdm). An initial model of the iron removal capabilities of constructed wetlands was also evaluated. Relationships between pH, concentration, flow, loading and removal were considered. At the Somerset Site (avg. influent pH=4.0), influent iron loading, which was primarily a function of concentration, was considerably more variable than iron removal. The relationship between the two appeared asymptotic, with removal being independent at loadings above 15 gdm (avg. removal = 10.6 gdm) and averaging 54% of loading at loadings less than 15 gdm. Conversely, at the Latrobe Site (avg. influent pH=3.0), variation in iron loading was primarily a consequence of flow variation. Removal averaged 2.7 gdm at flows > 100 L $min^{-1}$ and 4.3 gdm when flow < 100 L $min^{-1}$. The overall average removal was 3.6 gdm. A significant relationship between loading and removal was not found. At the Friendship Hill Wetland, influent pH was 2.7, and iron removal averaged 3.3 gdm. The narrow range of loadings at this site prevented detailed analysis of loading:removal relationships. Overall, these data were used to develop preliminary wetland sizing criteria based upon iron loadings. In situations where mine drainage has flow > 50 L $min^{-1}$ and iron concentration > 50 mg $L^{-1}$, loading-based criteria result in significantly larger wetlands than conventional flow-based criteria.

## Introduction

Constructed wetland technology has advanced considerably in recent years, but sizing criteria have evolved very little from initial suggestions that 5.0-15.0 $m^2$ of wetland were needed for each liter per minute (L $min^{-1}$) of contaminated flow (Kleinmann et al. 1986, Girts et al. 1987). Although this sizing standard was only intended for drainage with flow less than 40 L $min^{-1}$, pH greater than 4 and iron concentration less than 50 mg $L^{-1}$, it has been applied to systems that receive much higher flows and poorer water quality. Many of these systems do not perform satisfactorily, probably in part because they are undersized. A better sizing standard that incorporates contaminant concentrations as well as flow is needed.

The belief that a "correct" sizing methodology exists derives from an assumption that a square meter of wetland can remove a predictable amount of contaminant in a day's operation. This sizing hypothesis can be evaluated empirically by analyzing iron-removal by existing constructed

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]Robert S. Hedin, and Robert W. Nairn, are Research Ecologist and Research Biologist, at the U.S. Bureau of Mines, Pittsburgh Research Center, Pittsburgh, PA.

wetlands. In this paper, the iron-removal capabilities of three constructed wetland systems are evaluated. We focus on iron because it is the principal metal contaminant in coal mine drainage and appears, thus far, to be most suited to wetland treatment.

Evaluation of Wetland Performance

To make reliable evaluations of wetland performance, a measure should be used that allows comparison of contaminant removal for systems of different sizes that receive drainage with different flow rates and chemical compositions. In the past, concentration efficiency (CE%) has been a common measure of performance (Girts et al. 1987, Stark et al. 1988). For iron (Fe) the calculation is:

$$CE\% = \frac{Fe\ (mg\ L^{-1})_{in} - Fe\ (mg\ L^{-1})_{out}}{Fe\ (mg\ L^{-1})_{in}} \times 100 \quad (1)$$

In many cases, this is a poor measure of performance. Comparisons of iron removal efficiencies for different wetland systems fail to provide sizing insights because the calculations do not include any measure of contaminant masses or wetland size.

A better evaluation procedure involves several components. First, the daily mass of iron at each sampling station must be calculated:

$$Fe\ (g\ day^{-1}) =$$

$$1.44 \times flow\ (L\ min^{-1}) \times Fe\ (mg\ L^{-1}) \quad (2)$$

(where 1.44 is the adjustment factor needed to convert minutes to days [1,440] and milligrams to grams [0.001]). This value, grams of iron per day, is commonly referred to as the iron "loading." The amount of iron removed by the wetland between two sampling stations is calculated by comparing iron loadings at the two points.

$$Fe\ (g\ day^{-1})_{rem} =$$

$$Fe\ (g\ day^{-1})_{in} - Fe\ (g\ day^{-1})_{out} \quad (3)$$

Conventionally, a comparison of the influent and the final effluent station is made to determine the effect of the whole wetland on iron loading. However, when additional water samples are collected, the iron removed by individual cells can be calculated.

The second calculation involves adjusting the iron loading for wetland size. For inlet stations, the amount of the daily iron load apportioned to each square meter of downflow wetland can be estimated by:

$$Fe\ (g\ day^{-1}\ m^{-2}) = \frac{Fe\ (g\ day^{-1})_{in}}{area\ (m^2)} \quad (4)$$

An estimate of the amount of iron removed by each square meter of wetland between two sampling points can be calculated:

$$Fe\ (g\ day^{-1}\ m^{-2})_{rem} =$$

$$\frac{Fe\ (g\ day^{-1})_{in} - Fe\ (g\ day^{-1})_{out}}{area\ (m^2)} \quad (5)$$

These values will be referred to as "area-adjusted." The units, grams of iron per day per $m^2$ of wetland, will be referred to as "gdm." We will focus in this paper on 1) area-adjusted iron loading, which is the average amount of iron that each square meter of wetland is exposed to in one day, and 2) area-adjusted iron removal, which is the average amount of iron removed by each square meter of wetland per day.

To illustrate the use of iron loading and area-adjusted iron loading estimates, consider the hypothetical data presented in Table 1. In Systems A and B, changes in iron concentrations are the same (60 mg $L^{-1}$), but because system B receives four times more flow and thus higher iron loading, it actually removes four times more iron from the water (see rem in g $day^{-1}$). The concentration efficiencies of the two wetlands are equivalent, but masses of iron removed are quite different.

Hypothetical data are shown for System C for three sampling dates on which flow rates and influent iron concentrations vary. On the first date (C1), the wetland removes all of the iron that it receives. On the next two dates (C2 and C3), iron loadings are higher and the wetland effluent contains iron. From an efficiency standpoint, performance is best on the first date and worst on the third date. But from an iron-removal perspective, the system is removing the least amount of iron on the first date. On the second and third dates the wetland removes similar amounts of iron (2880 and 3024 g $day^{-1}$). Variation in effluent chemistry results, not from changes in the wetland's iron-removal performance, but from variation in influent iron loading.

Table 1.-- Hypothetical Wetland Data and Performance

| wet-land | size | flow | Fe (mg $L^{-1}$) | | | Fe (g $day^{-1}$) | | | Fe (g $day^{-1}$ $m^{-2}$) | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | $m^2$ | L $min^{-1}$ | in | out | %rem | in | out | rem | in | out | rem |
| A | 400 | 10 | 100 | 40 | 60 | 1440 | 576 | 864 | 3.6 | 1.4 | 2.2 |
| B | 400 | 40 | 100 | 40 | 60 | 5760 | 2304 | 3456 | 14.4 | 5.8 | 8.6 |
| C1 | 500 | 30 | 40 | <1 | 99 | 1728 | <40 | 1688 | 3.5 | <.1 | 3.4 |
| C2 | 500 | 80 | 35 | 10 | 71 | 4032 | 1152 | 2880 | 8.1 | 2.3 | 5.8 |
| C3 | 500 | 150 | 30 | 16 | 47 | 6480 | 3456 | 3024 | 13.0 | 6.9 | 6.1 |
| D | 750 | 50 | 100 | 25 | 75 | 7200 | 1800 | 5400 | 9.6 | 2.4 | 7.2 |

Lastly, consider a comparison of wetland systems of different size. System D removes more iron than any wetland considered (5400 g $day^{-1}$), but it is also larger. One would expect, all other factors being equal, that a larger wetland would remove more iron. When wetland area is incorporated into the measure by calculating area-adjusted iron removal values, System B emerges as the most effective wetland considered (see removal gdm values).

Iron Removal Model

In addition to evaluating iron removal on a whole wetland basis, we analyzed the relationship between area-adjusted iron loading and removal for individual wetland cells. The objective of this analysis was to initiate the development of a model of iron removal in constructed wetlands. Figure 1 is a plot of loading and removal for a hypothetical model wetland. The shape of the plot results from an assumption that iron removal processes are limited to a maximum rate (point A"). When inflow loading is less than the maximum rate, 100% removal can occur (removal = loading). The low flow data from hypothetical wetland C1 plots on this line. At inflow loads greater than the maximum rate (point A'), constant iron removal occurs. The high flow data from wetland C (C2 and C3) fall near this line.

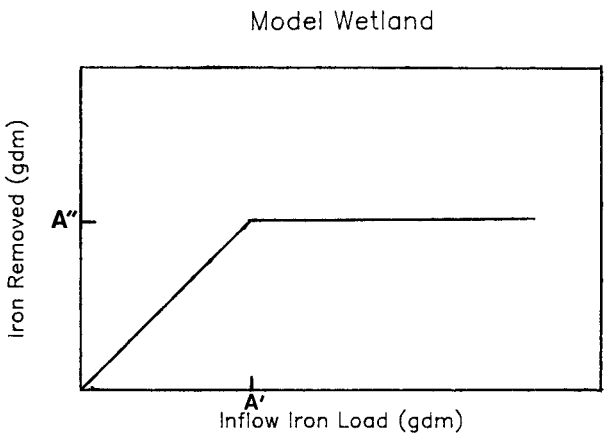



Figure 1. Hypothetical relationship between area-adjusted iron loading and area-adjusted iron removal in a constructed wetland.

One goal of this paper is to determine how accurately the performance of constructed wetlands is represented by this model. We were also interested in determining whether area-adjusted iron removal rates differed between wetlands and, if so, whether this variation could be attributed to particular characteristics of the influent water (e.g. pH, Eh, [Fe]).

Selection of Study Wetlands

Study wetlands must satisfy some basic standards if comparisons of iron removal are to provide insights into system performance. The wetlands being evaluated and compared must be built using similar designs and materials. In northern Appalachia, the "standard" wetland consists of 30-45 cm of compost substrate planted with cattails (Typha spp.). Water in the standard wetlands flows primarily by surface routes. The study systems must have distinct inlet and outlet stations where accurate flow measurements and representative samples can be collected. No significant or unquantifiable inputs of fresh or contaminated water to the wetland should exist. Data should be collected for at least one year after system construction and preferably for several years. This ensures that short term effects associated with chemical properties of the compost will not unduly bias results, and that the systems will be evaluated with an established plant cover.

The screening conditions described above exclude from analysis many wetlands that were constructed using different designs and materials (e.g. the TVA wetlands in southern Appalachia) or that do not have exact influent points. We do not intend to imply that these sites are inferior to the ones discussed. Many of these wetlands are very effective (Girts et al. 1987, Brodie et al. 1988), however, their inclusion in this paper would complicate the interpretation of the results. We leave comparisons of the effectiveness of wetlands constructed with different designs to other papers.

## Materials and Methods

The surface area of each study wetland was determined from field measurements or engineering drawings. Water flows were determined using a bucket and stop watch. All flow rates reported are the average of three to five measurements. Raw and acidified water samples were collected for chemical analysis. Samples were not filtered prior to acidification because 1) previous comparisons of filtered (0.45 um) and unfiltered samples showed negligible differences in metal concentrations, and 2) regulatory standards are based on total (unfiltered) metal concentrations. Total iron was determined on the acidified samples by ICP spectroscopy. Sample pH was measured both in the field and in the laboratory. For influent samples discussed in this paper, the results were not substantially different, therefore laboratory values are reported.

Study Sites

Somerset Wetland

The Somerset wetland, located in Somerset County, PA, treats water draining from 12-year-old surface mine spoils. It was built in 1984. Originally, seven seeps existed on the property, but a drainage system was installed that collects all drainage into a single pipe that serves as the influent to the wetland. The system consists of two cells (277 $m^2$ and 268 $m^2$) connected in series, each constructed with 30 cm of crushed limestone and 45 cm of mushroom compost. A dense growth of cattails covers both cells.

The site was first sampled in March 1987. Monthly sampling of water and flow rates was initiated in November 1988. Water samples were collected from the influent pipe, between cells, and from the effluent pipe. Flow measurements were made at the influent and effluent pipes. Loading calculations are based on the average of the inlet and outlet flow rates.

Latrobe Wetland

The Latrobe wetland, located in Westmoreland County, PA, treats water draining from both reclaimed surface spoils and an abandoned drift mine. The wetland was constructed in June 1987. Water is collected in a shallow pit and flows down a 30 m long ditch into a wetland system that consists of three rectangular cells in series (695 $m^2$, 802 $m^2$, and 1301 $m^2$). The wetland substrate consists of 10 cm of crushed limestone covered with 30-45 cm of mushroom compost. Cattails cover about 80% of wetland surface, most of the open area being due to muskrat activity during the summer of 1988.

Beginning in July 1988, water samples were collected from the influent, effluent and between each cell. Flows were measured at the influent and effluent stations beginning in October 1988. Periodic problems with leakage of water through a berm in the third cell prevented use of all the chemical and flow data collected from the final effluent station. We therefore focused our analytical efforts on the first two cells of the wetland. Loading calculations were based on influent flow rates.

In the summer of 1989, extensive modifications were made to the system. The pit, in which the seepage is collected before flowing into the wetland, was filled with spent mushroom compost. Following this modification the water flowing into the wetland had atypical chemistry (circumneutral pH, high concentrations of dissolved organics and hydrogen sulfide). Data from this period will not be presented. Most of the effects of the modification on wetland influent water chemistry appeared to stabilize in August 1989. Data for August, September and October 1989 are presented. Because of the unusual pretreatment of water at this site, these data should be interpreted with caution.

Friendship Hill Wetland

The Friendship Hill wetland was constructed by the Bureau of Mines for research purposes during the summer of 1988. It is located at the Friendship Hill National Historic Site in Fayette County, PA. Inflow water to the wetland system is drawn out of Ice Pond Run, a small first order stream that drains an abandoned drift mine about 1 km upstream of the site.

Six wetland cells were constructed. Details of the design are included in a paper by McIntire and Edenborn (1990) in these proceedings. All cells contain 15 cm of gravel covered with 45 cm of spent mushroom compost. The cells were planted with cattails in October 1988. Growth resulted in approximately 75% coverage by July 1989.

Data are presented for the summer of 1989 (first growing season) for the three wetland cells designated A2, B2 and C2. The surface areas of the cells are 104 $m^2$, 103 $m^2$, and 123 $m^2$, respectively.

While cells at the Somerset and Latrobe wetland are connected serially (cell 1 flows into cell 2), the Friendship Hill cells are parallel. Because all three cells receive water from a common source, they can be considered experimental replications. Flow rates to all cells are controlled by valves which are monitored and adjusted weekly. Water samples are collected from the common influent pool and from the effluent of each cell.

RESULTS AND DISCUSSION

Somerset Wetland

Table 2. Influent water at the Somerset Wetland.

| Date | Flow | pH | Inflow Fe | | |
|---|---|---|---|---|---|
| | L $min^{-1}$ | | mg $L^{-1}$ | g $day^{-1}$ | gdm |
| 03-20-87 | 19 | 3.2 | 193 | 5280 | 9.7 |
| 11-22-88 | 24 | 4.0 | 24 | 829 | 1.5 |
| 03-22-89 | 49 | 3.9 | 51 | 3599 | 6.6 |
| 04-11-89 | 29 | 3.7 | 122 | 5095 | 9.3 |
| 05-18-89 | 44 | 3.4 | 148 | 9377 | 17.2 |
| 06-16-89 | 25 | 3.8 | 337 | 12132 | 22.3 |
| 07-11-89 | 21 | 3.3 | 284 | 8588 | 15.8 |
| 08-16-89 | 10 | 3.3 | 310 | 4464 | 8.2 |
| 09-14-89 | 8 | 5.1 | 243 | 2799 | 5.1 |
| 10-24-89 | 11 | 5.1 | 169 | 2677 | 4.9 |
| 12-14-89 | 6 | 4.6 | 231 | 2096 | 3.9 |

Influent water at the Somerset wetland had an average pH of 4.0 and ranged from 3.2 to 5.1 (Table 2). Iron concentrations and loading varied considerably over the sampling period. Iron concentrations always decreased with flow through the first cell and, except when iron concentrations were less than 10 mg $L^{-1}$, showed further decreases with flow through the second cell (Figure 2a). Effluent iron concentrations varied considerably, ranging from 3 mg $L^{-1}$ to 159 mg $L^{-1}$.

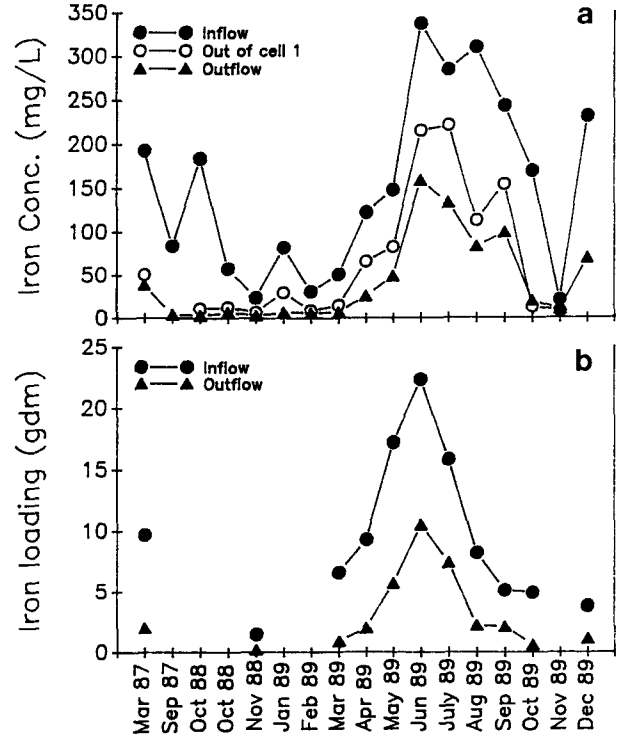



Figure 2. Iron chemistry at the Somerset wetland: a) iron concentrations at the inflow, cell one outflow and cell two outflow sampling stations; b) area-adjusted iron loading at inflow and cell two outflow stations.

Area-adjusted iron removal (the difference between influent loading and effluent loading in Figure 2b) displayed less variability than the area-adjusted loading. The average area-adjusted iron removal was 6.3 gdm (s=3.4). The lowest iron removal rates were associated with very low influent loading (e.g. Nov 1988, Dec 1989), not with any apparent failure of the wetland during periods of high loading. The highest iron removal rates, more than 11 gdm in May and June 1989, occurred when loading rates were also highest. During this period, performance of the wetland from a concentration efficiency standpoint was the lowest observed (53% in June).

Latrobe Wetland

Table 3. Influent water at the Latrobe wetland.

| Date | Flow | pH | Inflow Fe | | |
|---|---|---|---|---|---|
| | L min$^{-1}$ | | mg L$^{-1}$ | g day$^{-1}$ | gdm |
| 10-14-88 | 38 | 3.1 | 215 | 11,765 | 7.9 |
| 11-11-88 | 38 | 3.1 | 226 | 12,366 | 8.3 |
| 11-28-88 | 56 | 3.0 | 194 | 15,644 | 10.5 |
| 12-15-88 | 30 | 3.3 | 262 | 11,318 | 7.6 |
| 01-10-89 | 53 | 3.0 | 210 | 16,027 | 10.7 |
| 02-01-89 | 71 | 2.9 | 214 | 21,879 | 14.6 |
| 02-23-89 | 273 | 3.1 | 104 | 31,056 | 20.8 |
| 03-22-89 | 330 | 2.9 | 79 | 37,541 | 25.1 |
| 04-19-89 | 173 | 3.0 | 85 | 21,175 | 14.1 |
| 05-16-89 | 189 | 2.9 | 45 | 12,247 | 8.2 |
| 05-23-89 | 114 | 3.0 | 49 | 8,044 | 5.4 |
| 06-7-89 | 140 | 3.1 | 62 | 12,499 | 8.4 |
| | System modifications | | | | |
| 8-89[1] | 37 | 4.5 | 119 | 6,340 | 4.2 |
| 9-89[2] | 33 | 4.6 | 121 | 5,750 | 3.8 |
| 10-89[3] | 34 | 4.6 | 136 | 6,659 | 4.5 |

[1] average of two sampling dates
[2] average of four sampling dates
[3] average of three sampling dates

Before the system modifications, influent water at the Latrobe wetland had an average pH of 3.0. After the modifications, the influent pH averaged 4.6. Iron concentrations ranged from less than 50 mg L$^{-1}$ during high flow periods to more than 200 mg L$^{-1}$ during low flow periods in 1988. Loading rates increased significantly during the winter and spring of 1989 when flows increased by five to ten times, but iron concentrations only decreased by two-thirds.

From a concentration perspective, the wetland had a highly variable effect on water chemistry (Figure 3a). In autumn months of 1987 (not shown), 1988, and 1989, the wetland lowered iron concentrations by 100-150 mg L$^{-1}$. In spring months of 1988 (not shown) and 1989, iron concentrations were sometimes lowered less than 20 mg L$^{-1}$.

From an iron-removal perspective, the wetland's performance was less variable. Between October 1988 and April 1989, area-adjusted iron removal only ranged from 2.4 to 6.7 gdm. For an unknown reason, iron removal was very low in May 1989. Removal rates were lower in autumn 1989 than autumn 1988, but this change was a result of decreased loading, not any apparent failure of the constructed wetland.

Before the modification, the average area-adjusted iron removal rate for the wetland was 3.8 gdm (s=2.0). When the post-modification data are included, the overall average removal is 3.6 gdm (s=1.8).

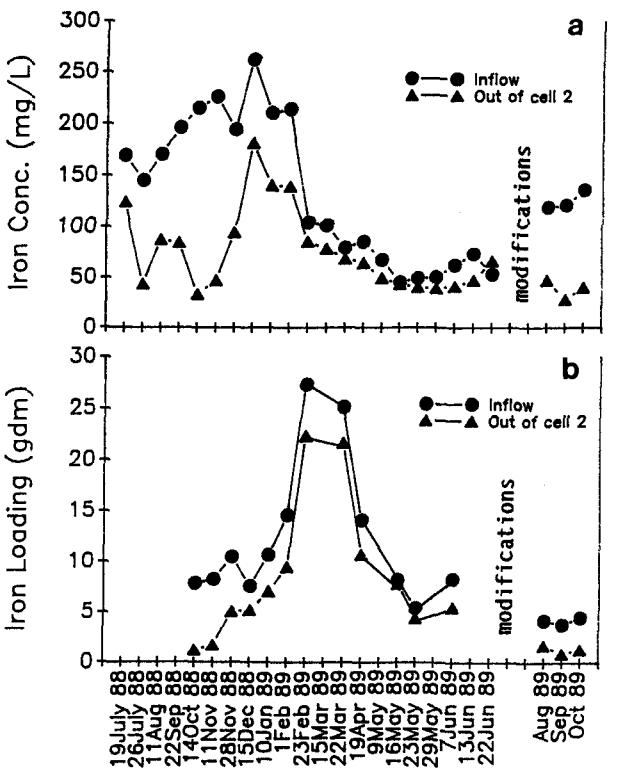



Figure 3. Iron chemistry at the Latrobe wetland: a) iron concentrations at the inflow and cell two outflow sampling stations; b) area-adjusted iron loading at inflow and cell two outflow stations.

Friendship Hill Wetland

The influent water at the Friendship Hill wetland had an average pH of 2.7 (Table 4). Iron concentrations varied from 82 to 195 mg L$^{-1}$. Flow rates were changed three times during the study period and ranged from 1.9 to 7.6 L min$^{-1}$.

Area-adjusted loadings into the cells varied from 3 to 19 gdm. Slight differences in area-adjusted loading rates for the individual cells were attributable to differences in cell surface areas (compare influent loading lines in Figure 4b, 4d, and 4f).

The wetland cells displayed similar patterns of iron removal (Figure 4). During the spring and summer, all cells displayed consistent iron removal. Average area-adjusted removal rates for this period (May 16 through Sept 19) were 5.0 gdm (s=1.9) for cell A2, 4.1 gdm (s=1.7) for cell B2, and 4.3 gdm (s=1.8) for cell C2. The overall average removal for this period was 4.5 gdm. With

Table 4. Influent Water at the Friendship Hill Wetland

| Date | Flow | pH | Inflow Fe | | |
|---|---|---|---|---|---|
| | L min$^{-1}$ | | mg L$^{-1}$ | g day$^{-1}$ | gdm[1] |
| 05-16-89 | 7.6 | 2.8 | 108 | 1182 | 10.8 |
| 05-23-89 | 7.6 | 2.6 | 119 | 1302 | 11.8 |
| 06-06-89 | 7.6 | 2.8 | 182 | 1992 | 18.1 |
| 06-21-89 | 7.6 | 2.7 | 85 | 930 | 8.5 |
| 07-06-89 | 7.6 | 2.9 | 82 | 897 | 8.2 |
| 07-25-89 | 7.6 | 2.8 | 82 | 897 | 8.2 |
| 08-08-89 | 3.8 | 2.8 | 139 | 758 | 6.9 |
| 08-23-89 | 3.8 | 2.6 | 144 | 785 | 7.1 |
| 09-06-89 | 3.8 | 2.6 | 195 | 1063 | 9.7 |
| 09-19-89 | 3.8 | 2.6 | 176 | 959 | 8.7 |
| 10-17-89 | 3.8 | 2.7 | 82 | 447 | 4.1 |
| 11-01-89 | 3.8 | 2.7 | 130 | 709 | 6.5 |
| 11-14-89 | 3.8 | 2.7 | 133 | 725 | 6.6 |
| 11-21-89 | 1.9 | 2.8 | 130 | 354 | 3.2 |
| 11-28-89 | 1.9 | 2.8 | 135 | 367 | 3.3 |
| 12-05-89 | 3.8 | 2.8 | 176 | 959 | 8.7 |
| 12-13-89 | 3.8 | 2.8 | 183 | 997 | 9.1 |
| 12-27-89 | 3.8 | 2.7 | 183 | 997 | 9.1 |
| 01-03-90 | 3.8 | 2.8 | 111 | 605 | 5.5 |

[1] gdm calculation based on an average surface area of 110 m$^2$ per cell, actual values were slightly lower for cell C2, higher for cells A2 and B2.

the onset of cooler (autumn) temperatures, iron removal by all cells became more variable. On several days, effluent samples contained more iron than the influent samples. During this period, water levels were lowered by 7-10 cm in all cells. This change may have influenced the iron removal results.

Over the entire observation period, the average removal of cell A2 was 3.5 gdm (s=2.8), of cell B2 was 2.8 gdm (s=3.3), and of C2 was 3.7 gdm (s=2.4). Iron removal for all cells over all dates averaged 3.3 gdm.

## Model Evaluation

The relationships between area-adjusted iron loading and iron removal for the three wetlands are shown in Figure 5. For the Somerset and Latrobe wetlands, which consist of cells connected serially, loading and removal rates were calculated for individual cells. This resulted in higher estimates of area-adjusted loading and removal than were obtained for the whole wetland systems (Figures 2b and 3b) because influent loadings were divided by only the area of each particular cell, not the entire wetland.

The Friendship Hill wetlands showed little relationship between iron loading and removal (Figure 5c). However, the maximum loadings, 18-19 gdm, were much less than the maximum loadings at the Somerset and Latrobe sites. Because the proposed model relies upon a broad range of loadings, conclusions about the iron removal capabilities of the Friendship Hill wetlands cannot be made at this time.

The Somerset site exhibited a strong relationship between area-adjusted iron loading and removal (Figure 5a). At iron loadings less than 15 gdm (n=11), a significant linear relationship between the two parameters existed.

$$\text{Removal} = 0.71 \times \text{Load} - 1.07 \quad (r=.88,\ P<.05) \qquad (6)$$

As loading increased beyond 15 gdm, no further increase in removal was observed.

$$\text{Removal} = 0.17 \times \text{Load} + 6.30 \quad (r=.45,\ P>.05) \qquad (7)$$

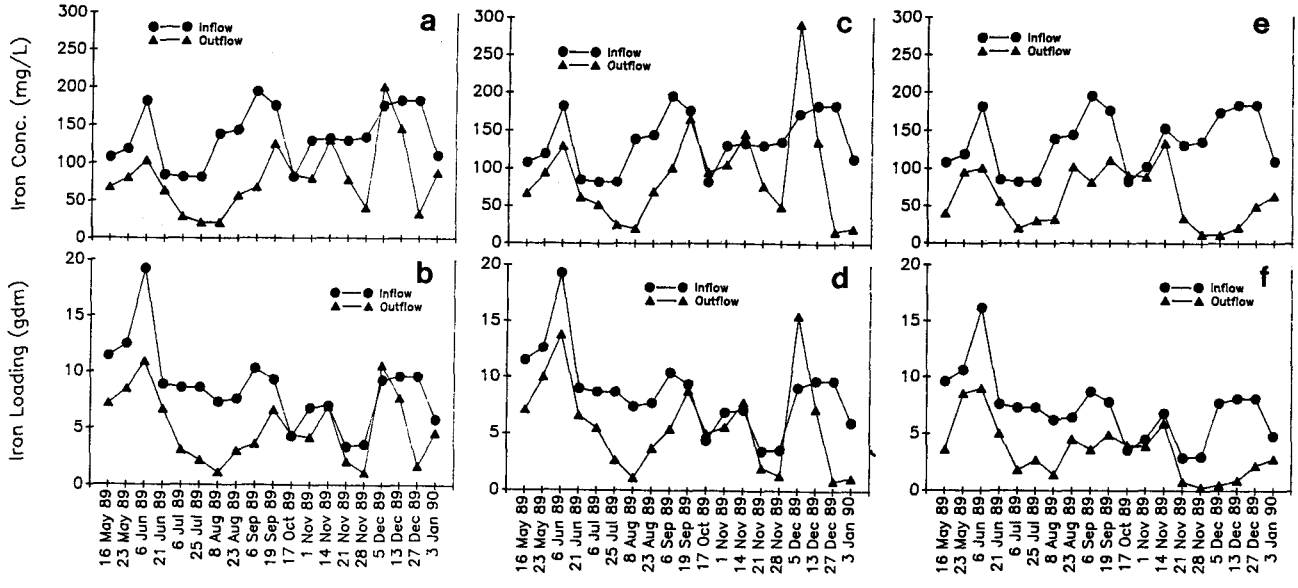



Figure 4. Iron chemistry at the Friendship Hill experimental wetlands. Inflow and outflow concentrations of iron are shown in a, c, and e. Area-adjusted iron loadings at the inflow and outflow stations are shown in b, d, and f.

The average removal of these nine high-loading observations was 10.6 gdm (s.e. = 1.1). This value is an estimate of the iron removal capability of the Somerset wetland.

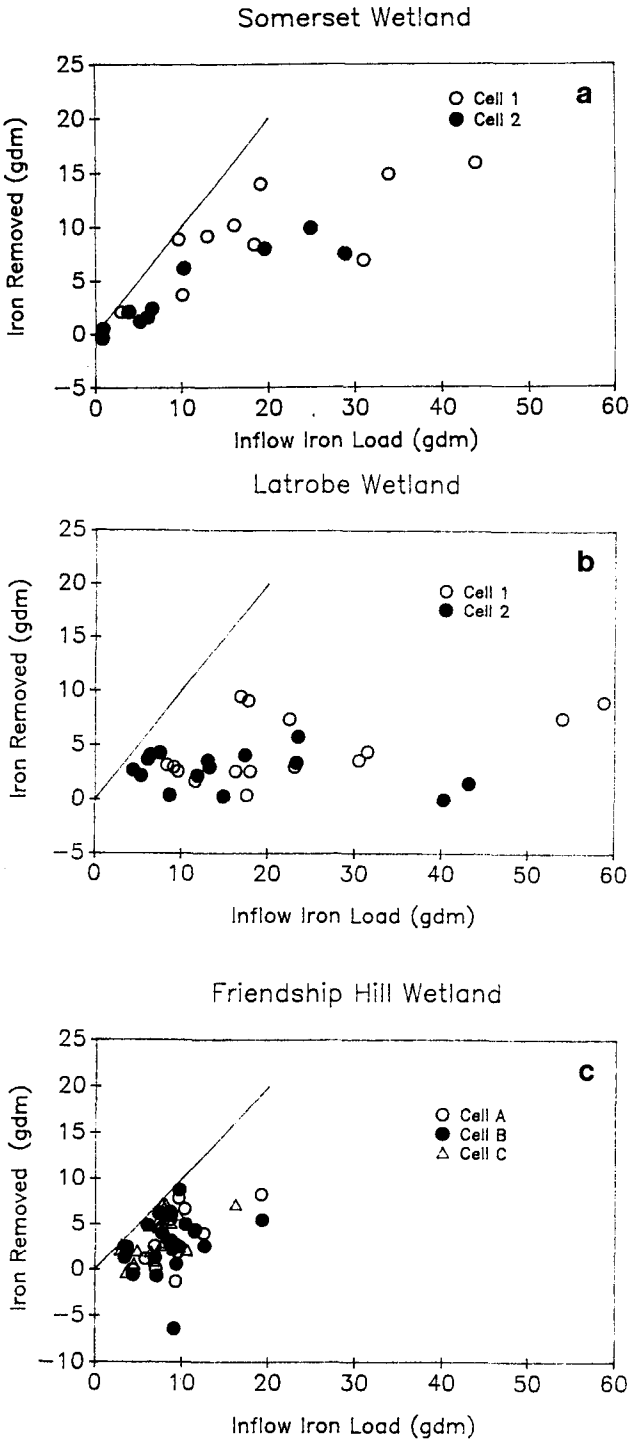



Figure 5. The relationship between area-adjusted iron removal and area-adjusted iron loading at the a) Somerset wetland, b) Latrobe wetland, and c) Friendship Hill wetlands. The straight line represents 100% removal (iron removal = iron loading).

No relationship existed between area-adjusted iron loading and removal at the Latrobe wetland (Figure 5b). This finding was not a result of insufficient variation in iron loading. The range of loadings at the Latrobe site was similar to that at the Somerset site. At loadings greater than 15 gdm, iron removal at the Latrobe site averaged 4.6 gdm, less than half that observed at the Somerset site. Two significant differences in water chemistry at the sites may contribute to these differences in iron removal. The influent pH at the Latrobe wetland averaged a full unit less than that at the Somerset site (3 vs 4). Lower pH may inhibit microbial processes that contribute to iron removal or decrease the stability of precipitated iron oxyhydroxide and sulfide compounds. Close scrutiny of the data results in contradictory evidence with regard to this hypothesis. The highest iron removal rates at the Somerset site were not associated with the highest influent pH values (which ranged as high as 5.1, see Table 2). However, at the Latrobe site the highest iron-removal rates occurred when the wetland was also increasing the pH of water as it flowed through the wetland (Hedin and Hammack, in review).

A second difference between the Somerset and Latrobe sites is the makeup of the high iron loadings. At the Somerset site, high loadings (>15 gdm) were due to high concentrations of iron (122-337 mg $L^{-1}$) combined with moderate flow rates (10-44 L $min^{-1}$). At the Latrobe site, high loadings were due to high flows of drainage (273-330 L $min^{-1}$) combined with moderate iron concentrations (79-104 mg $L^{-1}$). It is possible that iron removal rates are correlated with iron concentration. Higher iron removal rates may occur at higher iron concentrations. If this hypothesis is correct, then it may be necessary in future analyses of wetland performance to separate loading estimates into their flow and concentration components.

A common finding at all sites was that complete iron removal (< 1 mg $L^{-1}$) did not occur, even at very low loadings. Thus, the capability to remove 10 gdm at high loadings does not translate into complete iron removal at loadings less than 10 gdm. At the Somerset site, high loading events resulted in an average removal of 10.6 gdm. At loadings less than 15.0 gdm, iron removal averaged 54% of the inlet load. This decreased iron removal capability cannot be attributed to simply a concentration effect as several low loadings occurred when iron concentrations were greater than 100 mg $L^{-1}$.

Implications for Wetland Sizing

If the data presented here represent a crude estimate of iron removal capabilities in constructed wetlands, how does this bode for systems being built today using the original flow-dependent criteria? Sizing calculations for several hypothetical systems are shown in Table 5. Sizing estimates are made based on the original flow criteria and loading criteria developed from the data presented in this paper. We assume with the loading criteria that variation in iron removal results from pH effects, and that wetlands with pH 4 influents can remove 10 gdm of iron, while wetlands with pH 3 influents remove only 4 gdm of iron.

Table 5. Sizing Needs for Hypothetical Wetland Systems

| system | pH | Influent Fe mg $L^{-1}$ | Influent flow L $min^{-1}$ | Required Size ($m^2$) based on: flow[1] (5) | flow[2] (15) | load[3] |
|---|---|---|---|---|---|---|
| A | 4 | 25 | 30 | 150 | 450 | 108 |
| B | 3 | 25 | 30 | 150 | 450 | 270 |
| C | 4 | 150 | 50 | 250 | 750 | 1080 |
| D | 3 | 150 | 50 | 250 | 750 | 2700 |

[1] 5.0 $m^2$ per L $min^{-1}$ flow (Kleinmann et al. 1986)
[2] 15.0 $m^2$ per L $min^{-1}$ flow (Girts et al. 1987)
[3] pH 3: 4 gdm (this paper)
pH 4: 10 gdm (this paper)

Comparisons of wetland sizes based on flow and loading criteria indicate that wetland size estimates differ considerably depending on iron loading and pH (Table 5). For the highly contaminated drainage shown in systems C and D, the loading criteria suggest larger wetland needs than are suggested by either flow criterium. At pH 3, the methods differ by more than 300%.

For low flows of moderately contaminated water (systems A and B), the largest wetland sizes are produced by the 15 $m^2$ flow criterium. This finding probably reflects the fact that the 15 $m^2$ criterium was developed empirically from the size:flow relationships of wetlands that successfully lowered iron concentrations to regulatory levels. The iron removal criteria developed in this paper were derived from high-loading situations. Because iron removal capabilities appeared to decrease under low-loading situations, it is possible that larger areas of wetland are needed for removal of the last 10-20 mg $L^{-1}$ of iron.

These calculations should be considered preliminary, particularly for the low pH systems. Clearly, more data from these and other systems are necessary to confidently characterize the iron removal capabilities of constructed wetlands. Papers in this session by Kepler (1990) and Stark et al. (1990) should provide additional iron removal information for sites with pH 5 and pH 6, respectively.

Literature Cited


Brodie, G. A., Hammer, D. A. and D. A. Tomljanovich. 1988. Constructed wetlands for acid drainage control in the Tennessee Valley. In: Mine Drainage and Surface Mine Reclamation, Vol. I: Mine Water and Mine Waste, BuMines IC 9183, pp. 325-331.

Girts, M. A., R. L. P. Kleinmann and P. E. Erickson. 1987. Performance data on Typha and Sphagnum wetlands constructed to treat coal mine drainage. In: Proc. of the 8th Annual Surface Mine Drainage Task Force Symp., Morgantown, WV.

Kleinmann, R. L. P., Brooks, R., Huntsman, B. and B. Pesavento. 1986. Constructing Wetlands for the Treatment of Mine Water. Mini-course Notes, 1986 National Symposium on Surface Mining, Hydrology, Sedimentology, and Reclamation, University of Kentucky, Lexington, KY.

Kepler, D. 1990. Wetland sizing -- design and treatment effectiveness. In J. Skousen and J. Sencindiver (eds.) Proceedings of the 1990 Mining and Reclamation Conference and Exhibition, Charleston, WV.

McIntire, P. and H. Edenborn. 1990. The use of bacterial sulfate reduction in the treatment of drainage from coal mines. In J. Skousen and J. Sencindiver (eds.) Proceedings of the 1990 Mining and Reclamation Conference and Exhibition, Charleston, WV.

Stark, L. R., Kolbash, R. L., Webster, H. J., Stevens, S. E., Dionis, K. A. and E. R. Murphy. 1988. The Simco #4 wetland: biological patterns and performance of a wetland receiving mine drainage. In: Mine Drainage and Surface Mine Reclamation, Vol. I: Mine Water and Mine Waste, BuMines IC 9183, pp. 332-344.

Stark, L. R., Stevens, S. E., Webster, H. J., and W. Wenerick. 1990. Iron loading, efficiency, and sizing in a constructed wetland receiving mine drainage. In J. Skousen and J. Sencindiver (eds.) Proceedings of the 1990 Mining and Reclamation Conference and Exhibition, Charleston, WV.

# IRON LOADING, EFFICIENCY AND SIZING IN A CONSTRUCTED WETLAND RECEIVING MINE DRAINAGE[1]

by

Lloyd R. Stark, S. Edward Stevens, Jr.,
Harold J. Webster, and William R. Wenerick[2]

Abstract. The Simco #4 wetland was constructed near Coshocton, Ohio to treat a deep mine discharge in November 1985. It consists of three wetland cells totaling an area of 2623 $m^2$. It was planted with rhizomes of Typha latifolia (cattail) and over the course of three growing seasons it has evolved into a cattail-rice cutgrass (Leersia oryzoides) wetland. In 1988, Simco had 80% vegetation cover and a cattail stem density approaching 19 shoots/$m^2$. Wetland substrate depth was 65 cm, and the depth of surface water was 11 cm. The discharge averaged 111 mg/L total iron and a flow of 328 L/min. The treatment of water with respect to total iron has generally improved with age, averaging 62% in efficiency (Efficiency = Influent Fe - Effluent Fe/Influent Fe). Higher treatment efficiencies occurred in the summer and fall. Iron loading to the wetland is inversely correlated to treatment efficiency for total iron ($r^2$=47%). Therefore, as the daily mass of iron entering the wetland increased, the ability of the wetland to remove iron from the water decreased. This relationship, however, appeared to be dependent on the age of the wetland. With increasing wetland age, iron loading had less effect on efficiency. An analysis of loadings and outlet iron concentrations revealed a recommended loading of <47 kg Fe/day and a preferred loading of <28 kg Fe/day. Simco has removed an average of 13.6 g Fe/day per $m^2$ of wetland. However, Simco has effectively treated water having iron loadings of 10 g Fe/day per $m^2$ of wetland. The implication of this study at Simco was to recommend a maximum iron loading to the system in order to ensure adequate water treatment. The great variation in flow rates at Simco translates into the need to expand the current constructed wetland area in order to handle the high iron loadings of spring.

Additional key words: acid mine drainage, water treatment, Typha latifolia L., flow rates, Leersia oryzoides (L.) Sw.

## Introduction

Constructed wetlands are currently used as an alternative to the chemical treatment of acid mine drainage. In the last decade, well over 100 wetlands have been installed in northern Appalachia at active or abandoned coal mining sites (Kleinmann and Girts 1987). As of June of 1989, there were about 140 constructed wetlands currently in place as treatment systems in Pennsylvania (pers. comm., B. Hellier, PA Department of Environmental Resources, Hawk Run office). The principal water quality problems are high levels of Fe, Mn, Al, and low pH. Federal water quality standards require mine discharges to have a pH of 6 - 9, average total Fe of ≤3 mg/L, and average total Mn of ≤2 mg/L (U.S. Code of Federal Regulations 1981). Because of the prohibitive cost of using conventional chemical treatment (lime, soda ash, sodium hydroxide, limestone, hydrated lime, or caustic soda), wetlands are now commonly suggested as an inexpensive alternative or supplemental treatment system for coal mine drainage. The popularity of using wetlands for this purpose stems from reports of freshwater wetlands improving the quality of influent mine drainage (Wieder and Lang 1982).

With respect to total iron, treatment efficiency (Inlet [Fe]-Outlet [Fe]/Inlet [Fe]) is often used as an indicator of wetland performance (Kleinmann and Girts 1987). This measure yields a percentage which is useful in determining the performance of the wetland. For example, an efficiency of 70% indicates that the treatment system is lowering Fe concentrations between inlet and outlet by 70%. However, efficiency does not take into

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]Lloyd R. Stark is a Research Associate and William R. Wenerick is a Research Assistant in the Department of Biology, The Pennsylvania State University, University Park, PA 16802; S. Edward Stevens, Jr. occupies the Chair of Excellence in Molecular Biology, Department of Biology, Memphis State University, Memphis, TN 38152; Harold J. Webster is an Assistant Professor, Department of Biology, DuBois Campus, The Pennsylvania State University, DuBois, PA 15801. Publication in this proceedings does not preclude authors from publishing their manuscripts, whole or in part, in other publication outlets.

account flow rates, and thus taken alone is not very informative (Wieder 1988). When flow rates are included in calculations of wetland performance, it is possible to construct a quantitative budget reflecting the mass of iron (or other parameter) entering and leaving the wetland (e.g., Stillings et al. 1988). Influent iron to the system is termed iron "loading," and can be calculated as kg Fe/day. Loading (L/min x mg/L x 0.00144) is a better means of assessing influent iron levels than iron concentration, especially if inflow rates vary. For example, source water with 25 mg/L Fe at 378 L/min (100 GPM) discharges more than twice the iron to a wetland or stream than source water with 50 mg/L Fe at 75.6 L/min (20 gal/min): 13.6 vs. 5.4 kg Fe/day, respectively.

There have been no comprehensive studies to determine the optimal size of a wetland receiving a particular water flow rate and quality. Initially, flow rate alone was used to estimate wetland size. Kleinmann et al. (1986) suggested 5 $m^2$ of wetland for each L/min of mine drainage. After surveying constructed wetlands that were one to two years old, Girts et al. (1987) observed that the most efficient systems had a ratio of 15 $m^2$ per L/min flow. Brodie et al. (1988) suggested an iron loading of 1.92 g Fe/day per $m^2$ wetland when the influent pH exceeds 5.5 (=0.75 $m^2$ per mg Fe/min). This figure was based on efficiency studies of several wetlands in Tennessee, and represents the first time that loading, rather than flow, has been used in determining the appropriate wetland size. Although the optimal wetland size may be infinitely large, the area available for wetland construction is usually limited, necessitating calculations on minimum sizing criteria for specific loadings. Ultimately, several chemical parameters of the water in conjunction with flow ranges should be factored into a determination of minimum size. This article explores the seasonal performance of a constructed wetland with respect to iron over a period of four years, presents an inverse relationship between iron loading and treatment efficiency, and discusses the sizing parameters for the wetland under study.

## Materials and Methods

### The Simco #4 Wetland

In November 1985, construction of the Simco #4 wetland in Coshocton Co., Ohio was completed. It consisted of three wetland cells in sequence separated by small settling pools. The substrate consisted of a layer of crushed limestone (15 cm) covered with a layer of spent mushroom compost (45 cm). The total area of the system was 2,623 $m^2$ (28,233 $ft^2$; Cell 1 = 850 $m^2$, Cell 2 = 771 $m^2$, Cell 3 = 1,002 $m^2$). The average substrate depth in 1988-89 was 65 cm, while the average depth of standing water above the substrate was 11 cm. Typha latifolia (cattail) rhizomes were transplanted to an initial density of 3-4/$m^2$. Over three growing seasons, Simco has evolved into a cattail-rice cutgrass (Leersia oryzoides) wetland with nearly 80% cover and a cattail stem density approaching 19 shoots/$m^2$. Other details regarding construction and biological monitoring can be found in Stark et al. (1988).

In August 1987, the flow configuration was changed so that Cells 1 and 2 received one-half of the mine discharge, and a series of straw dikes was installed (Figure 1). The chief concern at the site has been to reduce the amount and necessity of chemical treatment used to control iron concentrations. Total iron at the seep has averaged 111 mg/L, and flow has averaged 328 L/min (87 gal/min). Influent pH is within compliance, averaging 6.5, and total manganese is near 2 mg/L. Other relevant water data for the inlet and outlet are given in Table 1.

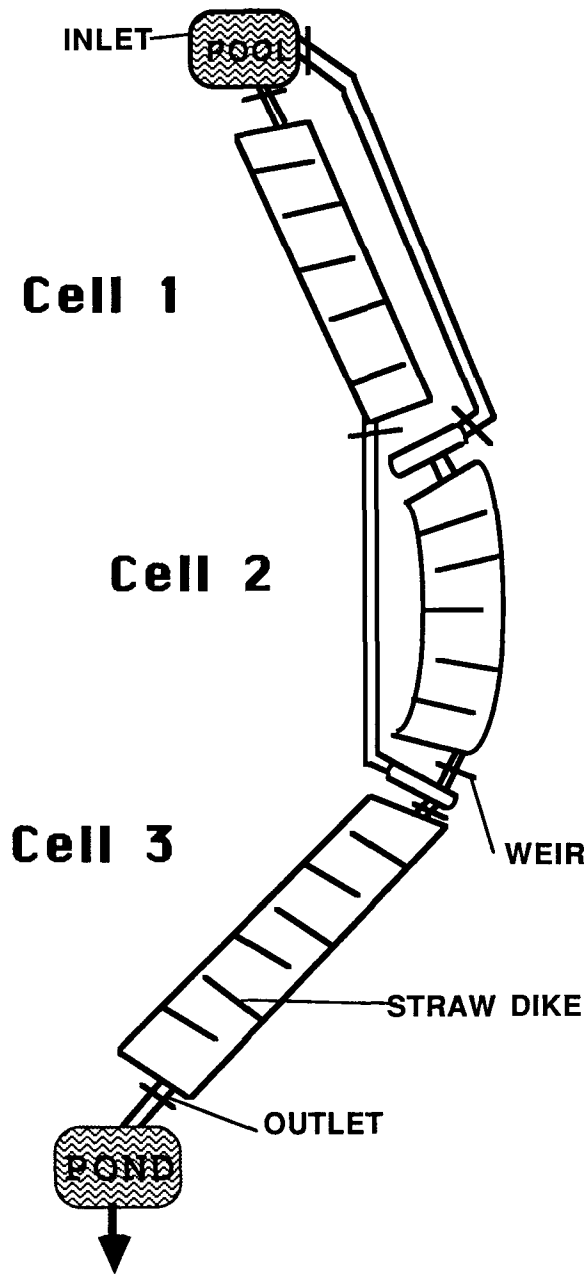



Figure 1. Diagram of the Simco constructed wetland.

### Water Sampling and Analysis

Water was collected at the inlet, outlet, and individual cell outlets at two or three week intervals. Water was analyzed by the Peabody Coal Lab (through 1987) and the American Electric Power Central Coal Lab (1988 on) for total iron. In the field, pH and flow rates were taken. Total iron was measured using a Perkin/Elmer 4000 and 5000 atomic absorption spectrophotometer using standard techniques. Water data were stored and manipulated on the IBM 370/3081 mainframe computer using the Statistical Analysis System (SAS Institute Inc. 1982).

**Table 1. Mean (± one standard deviation) water quality at Simco #4 inlet and outlet, November 1985 through October 3, 1989.**

| | Outflow* (L/min) | pH (S.U.) | Tot. Fe (mg/L) | Tot. Mn (mg/L) | Acidity (mg/L) | Alkalinity (mg/L) | Sulfate (mg/L) | T.S.S. (mg/L) | Temp. (°C) |
|---|---|---|---|---|---|---|---|---|---|
| Inlet | ---- | 6.5± 0.2 | 111± 26 | 2.0 ± 0.6 | 123 ± 82 | 87± 33 | 1146 ± 225 | 40 ± 24 | 12.8±1.6 |
| Outlet | 328±186 | 6.4± 0.3 | 42 ± 23 | 2.1 ± 0.6 | 32± 25 | 26± 18 | 1067 ± 230 | 50± 27 | ---- |
| Efficiency | ---- | -2% | -62% | +5% | -74% | -70% | -7% | +25% | ---- |
| (N) | (105) | (74) | (86) | (86) | (86) | (86) | (70) | (73) | (43) |

*Inflow rates were first sampled in September 1988 and have been very close to outflow rates through October 1989.

## Results and Discussion

### Concentration

The inlet water station is the deep mine portal, while the outlet water station is the effluent from Cell 3 (Figure 1). Total iron concentration at the inlet has fluctuated around 100 mg/L, ranging from 73 to 160 mg/L. Outlet concentrations have ranged from over 100 mg/L shortly after construction to 7 mg/L during the summer of 1988 (Figure 2A).

### Efficiency

Efficiency of the Simco #4 wetland in removing total iron has averaged 62 percent from wetland installation through October 3, 1989 (about four years; Table 1). Efficiency, computed as follows:

$$\frac{\text{Inlet Fe concentration - Outlet Fe concentration}}{\text{Inlet Fe concentration}}$$

has ranged from a low of 11% in December 1985, shortly after construction, to a high during July through September 1988 of 94%. Generally, the system has improved in efficiency with age (Figure 2B), with the greatest improvement in the first two years after construction.

A seasonal component to efficiency is illustrated by averaging efficiencies for each of the four seasons (Table 2). Higher efficiencies occurred in the summer and fall, whereas lower efficiencies usually occurred during the winter and spring. In the summer of 1988, the average system efficiency exceeded 90% for the first time. However, during the following winter, efficiency reached only 62%, 10% lower than the efficiency of the previous winter. This marked the first decrease in average seasonal efficiency (i.e., for a given season) since wetland installation. Previous to this, winter efficiency had improved with each successive winter (Table 2). This finding prompted an investigation into the patterns of iron loading for the wetland, and the relationship between loading and efficiency.

### Loading

Loading is defined as an estimate of the daily mass of iron entering the wetland system:

kg Fe/day (loading) =
Inlet Fe concentration (mg/L) x Inflow (L/min) x 0.00144

The calculated loading rate is an estimate because water samples and flow rates were sampled every second or third week. The highest loads to the system occurred in winter and spring, whereas relatively low loadings occurred during summer and fall. Iron loading closely

**Table 2. Seasonal efficiencies (%: Inlet Fe – Outlet Fe/Inlet Fe) for total iron at Simco #4.**

| | 1985 | 1986 | 1987 | 1988 | 1989 |
|---|---|---|---|---|---|
| Spring | —— | 37 | 55 | 63 | 61 |
| Summer | —— | 56 | 62 | 91 | 73 |
| Fall | —— | 53 | 73 | 82 | 64 |
| Winter | 28 | 51 | 72 | 62 | —— |

Spring: March – May
Summer: June – August
Fall: September – November
Winter: December – February

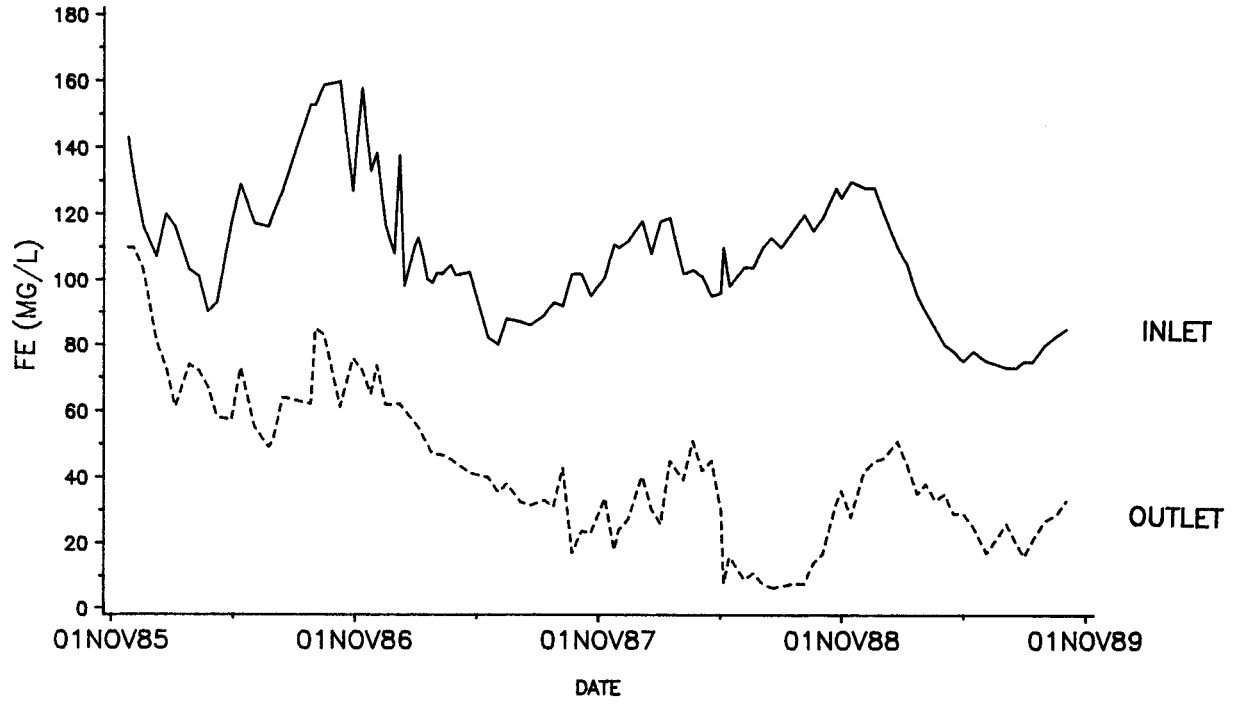



Figure 2A. Inlet and outlet concentrations of total iron at the Simco wetland.

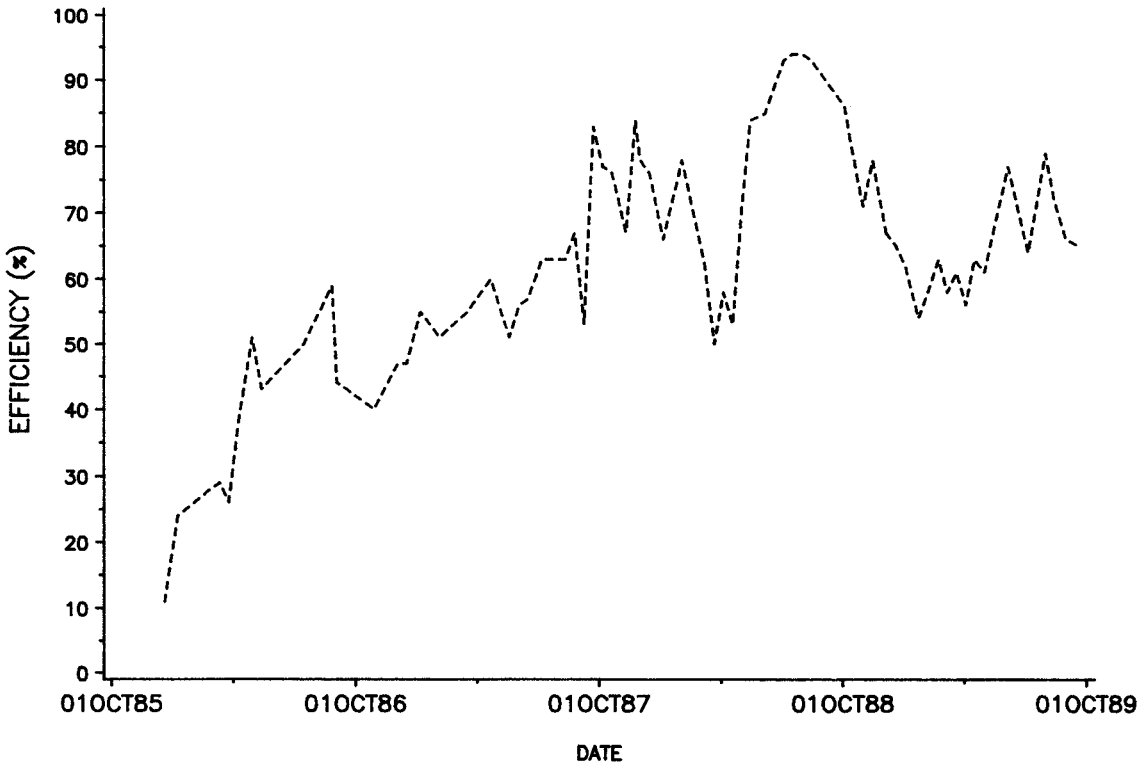



Figure 2B. Treatment efficiency (%) for total iron at the Simco wetland (Efficiency = Influent Fe - Effluent Fe/Influent Fe).

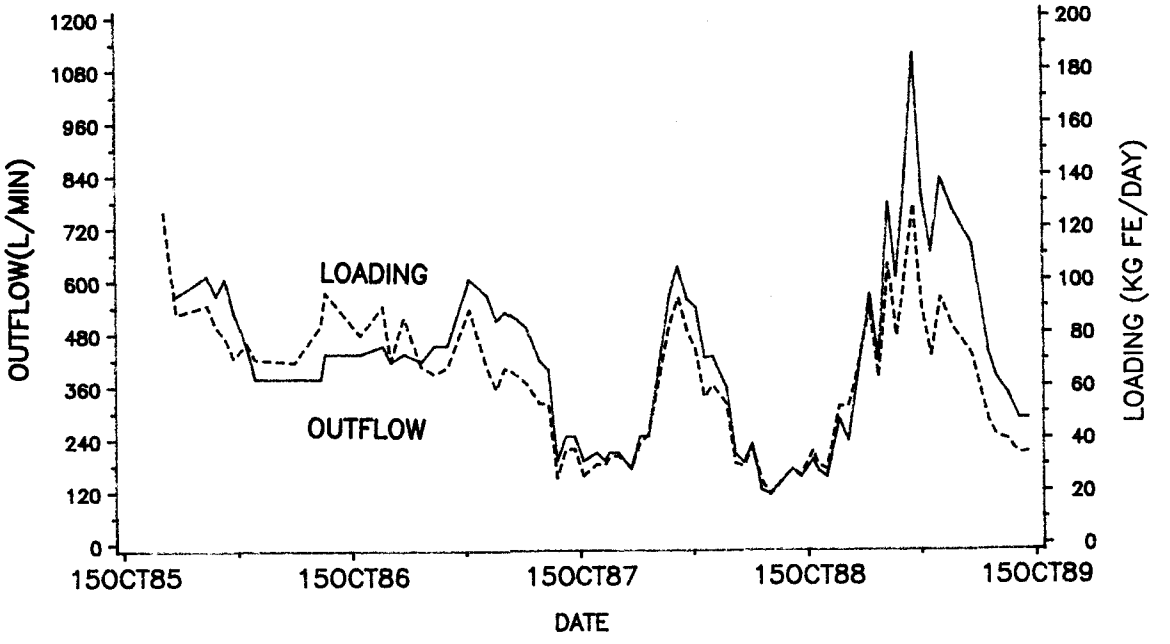



Figure 3. Iron loading in relation to outflow at the Simco wetland.

corresponded to flow rates, especially from mid-1987 on (Figure 3). While the concentration of iron in the seep water has remained relatively stable, flow rate has ranged from 114 to 1,116 L/min, and closely mirrored loading. Higher flows have occurred each spring, with the greatest flows recorded since installation during spring of 1989, when rainfall totals were above normal. Lower flows have coincided with the summer and fall months, with the unusually dry summer/fall of 1988 resulting in the lowest flows since wetland installation. Generally, as flow rate declined, the concentration of iron at the seep increased; as flow increased, iron concentration declined (Figure 4). This rough inverse relationship between inlet total iron and flow results in higher iron concentrations at the seep during summer and fall, and lower concentrations during winter and spring. Despite this relationship, the high variation in flow rates results in a close correspondence between iron loading and flow. Therefore, iron loading is determined moreso by flow rates than by iron concentration.

Noting that (1) the very low flows of the summer of 1988 coincided with the highest treatment efficiencies recorded at Simco, and (2) that efficiencies dropped off somewhat when normal flows resumed, an analysis of the relationship between iron loading and treatment efficiency was undertaken. A plot of efficiency versus iron loading by year (Figure 5) is revealing. This scatterplot represents water data from December 19, 1985 (when flow data and water data were first available) through October 3, 1989.

A regression of efficiency vs. loading yields the following linear equation:

(a) Efficiency = (-0.46) (Loading) + 90
$r^2$ = 47% ($p<0.01$)

As loading to the wetland system increased, the efficiency decreased with respect to total iron. Conversely, the higher efficiencies corresponded with the lower loadings. Because a major component of loading is flow, the inverse relationship between flow and efficiency is expected (Figure 6). While other factors certainly affect efficiency, the relationship between loading and efficiency was apparent from 1985 to 1988. However, in 1989 this relationship between iron loading and wetland treatment efficiency weakened noticeably. In 1989, flows increased markedly following heavy spring rains; however, system efficiency remained high and did not reflect the increases in iron loading. During February to July 1989, flows averaged 782 L/min, and the average loading was 91 kg Fe/day. The resulting average efficiency of 63% over this period was more than twice the predicted efficiency of 27% going into this period. This may indicate a wetland aging effect where the wetland processes are improving as the system becomes fully established. Hence, the Simco wetland treated higher iron loadings more efficiently in its fourth spring than in earlier springs. Previous to 1989, low loadings always occurred in the warmer months and yielded high efficiencies. However, late in 1989 (December) loadings dropped below 30 kg Fe/day, and yet efficiencies remained below 70%. We attribute these depressed efficiencies to a cold weather effect.

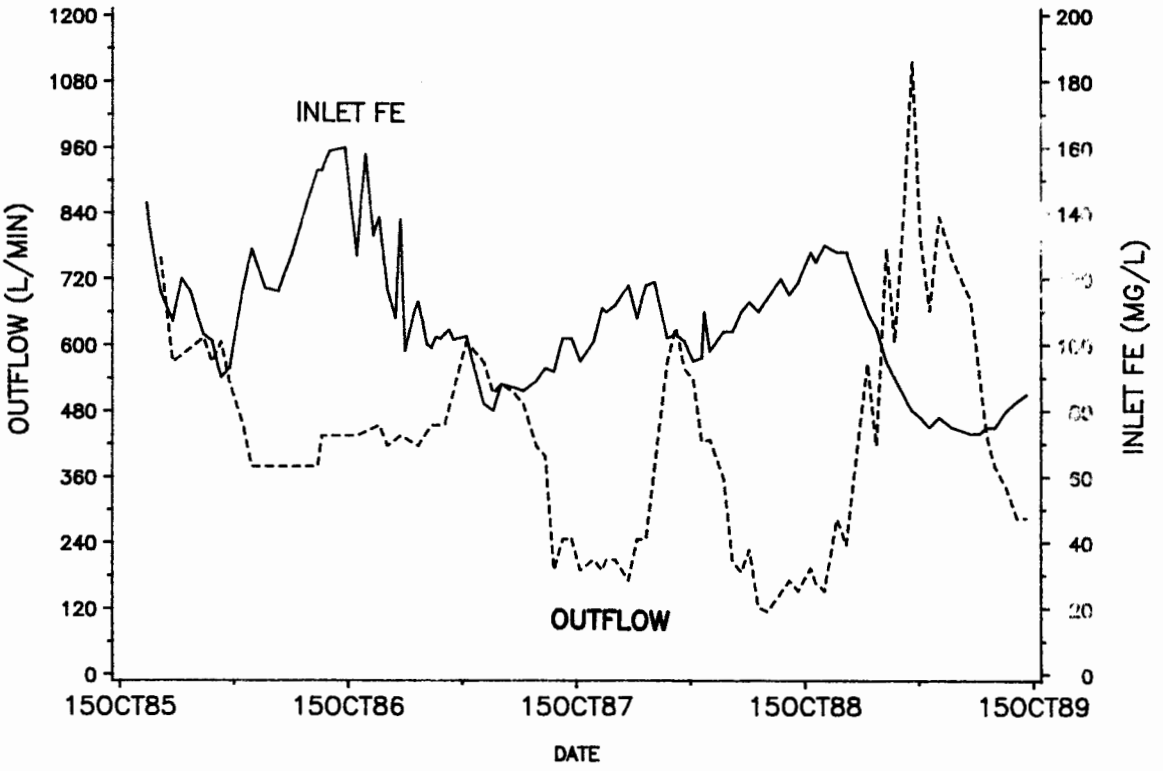



Figure 4. The relationship between outflow and total iron concentration at the inlet at the Simco wetland.

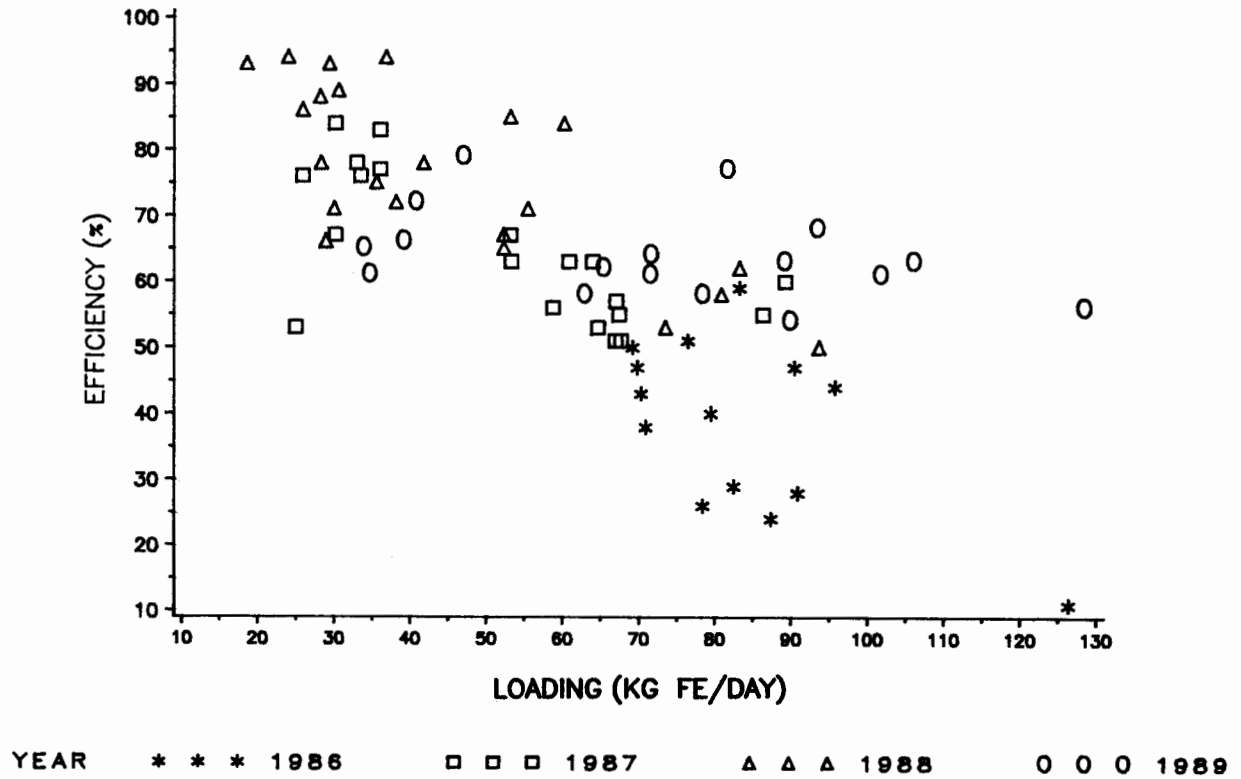



Figure 5. Plot of treatment efficiency for total iron versus iron loading at the Simco wetland (by year).

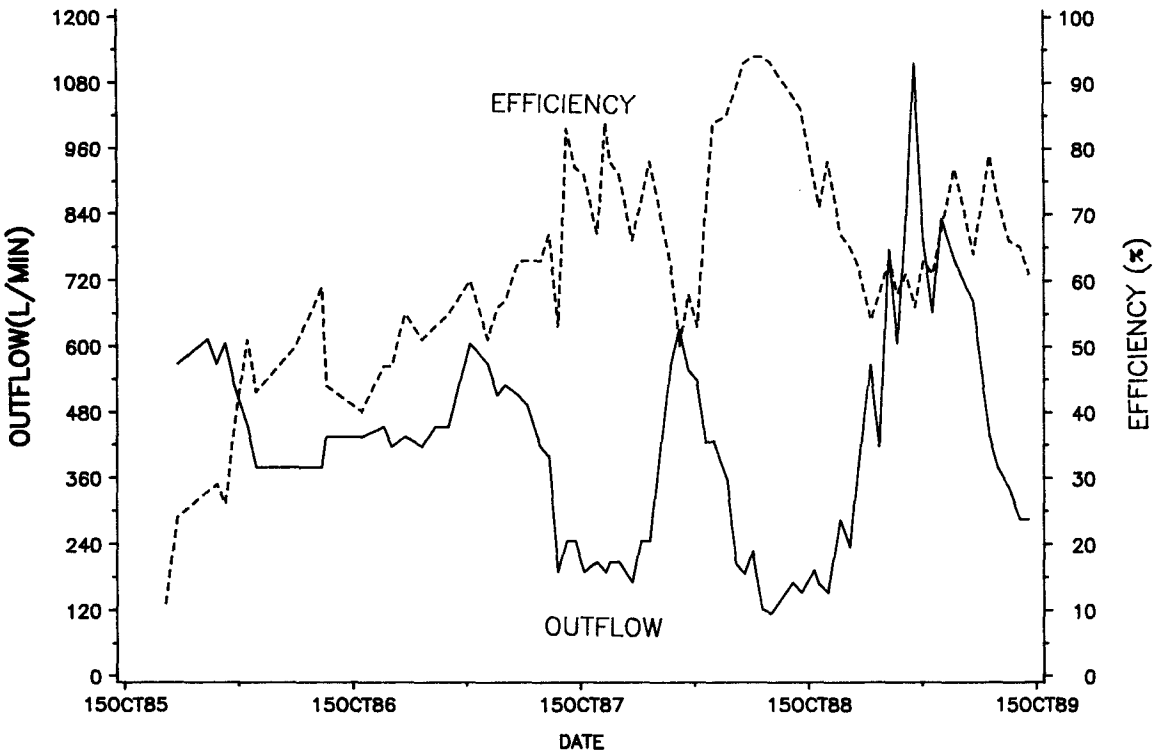



Figure 6. The relationship between treatment efficiency for total iron and outflow at the Simco wetland.

Wetland Sizing

One approach to the question of sizing is to (1) isolate those periods during which the wetland was treating the water effectively; (2) determine the loading to the wetland at these times; (3) assume that these loadings represent the preferred loading maxima for the system; and (4) derive a coefficient of size based on the loading at these times and the size of the wetland in question. Working backwards like this is necessary because at the time of wetland construction, no data on sizing existed. Because of the presence of settling ponds downstream of the Simco #4 wetland and within the permitted area, total Fe at the wetland outlet does not have to be <3.0 mg/L to meet compliance. Rather, if total Fe is lowered by the wetland to approximately 25 mg/L at the outlet, on-site chemical treatment can be avoided. When outlet iron concentrations were <25 mg/L, the iron loading at Simco averaged 47 kg Fe/day (Table 3). When outlet iron concentrations exceeded 25 mg/L, loadings averaged about 57 kg Fe/day. Correspondingly, when loadings were <47 kg Fe/day, the mean outlet iron concentration was 25.1 ± 11.0 mg/L ($\bar{x}$ ± S.D.). Therefore, the recommended iron loading at Simco should be under 47 kg Fe/day. In order to ensure that chemical treatment is not required, preferred loadings would be in the neighborhood of 28 kg Fe/day; this would eliminate much of the variation in outlet iron concentration.

In order to make comparisons among several wetlands with respect to the question of sizing, iron loading and removal can be based upon a unit wetland area, a square meter. Therefore, the grams of iron loaded or removed to a wetland per day per square meter can be termed gdm load and gdm removed. This convention is used for the other similar papers in this session. A recommended loading not to exceed 47 kg Fe/day and preferred loading of <28 kg Fe/day correspond to 17.9 and 10.6 gdm, respectively.

The gdm load and gdm removed are positively correlated ($r^2$=46%; P<0.01); as loading to the wetland increased, removal rates per $m^2$ also increased. With respect to efficiency, as loading increased while efficiency decreased, the mass of iron removed actually increased. This finding underscores the reason why iron removal averages (about 13 gdm for Simco) cannot be used in computations of wetland sizing: while Simco has removed a high of 27 gdm (April 4, 1989) it did so when the loading was 49 gdm, and thus treatment was not effective on this date. A plot of gdm load and gdm removed over time indicates how the mass of iron removed at Simco corresponds to the mass of iron loaded to the wetland (Figure 7).

## Discussion

At the Simco wetland, we can address wetland sizing using past performance and water quality data. For the site in question, iron loading should not exceed 47 kg Fe/day, and preferably be <28kg Fe/day. This translates to a preferred iron loading of 10.6 gdm, or roughly five times the estimate of Brodie et al. (1988) for sites having an influent pH greater then 5.5: 1.92 gdm. Note, however, that compliance levels for iron are not required at the

**Table 3. Mean iron loadings at Simco at a range of wetland outlet iron concentrations (Sept. 1, 1987 through Dec. 31, 1989; gdm = g Fe per day per $m^2$ wetland).**

| Outlet Fe (mg/L) | Loading (kg Fe/day) | Equivalent gdm | Loading Range | N |
|---|---|---|---|---|
| 0–15 | 28.0 | 10.6 | 19–37 | 6 |
| 16–25 | 47.0 | 17.9 | 26–94 | 12 |
| 26–35 | 56.8 | 21.6 | 24–128 | 17 |
| 36+ | 57.5 | 21.8 | 25–90 | 15 |

Note: When iron loading is <47 kg Fe per day (equivalent to 17.9 gdm), outlet iron averages 25.1 ±11.0 mg/L, N=30.

Simco outlet; the iron concentration must be <25 mg/L at the outlet in order to avoid chemical treatment. If compliance levels for iron were required at the outlet, the gdm estimate of Brodie et al. (1988) would likely apply. Our estimate is specifically for Simco, and should be cautiously considered in connection with other sites. The water at Simco is mitigated by the age of the wetland, relatively high pH, low levels of Mn, and presence of alkalinity. Should the pH be lower and influent Mn be greater, then the figure of 10 gdm is probably too high.

The efficiency of the Simco wetland is directly related to the iron loading; 47% of the observed variation in efficiency can be explained by the iron loading rate. The implication of this relationship for Simco was that a maximum recommended flow could be established that corresponded to an acceptable level of metal removal. In addition, it points up a possible explanation for the observed seasonal patterns of treatment efficiency in constructed wetlands. Lower treatment efficiencies at Simco occurred in the winter and spring; higher treatment efficiencies occurred during summer and fall. The highest loadings occurred during the highest flows (winter and spring), whereas the lowest loadings occurred during the lowest flows (summer and fall). Therefore, the depressed efficiencies of winter and spring may simply be a result of higher loadings, as opposed to temperature-dependent biological reactions. Seasonally depressed efficiencies at Simco probably result from a combination of factors, one of which is higher iron loadings.

How long wetland treatment systems will continue to remove iron is of concern to all involved with wetlands. A quantitative estimate for Sphagnum-based wetlands is provided by Wieder (1988). However, cattail-compost systems may be fundamentally different and require further study. Perhaps the greatest concern is over precipitated iron accumulating in the sediments over many years, leading to the eventual filling and probable short-circuiting of flow patterns, negating much of the water treatment.

Future efforts at Simco will focus on projecting rates of sediment buildup with respect to loading and removal.

Acknowledgments. This research was supported by American Electric Power Service Corporation (Simco, Inc., contract C-6900). We extend thanks to T. Romanoski, R. Kolbash, R. Hedin, D. Preston, S. Smith, S. Anderson, C. Bahler, E. de Veau, E. Murphy, B. Will, R. Stark, B. Pesavento and D. Kepler, all of whom provided assistance. Computer time was provided by The Pennsylvania State University.

## REFERENCES

Brodie, G. A., D. A. Hammer, and D. A. Tomljanovich. 1988. Constructed wetlands for acid drainage control in the Tennessee Valley. In Mine Drainage and Surface Mine Reclamation. Vol. 1: Mine Water and Mine Waste. Bureau of Mines Circular No. 9183, Pittsburgh, PA.

Girts, M. A., R. L. P. Kleinmann, and P. M. Erickson. 1987. Performance data on Typha and Sphagnum wetlands constructed to treat coal mine drainage. In: Proc. of the 8th Annual Surface Mine Drainage Task Force Symposium, Morgantown, WV.

Kleinmann, R. L. P., R. Brooks, B. Huntsman, and B. Pesavento. 1986. Constructing Wetlands for the Treatment of Mine Water, mini-course notes, Lexington, KY.

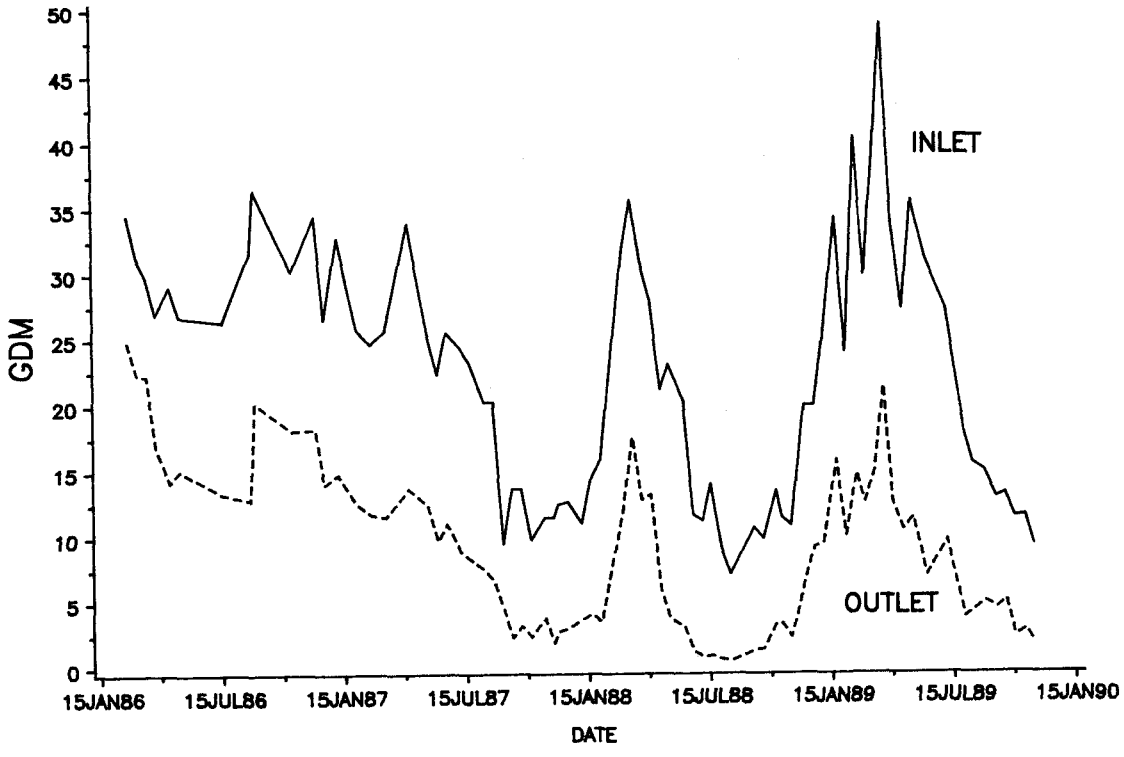



Figure 7. Iron loading and removal on the basis of grams per day per square meter of wetland (gdm load vs. gdm removed).


---------- & M. A. Girts. 1987. Acid mine water treatment in wetlands: an overview of an emergent technology. In K. R. Reddy and W. H. Smith (eds.), "Aquatic Plants for Water Treatment and Resource Recovery." Magnolia Publishing, Orlando, FL.

SAS Institute Inc. 1982. SAS User's Guide: Statistics, 1982 Edition. SAS Institute Inc., Cary, NC.

Stark, L. R., R. L. Kolbash, H. J. Webster, S. E. Stevens, Jr., K. A. Dionis, and E. R. Murphy. 1988. The Simco #4 wetland: biological patterns and performance of a wetland receiving mine drainage. In Mine Drainage and Surface Mine Reclamation. Vol. I: Mine Water and Mine Waste. Bureau of Mines Information Circular No. 9193, Pittsburgh, PA.

Stillings, L. L., J. J. Gryta, and T. A. Ronning. 1988. Iron and manganese removal in a Typha-dominated wetland during ten months following its construction. In Mine Drainage and Surface Mine Reclamation. Vol. I: Mine Water and Mine Waste. Bureau of Mines Information Circular No. 9193, Pittsburgh, PA.

U.S. Code of Federal Regulations. 1981. Title 30 - Mineral resources; Chapter VII - Office of Surface Mining Reclamation and Enforcement, Department of the Interior; Subchapter B - General Performance Standards; Part 715 - General Performance Standards. July 1, 1981.

Wieder, R. K. 1988. Determining the capacity for metal retention in man-made wetlands constructed for treatment of coal mine drainage. In Mine Drainage and Surface Mine Reclamation. Vol. I: Mine Water and Mine Waste. Bureau of Mines Information Circular No. 9193, Pittsburgh, PA.

---------- & G. E. Lang. 1982. Modification of acid mine drainage in a freshwater wetland. In B. R. McDonald (ed.), "Proceedings of the Symposium on Freshwater Wetlands of the Unglaciated Appalachian Region." West Virginia University, Morgantown, WV.

This page intentionally left blank.

# WETLAND SIZING, DESIGN, AND TREATMENT EFFECTIVENESS FOR COAL MINE DRAINAGE [1]

by

Douglas A. Kepler [2]

Abstract. Constructed wetlands are a proven means of treating and improving the quality of mine drainage, but they do so with varying degrees of effectiveness. The area and/or volume of constructed wetlands may influence effluent quality as much as the vegetative make-up of these systems and therefore, becomes a primary design consideration. Accurate flow data is paramount in determining wetland sizing requirements and ultimately in treatment efficiency, and should become a routine part of any wetland monitoring program. Evidence exists that seasonal variations occur for iron removal in constructed wetlands, but these variations may be influenced as much by "total loads in" as by treatment area and biological efficiency. Furthermore, there is an inverse relationship between the percent of ferrous iron ($Fe^{2+}$) to ferric iron ($Fe^{3+}$) in the total iron load and the removal efficiency of total iron in the wetland, indicating that the actual flow regime in the system is as important as area and vegetation. The oxidation and removal of iron from mine drainage often results in decreased pH values. However, iron removal and pH increases have been demonstrated in constructed wetlands without the use of alkaline substrates.

Additional key words: Mineral loading, seasonal efficiency, flow regime.

## Introduction

Constructed wetlands are a proven means of treating and improving the quality of mine drainage, but they do so with varying degrees of effectiveness. There are numerous reasons as to why constructed wetlands vary in treatment effectiveness, but one important consideration relates to the sizing of such systems. That is, "how large does a wetland need to be to effectively treat a given mine discharge?" Iron loads are given prime consideration in this discussion of sizing because of the various implications iron concentrations have on the entire functioning of wetland treatment systems.

The presence of iron in discharges can inhibit the removal of other minerals from those discharges in both constructed and natural wetlands (Kepler 1986; Stevens, et al. 1989). Additionally, the presence and/or chemical state of iron will be shown to be critical in the complete wetland treatment system; i.e., one that is designed to increase pH and alkalinity values as well as remove iron and acidity.

## Study Area

The Site 1 wetland is located in Venango County, PA and was constructed in October of 1987. The wetland is divided into two parallel series of four ponds each, with a common initial collection pond (Table 1). An attempt was made to provide equal flows to both series over the course of the study, although variations did occur. However, the data are presented in loadings, which does allow for comparisons of iron removal between the parallel channels. The mean total flow for these months was 106 L/min, with a range of 77-143 L/min.

The drainage at Site 1 is associated with the Lower Kittanning Coal Seam and has been flowing unabated from an abandoned drift mine opening since at least 1940. The sample period was November 1987 through March 1989 (May and June, 1988 are discussed separately because of anomalies in the data resulting from physical alterations to the wetland during this time period).

The Site 2 wetland is located in Clarion County, PA and was constructed in August, 1988 through funding from the Pennsylvania Department of Environmental Resources, Bureau of Abandoned Mine Reclamation (in conjunction with EADS acting as biological consultant). The quality of this mine discharge is a reflection of the overburden associated with the Upper Clarion Coal Seam which was surface mined at this site in the early 1970's. Site 2 is characterized by an open water pond covering roughly 125 $m^2$ (depth = 1.5m), followed by a series of four *Typha* dominated cells, each measuring approximately 6m by 30m (180$m^2$ each).

---

[1] Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia April 23-26, 1990.

[2] Douglas A. Kepler is an Environmental Scientist with The EADS Group, General Engineering Division, Clarion, PA 16214

Table 1. Site 1: Individual pond surface areas and volumes.

| Configuration* | Surface Area ($m^2$) | Volume (L) |
|---|---|---|
| TTAA | | |
| T1 | 42 | 15,897 |
| T2 | 49 | 10,087 |
| A1 | 67 | 45,420 |
| A2 | 63 | 39,080 |
| Total | 221 | 110,484 |
| AATT | | |
| A1 | 80 | 47,483 |
| A2 | 60 | 39,440 |
| T1 | 41 | 11,241 |
| T2 | 49 | 11,241 |
| Total | 230 | 109,404 |
| Initial Pond | 102 | 53,368 |

*Configuration refers to planting strategy in parallel systems. T = *Typha*, A = Algae

Flows were inadvertently not monitored at this site during the initial months of collection, but were determined for the latter half of the existing analyses. This oversight is typical of many monitoring programs and has been taken into consideration in the ensuing discussions. Accurate flow data is paramount in the determination of wetland treatment efficiency and ultimately sizing requirements, and should become a routine part of any wetland monitoring program. The sample period for this discussion is August 1988 through September 1989, with the mean total flow for the measured months being approximately 12 L/min, at a range of 2 to 50 L/min. A characterization of the quality of the two source discharges is given in Table 2.

The predominant planting (excluding the noted algal ponds) in the two wetlands was *Typha latifolia* . Both wetlands were excavated and sealed with on-site clay and incorporated 30cm of composted manures and hay as substrate in the areas dominated by the *Typha* plantings. The *Typha* were planted as core samples on roughly 0.6m centers, but spread to essentially one hundred percent surface coverage by the end of the first complete growing season. Core samples refer to hand dug, intact rhizome/soil units.

## Methods

The two principle wetlands discussed in this paper have at a minimum, fourteen months of weekly water analyses collected from various points within the systems. The data are discussed mainly in terms of total mineral loadings as opposed to concentrations; therefore, flow data plays a large role in these evaluations. Loadings are calculated by multiplying concentration data (mg/L) by flows (L/min) by a conversion factor (1.44 ) = gms/day. The loading values will be adjusted to reflect wetland sizing by dividing gms/day by area ($m^2$). The resultant values therefore indicate gms iron/day/$m^2$ and for brevity will be given as "gdm." The importance of evaluating wetland performance by this method is illustrated in the following two examples.

First, the importance of loading data as opposed to concentration data is shown in a comparison of two water collection points, one showing an iron concentration of 1mg/L at a flow of 100L/min and the other a concentration of 100mg/L at a flow of 1L/min. Any conclusions based on the loading data would indicate identical values. The concentration values alone could easily lead to erroneous conclusions.

In like fashion, the importance of area in treatment evaluations may be seen in the comparison of two wetlands that remove equal amounts of iron from their respective flows, eg., 100gms/day. If one of these wetlands encompasses twice the treatment area of the other, say 1,000$m^2$ versus 500$m^2$, than the smaller system is twice as effective in removing iron as is the larger wetland. However, there are additional factors that must be considered in regards to both loading and sizing evaluations in any wetland treatment discussion that will be subsequently examined in this paper.

Flows were calculated by means of a portable cutthroat flume and by timing with a bucket and stopwatch. Site 1 water samples were collected and analyzed by the EADS Group using standard methods (e.g. American Public Health Association), while the vast majority of data from Site 2 was taken from samples collected and analyzed by PA D.E.R. personnel. Monitoring points were established at AMD source and final wetland discharge points and at the distinct inflow/outflow points of the individual sections of the entire wetland systems. Typical analyses inclued pH, alkalinity, acidity, conductivity, total and ferrous iron, total and dissolved manganese, aluminum, and sulfate. The water analyses were examined and are discussed as a whole and by month.

## Results

The total iron removal rates per pond in gdm for Site 1 are shown in Table 3. The monthly removal rates for total iron from the first two sets of opposite ponds at site 1 are graphically depicted in Figures 1 and 2. The arrangement of ponds at Site 1 was such that the order of planting was reversed in the opposite channels. This design allowed for comparisons to be made as to the effectiveness of both vegetative make-up and area/volume considerations in iron removal.

The *Typha* dominated ponds at this site removed more iron from the flows on a per area basis than did their counterpart algal ponds. This removal translates into an even greater efficiency on a per volume basis since the algal ponds were built with a greater depth than the *Typha* ponds. Ponds T1 and T2 in the TTAA series showed the greatest, consistent propensity toward iron removal in the wetland, while three of the algal ponds and the two remaining *Typha* ponds displayed relatively consistent removal rates (Table 3). The final algal pond in the TTAA series consistently displayed

Table 2. Mean source quality of AMD at Sites 1 and 2*.

| | pH | Alkalinity | Acidity | Sulfate | Fe2+ | Fe(tot) | Mn | Al |
|---|---|---|---|---|---|---|---|---|
| site 1 | 5.5 | 50 | 250 | 2100 | 35 | 90 | 50 | 1 |
| site 2 | 5.7 | 28 | 145 | 970 | 135 | 180 | 20 | <1 |

*Alkalinity and acidity = $CaCO_3$ equivalent, others in mg/L

Table 3. Site 1: Total iron removal rates per pond in gdm.

| Configuration | gdm (mean) | gdm (range) |
|---|---|---|
| TTAA | | |
| T1 | 25.3 | 8-53 |
| T2 | 26.1 | 7-68 |
| A1 | 11.1 | 5-28 |
| AATT | | |
| A1 | 16.5 | 1-35 |
| A2 | 14.7 | 5-35 |
| T1 | 13.3 | 4-28 |
| T2 | 14.2 | 1-31 |

Figure 1. Total iron removal in gdm, AATT A1 vs. TTAA T1.
November 1987 = 1 through March 1989 = 17

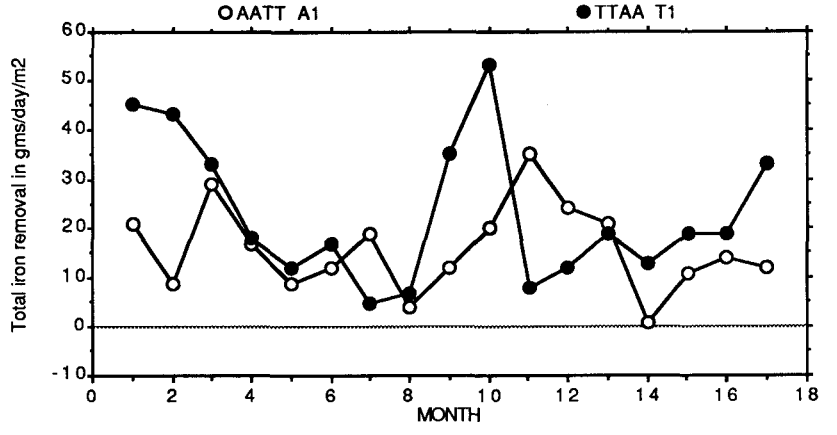
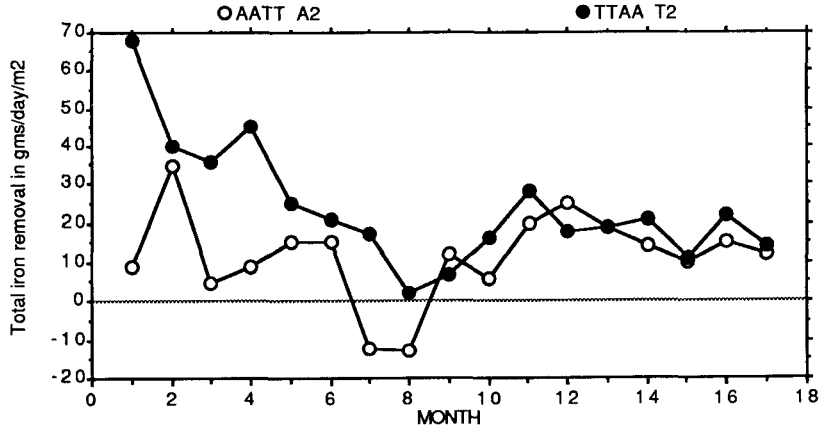



Figure 2. Total iron removal in gdm, AATT A2 vs. TTAA T2.
November 1987 = 1 through March 1989 = 17



greater effluent iron values than influent values. This finding resulted from an inability to collect "clean" samples because of concentrated iron precipitates at the sampling location and does not reflect the true treatment efficiency in this pond. TTAA A2 is not included in subsequent calculations or discussions.

There is a positive relationship between total inflow iron loads and iron removed in gdm in the TTAA T1, T2, and A1 ponds and pond T2 of the AATT series. The *Typha* dominated areas in particular appear to be more effective at removing iron at higher loads than at lower loading rates. The Site 2 data further suggest that iron is more effectively precipitated in large versus small quantities. The initial open water pond at Site 2 was designed to act as a sedimentation basin and as a retention pond for iron removal, because the relatively low flows and high pH values found here lent well to this design. The mean removal rates (gdm) on a monthly basis for this pond are shown in Figure 3, with the actual mean removal for the extent of monitoring being 15.8 gdm (range = 4.4 - 81.6). The high value occurred in May of 1989 during the highest measured flow value period for this system. The mean removal rate is similar to much of the Site 1 data, but is rather high when compared with the next two cells in series, where mean removal rates equalled 1.4 and 0.7 gdm respectively (Figure 3).

Seasonally, wetlands are generally considered to be less effective regarding treatment during the colder months, since biological functions slow with decreasing temperatures. Consideration should be given to whether there is an actual decrease in treatment effectiveness during the winter, or if effluent values are more of a reflection of influent loadings as described above. For instance, it was noted that at both sites, removal rates correlated positively with loadings at the source. Iron removal at Site 2 proved to be most effective during May, June, and July (the three highest flow periods) and lowest during February, which was the lowest flow period. These months also represented the respective extremes in actual iron loadings. Seasonal findings are limited in this paper because both wetlands were relegated to essentially one season's worth of data, but an additional Site 2 trend should be noted. The iron removal rate in December was slightly greater than that in August (Figure 3), a contradiction to "expected" seasonal wetland performance, but December's source loading was also greater than in August.

The state of iron, ferrous ($Fe^{2+}$) or ferric ($Fe^{3+}$) also plays a role in iron removal efficiency in wetlands. The mentioned anomolous May and June data for Site 1 resulted from physical alterations to the wetland that routed the mine drainage beneath the substrate (rather than across the substrate) of the initial collection pond. During this period, anaerobic conditions in the substrate tended to convert the $Fe^{3+}$ to $Fe^{2+}$ in the mine flow, essentially shuting down iron removal and in some instances actually leading to increases in total iron under these conditions. $Fe^{2+}$ constituted a very minor portion of the total iron load at the discharge of this wetland during the other sample months (under aerobic conditions) and the wetland was extremely effective at removing iron from the drainage.

Site 2 characterizes a "complete" wetland as noted in the introduction, in that iron removal and pH increases are shown in the same wetland without the use of a limestone substrate (Table 4). The compost utilized as substrate in this wetland has an approximate neutralization potential of 3.5% $CaCO_3$ equivalent which cannot account for the changes seen in pH and alkalinity values. The results in Table 4 can be attributed to the design of the system which utilizes an initial, iron oxidizing pond followed by subsurface flows and anaerobic conditions in cells 1 and 2. Sulfate reducing bacteria have demonstrated the ability to cause dramatic changes in constructed wetland substrate water quality (Hedin et al. 1988) and likely are responsible for the quality improvements shown here. Further discussions concerning the effectiveness of this design will be presented in future preceedings, but at this time it is important to document that the Site 2 wetland has been consistently functioning at the level shown in Table 4 for 14 months.

## Discussion

The above results indicate that wetland sizing criteria are of little consequence if regard is not given to the relationship between pH, $Fe^{2+}$, $Fe^{3+}$, alkalinity, acidity, and sulfate. Of necessity, there is a maximum iron load where removal efficiency decreases, and below that maximum value, removal rates must be limited by influent values; i.e., there is a greater opportunity to precipitate more iron at a higher loading than at a lower loading value. A wetland cannot remove 25 gdm if the source value is less than 25gms. Flow manipulation can lead to dominant aerobic or anaerobic conditions in constructed wetlands. Both conditions coexist in any wetland, constructed or natural, and probably play a critical role in both the rapid precipitation of $Fe(OH)_3$ and in the decreasing efficiency of iron removal with decreasing loads in the later stages of wetland treatment.

Typically, $Fe^{2+}$ makes up the vast majority of the total iron load at the actual source of mine drainage. Exposure to the atmosphere, retention time, and a moderately high pH (as seen at Site 2, pH = 5.7) leads to the rapid oxidation and precipitation of iron as an oxyhydroxide ($Fe(OH)_3$). Particles of $Fe(OH)_3$ are commonly positively charged, and upon contact with negatively charged clay particles or organic colloids, tend to become neutralized. These uncharged aggregates can then join to form rapidly settling precipitates (Wetzel 1975) which are generally conspicuous in wetlands receiving mine drainage. The combination of high concentrations of organic matter found in *Typha* -dominated areas, and high iron loads, readily encourages a system where large aggregates of iron floc form and precipitate. This phenomenon was seen in the Site 1 TTAA/AATT comparisons. At lower loading rates, the potential for rapidly combining $Fe(OH)_3$ precipitates to form decreases.

Most remaining iron at the discharge point of these wetlands is in the $Fe^{2+}$ state. The first pond ($125m^2$) in the Site 2 system removes an average of almost 2,000 gms/day of iron from the drainage, but the following two cells in total (totalling $360m^2$) remove only an additional average of 250 gms/day of iron from the flows. Within the substrate where high organic matter exists, and at the sediment-water interface of wetlands, reducing conditions prevail and tend to control the quality of effluent in the wetland. Ferrous iron can form a strong soluble complex with dissolved organic matter (Gjessing 1964) and therefore maintain a measurable level of iron in even a typical (oxidizing) constructed wetland. This reaction is even more pronounced in subsurface flow designs, and may be one reason why little iron was removed under anaerobic conndditions as noted in the May and June data at Site 1.

Seasonally, biological treatment efficiency decreases with decreasing temperatures. However, if the primary removal of iron is based on oxidation (with precipitation accelerated by the presence of organic matter) winter wetland efficiency should not decrease to any great degree when compared to summer values. Oxygen solubility increases with decreasing temperatures and organic matter is in abundance in the fall and winter in a system dominated by persistent emergents (*Typha*). The finding here that seasonal trends are not evident and in fact that some winter removal rates exceed summer removal rates, adds credence to the argument that colder temperatures may not significantly affect wetland iron removal efficiency.

Figure 3. Total iron removed in gdm, Site 2.
August 1988 = 1 through September 1989 = 14

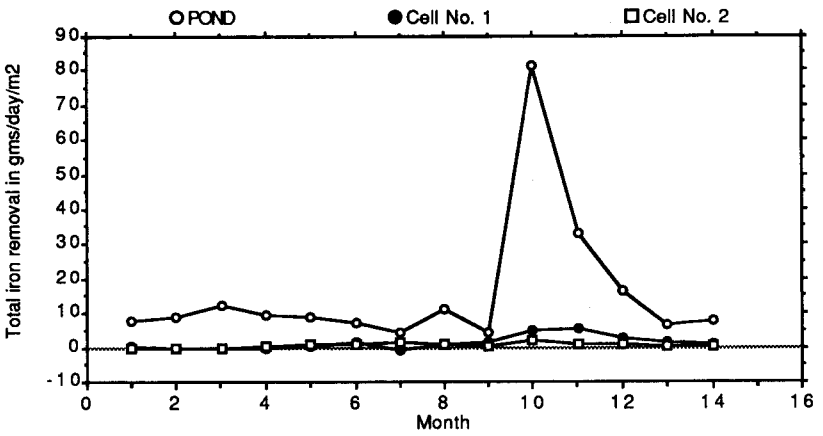



Table 4. Typical water quality at Site 2.

| | pH | Alkalinity* | Acidity* | Sulfate | Fe2+ | Fe(tot) | Mn | Al |
|---|---|---|---|---|---|---|---|---|
| source | 5.7 | 28 | 145 | 970 | 135 | 180 | 20 | <1 |
| pond outlet | 3.0 | 0 | 125 | 825 | 32 | 34 | 19 | <1 |
| cell 1 outlet | 6.7 | 85 | 0 | 650 | 12 | 15 | 18 | <1 |
| cell 2 outlet | 6.9 | 145 | 0 | 580 | 6 | 8 | 18 | <1 |

*Alkalinity and acidity = $CaCO_3$ equivalent, others in mg/L

On an average annual basis, it appears that these wetlands are capable of removing approximately 15 gdm of iron. While this value may eventually be supported, at this time it should at best be used as a general "rule-of-thumb" in design. Accurate, seasonal baseline data of a mine discharge regarding both quality and quantity is critical if any sizing criteria at all is to be employed. Maximum iron loads should be multiplied by a margin of safety (eg., 25%) in the designing of treatment wetlands.

## Conclusions

Aristotle once said that "We cannot expect a degree of accuracy that is not inherent to our subject." Unfortunately, this statement is presently appropriate to definitive sizing criteria for constructed wetlands. The presented information has proven valuable in designing wetlands to treat AMD, but is little more than a starting point in the development of a "guaranteed" design manual. There are numerous avenues of research that are required to complement and confirm the data presented here.

Generally, seasonal wetland performance requires more research, with an excellent starting point being the monitoring of substrate temperature. Monitoring substrate temperature is especially important in understanding the mechanisms of treatment in designed, reducing wetlands that depend on bacteria for their effectiveness. Biomass balances within the wetland system must also be considered if treatment is based on a bacterially mediated system. High density stands of emergents such as *Typha* are commonly used in constructed wetlands and probably provide adequate organic material to the wetlands on a year-round basis. Low pH values in mine drainage usually negatively affect iron removal and should also be given more consideration in future work. The hypothesis that iron removal efficiency is more of a function of loading than seasonal temperature variations need also be examined.

It must also be stated that all of these concerns approach the aspect of design and sizing criteria from a rather simplistic approach. It is much more likely that there are significant interactions between numerous considerations that influence the efficiency of a wetland treatment system. If accurate design models are to be developed, they will be multidimensional and will address the issue of substrate depth equally with source iron loads. An example of this complexity follows.

Although data must be presented in terms of loadings and/or gdm for purposes of comparison, simple loading to loading comparisons may also be misleading. The earlier example of 1mg/L of iron at a flow of 100L/min and a second flow of 1L/min with an iron concentration of 100mg/L led to identical loads and reportedly identical conclusions. However, experience has shown that this discrepancy between flows would require two distinct wetland designs. The lesser flow/higher iron discharge could be treated more efficiently and with less treatment area than the higher flow/lower iron discharge regardless of equal loading values primarily because of the ease of manipulating smaller volumes of water.

These statements are not intended to discourage those looking for a definitive design manual. The effective treatment of AMD in constructed wetlands is consistently improving. The mere fact that the above questions and considerations are being raised is proof enough that the field of wetland treatment has progressed significantly from questioning simply "how to remove iron from AMD."

## Literature Cited


Gjessing, E.T. 1964. Ferrous Iron in Water. Limnol. Oceanogr. 9: 272-274.

Hedin, R.S., D.M. Hyman, and R.W. Hammack. 1988. Implications of Sulfate-Reduction and Pyrite Formation Processes for Water Quality in Constructed Wetlands: Preliminary Observations. BuMines IC 9183: 382-388.

Kepler, D.A. 1986. Manganese Removal From Mine Drainage by Artificial Wetland Construction. p.74-80. Proceedings of the 8th Annual National Abandoned Mine Lands Conference, Billings, MT

Stevens, S.E., Jr., K.A. Dionis, and L.R. Stark. 1989. Manganese and Iron Encrustation on Green Algae Living in Acid Mine Drainage. Unpublished Data. 19p.

Wetzel, R.G. (Ed.) 1975. Limnology. Sanders College Publishing, Philadelphia, PA. 743p.

# THE USE OF BACTERIAL SULFATE REDUCTION IN THE TREATMENT OF DRAINAGE FROM COAL MINES[1]

Pamela E. McIntire and Harry M. Edenborn[2]

Abstract. Bacterial sulfate reduction is a naturally-occurring process that proceeds only in the absence of oxygen and in the presence of sufficient organic carbon and sulfate. The net by-products of sulfate reduction, hydrogen sulfide and bicarbonate, can precipitate metal sulfides and neutralize acidic water, respectively. An experimental wetland was designed and built by the Bureau of Mines to maximize contact between mine drainage and the anaerobic zone of the organic substrate, where sulfate reduction takes place. Preliminary results from the wetland were encouraging. Iron concentrations decreased from 237 to 27 ppm, pH increased from 2.9 to 6.5, and alkalinity increased from 0 to 1077 ppm when the mine drainage was forced through the anaerobic zone. The observed improvement in water quality was due to both the inherent chemical characteristics of the organic substrate as well as bacterial activity. Bacterial sulfate reduction rates in the organic substrate ranged from 2 to 600 nmol $cm^{-3}$ $day^{-1}$. Sulfate reduction rates are high enough to significantly affect the water quality of acid mine drainage. The sulfate-reducing bacteria effectively precipitate many heavy metals as insoluble sulfides and may also be useful in treatment processes designed to improve the water quality of metal mine drainage.

## Introduction

At over 300 mine sites in the bituminous coal region of the eastern United States, acid mine drainage (AMD) is being treated biologically in constructed wetlands (Hedin 1989). In general, the processes at work in these systems are aerobic. The oxidation of ferrous iron to ferric iron and the subsequent precipitation of iron oxyhydroxide floc, for example, are dominant processes:

$$Fe^{2+} + 0.25O_2 + 1.5H_2O \text{ --> } FeOOH \text{ (solid)} + 2H^+ \quad (1)$$

Ferrous iron tends to autoxidize in aerated solutions at pH values greater than 6, while in more acidic water naturally-occurring bacteria catalyze the reaction (Singer and Stumm 1970). Although iron oxidation and hydrolysis processes are effective at removing much of the iron from AMD, these processes do nothing to help raise the pH of the water or decrease the acidity. In fact, the pH of water can be lowered by these reactions (Equation 1). Many constructed wetlands with circumneutral pH and iron-contaminated inflow water actually produce water with a lower pH (Brodie et al. 1988; Kepler 1990).

Ironically, bacterial processes capable of increasing the pH and alkalinity of AMD entering constructed wetlands are already found there, but current wetland designs do not take advantage of them. Probably the most useful of these processes for treating AMD is bacterial sulfate reduction, a naturally-occurring process that proceeds in many environments in the absence of oxygen and in the presence of suitable organic substrates and sulfate. Sulfate-reducing bacteria use organic carbon and sulfate in the process of anaerobic respiration:

$$2CH_2O + SO_4^{2-} \text{ ---> } H_2S + 2HCO_3^- \quad (2)$$

The reaction has promise in the treatment of acid and metal-contaminated mine waters because the by-products of the reaction, hydrogen sulfide and bicarbonate, can be used to precipitate many metals and raise the pH of the water, respectively. When AMD percolates into stream bottoms or constructed wetlands containing abundant organic matter and debris, bacterial sulfate reduction can proceed. Wetlands constructed to treat AMD are generally built to include a large amount of organic matter, and anaerobic conditions develop rapidly within

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]Pamela E. McIntire, Research Biologist, and Harry M. Edenborn, Supervisory Research Biologist, are with the Biotechnology Section, Environmental Technology Group, U.S. Bureau of Mines, Pittsburgh Research Center, P.O. Box 18070, Pittsburgh, PA 15236.

that material. Calculations using data from natural wetlands and constructed wetlands have suggested that the use of a sulfate reduction treatment system is theoretically feasible (Hedin, Hammack, and Hyman 1989). In this study, we present data collected from a constructed wetland designed to incorporate bacterial sulfate reduction as a treatment process.

## Materials and Methods

An experimental wetland was constructed at the Friendship Hill National Historic Site, located in Fayette County, PA. The inflow to the wetland comes from a stream severely polluted with water draining from an abandoned drift mine 1 km upstream of the wetland. The inflow water has a pH of about 2.5, total iron concentration of 50 to 250 ppm, and a sulfate concentration of 1000 to 2500 ppm.

The constructed wetland was built in the summer of 1988 and was divided into three cells (Fig. 1a). The first cell is a holding pond that does not treat the AMD. The remaining two cells are each divided into three different treatment lanes separated by Fiberglas sheeting. In cell two, the AMD flows across the surface of the organic substrate, but in cell three, subsurface infusion pipes can be used to carry AMD down to the bottom of the organic substrate layer (Fig. 1b). Individual ball valve controls are used to adjust AMD water flow into each lane.

The six treatment lanes were each constructed with a 15 cm (6 in) layer of gravel on the bottom, covered by a 46 cm (18 in) layer of spent mushroom compost obtained from a mushroom farm near Worthington, PA. The basal gravel layers in the cells consisted of crushed limestone in lanes B2, B3, C2, and C3, and noncalcareous river gravel in lanes A2 and A3. Whole cattail plants were transplanted to all cells and the compost was flooded with 8 to 20 cm (3-8 in) of AMD water.

Surface water samples were collected from nine locations in the wetland biweekly for eight months. Measurements of pH in the field were made using an Orion SA230 portable pH meter. Alkalinity (as $CaCO_3$ equivalents) was determined by titrating a 50-ml sample with 0.2 N $H_2SO_4$ to pH 4.8. Acidity was measured on a 50-ml sample that was first boiled with 1 ml of 30% $H_2O_2$ and then titrated with 0.1 N NaOH to pH 8.3. Total iron concentrations were determined for acidified water samples using an ICP atomic emission spectrophotometer. Sulfate concentrations were determined by barium chloride titration, using thorin as the end point indicator.

Platinum electrode potentials in organic substrate cores taken from the wetland were measured within 20 minutes of core removal. Measurements were made at 4 cm intervals throughout each core. A platinum wire was inserted through a silicone plug located at each sampling interval and a reference electrode was suspended in the surface water overlying the substrate. The wire and electrode were attached to a voltage meter which was standardized using Zobell's solution (Eh = +430 mV) prior to each measurement series.

Sediment sulfate reduction rates and total reduced sulfur concentrations were determined using the method of Herlihy and Mills (1985), which is a modification of the method of Jorgensen (1978). Vertical cores of organic substrate from the wetland were transported to the laboratory within eight hours of their removal. Five to 8 ml subcores were taken from two depths in each core. The subcores were immediately injected with 50 uL of carrier-free $^{35}S$-labeled $Na_2SO_4$ (5 mCi/mL; New England Nuclear). Subcores were incubated at in-situ temperature for 2 to 14 days. Incubation times were chosen to allow 5 to 50% of the labeled sulfate to be converted to sulfide. After incubation, the subcores were frozen and stored at $-4^0$ C prior to analysis. The amount of labeled sulfide formed during incubation was determined by acid distillation of the subcores and the collection of the released hydrogen sulfide in traps containing a solution of 10% NaOH. Total reduced sulfur concentrations reported in this paper consist of the sum of an acid volatile sulfide (AVS) fraction (hydrogen sulfide, bisulfide ion, and iron monosulfide) and a chromium reducible sulfur (CRS) fraction (elemental sulfur and pyrite). Sulfate reduction rates were determined as described by Jorgensen (1978).

Interstitial porewater samples were collected using equilibrators similar to those designed by Hesslein (1976). These equilibrators consist of thick Plexiglas stakes containing wells of approximately 30 ml volume spaced 2 cm apart along

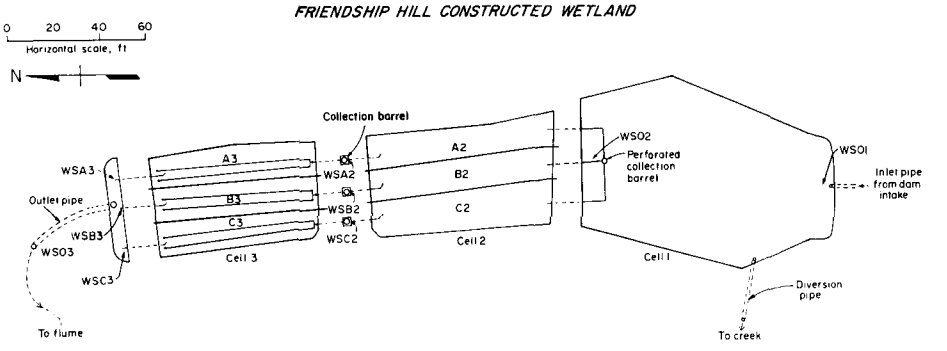



Figure 1a. Plan view of Friendship Hill constructed wetland.

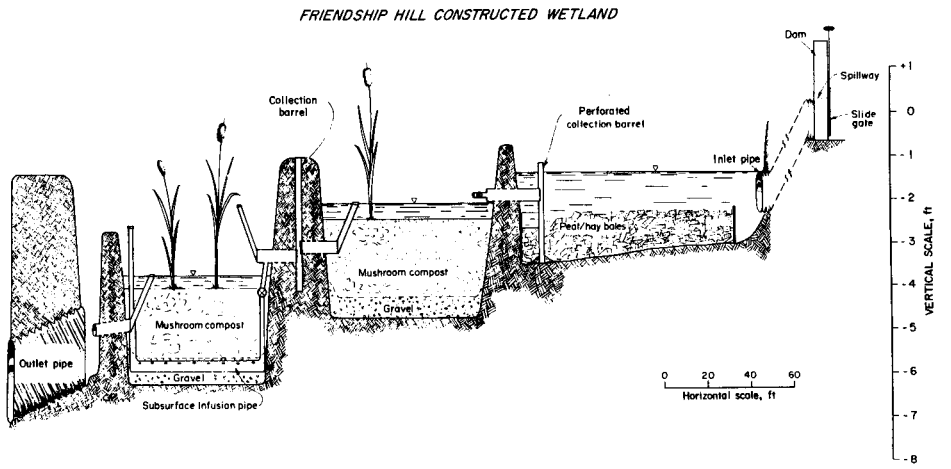



Figure 1b. Typical longitudinal cross-section of the Friendship Hill constructed wetland.

the length of the stake. The wells were filled with deionized and deoxygenated water and covered with a 0.2 um membrane filter sheet. The apparatus was pushed down into the organic substrate of the wetland and allowed to equilibrate with the interstitial water for three to four weeks. Ten ml samples were then extracted from each well and placed in vials containing 1 ml of 2 N zinc acetate. Sulfate concentrations in these samples were determined by liquid chromatography and were used in the calculation of sulfate reduction rates.

The percent volume of water in subcores used to measure sulfate reduction rates was determined by measuring the difference in weight of similar subcores before and after drying, and assuming the density of water to be equal to 1 g $cm^3$.

Laboratory experiments were also done to evaluate potential metal toxicity to sulfate-reducing bacteria. Metals were added as chloride salt solutions to Postgate's medium B for sulfate-reducing bacteria (Postgate 1984). Final metal concentrations ranged from 5 to 380 ppm. The media were inoculated with a mixed culture of sulfate-reducing bacteria isolated from mushroom compost. The bacteria were incubated at 30° C for four weeks and the growth or lack of growth was noted. After four weeks, the media were filtered and metal concentrations in the supernatant were determined.

## Results and Discussion

### Experimental Wetland

Study of the Friendship Hill constructed wetland showed that increasing contact between AMD and the anaerobic zone of the organic substrate improved the quality of the water leaving the wetland. When subsurface infusion pipes were turned on and the AMD was forced down into the anaerobic substrate, an immediate increase in pH and alkalinity and a net removal of total iron were observed (Fig. 2, 3, and 4). During this time, the pH of the outflow water increased to as high as 6.5, the alkalinity increased up to 500 ppm, and the total iron concentration decreased from 200 to 20 ppm. In contrast, little improvement in water pH or alkalinity was observed when the AMD flowed predominantly across the surface of the organic substrate. When the subsurface infusion pipes were turned off, the observed improvements in water quality disappeared.

It is probable that much of the initially observed improvement in water quality was due to the chemical characteristics of the spent mushroom compost used to construct the wetland. The neutralization potential capacity of fresh mushroom compost is about 3.5% $CaCO_3$ equivalents (Hammack and Hedin 1989). If we consider that approximately 35 tons of compost were placed in each lane containing a subsurface drainage system, and we know that approximately 3.3 x $10^5$ liters of AMD of 1000 mg $L^{-1}$ acidity passed through the compost during 31 days of operation, approximately 30% of the total initial alkalinity of the compost would have been exhausted. If all of this alkalinity could be used to treat AMD before it left the wetland, chemical neutralization might be expected to continue for three months. However, this assumes that the water flows through the wetland in such a way that all compost is contacted equally and no portion of the compost is exhausted of its neutralization potential before any other. In reality, flow patterns are likely to occur that use up the neutralization potential of the compost along major flow paths and result in less effective neutralization of the AMD. In addition, bacterial sulfate reduction can be expected to contribute neutralization potential to the compost with time, adding to the total pool of alkalinity. The potential significance of this biological process is addressed later in this section.

The first time that underflow drains were turned on in the Friendship Hill wetland, there were large differences in the chemical composition of water flowing from the lanes containing either calcareous or noncalcareous gravel. The pH, alkalinity, and total iron concentration in the treated water were all improved in the subsurface flow lanes, but especially in those containing calcareous gravel (Fig. 2, 3, and 4). This suggests that dissolution of the limestone initially played a major role in improving the water quality. However, the second time the subsurface infusion

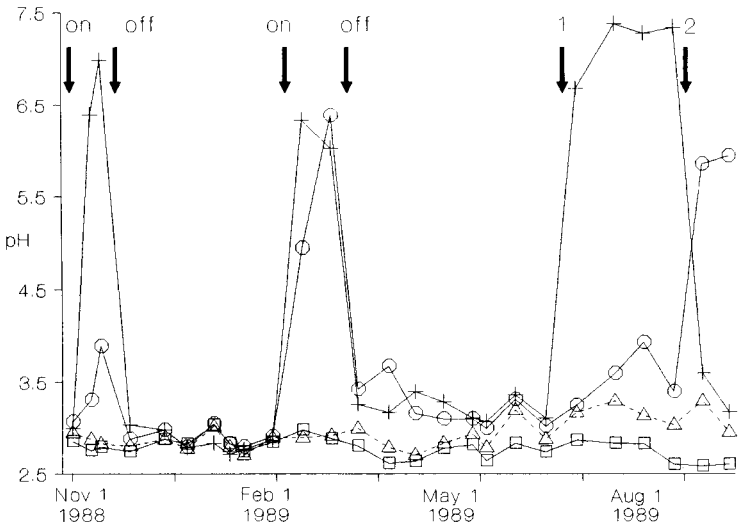



Figure 2. The pH of water in the Friendship Hill constructed wetland system during the study period. Symbols indicate: untreated water flowing into the wetland (open squares); water leaving subsurface flow only cell in wetland (open triangles); water leaving subsurface flow wetland cell containing noncalcareous gravel (open circles); water leaving subsurface flow wetland cell containing limestone gravel (plus signs). Arrows mark times when the subsurface system was turned on and off. Symbols "1" and "2" denote when underflow dams were installed in the calcareous and noncalcareous gravel lanes, respectively.

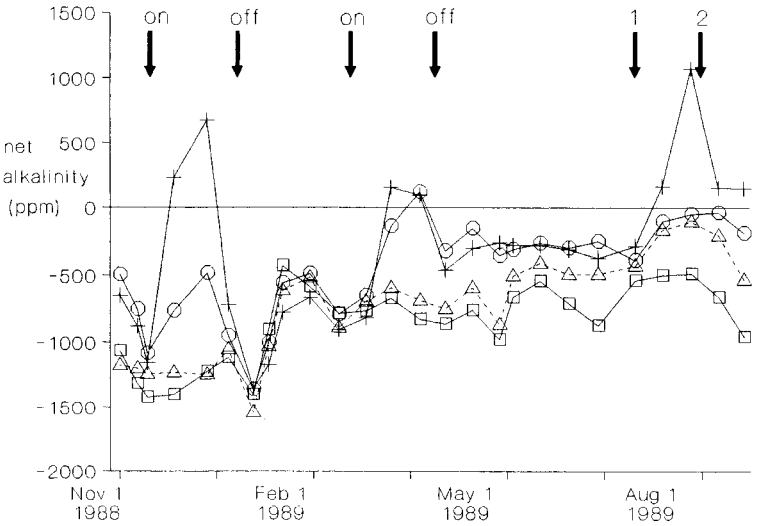



Figure 3. Alkalinity (as ppm CaCO3) of water in the Friendship Hill wetland system during the study period. Symbols are the same as described for Fig. 2. Acidity is considered negative alkalinity.

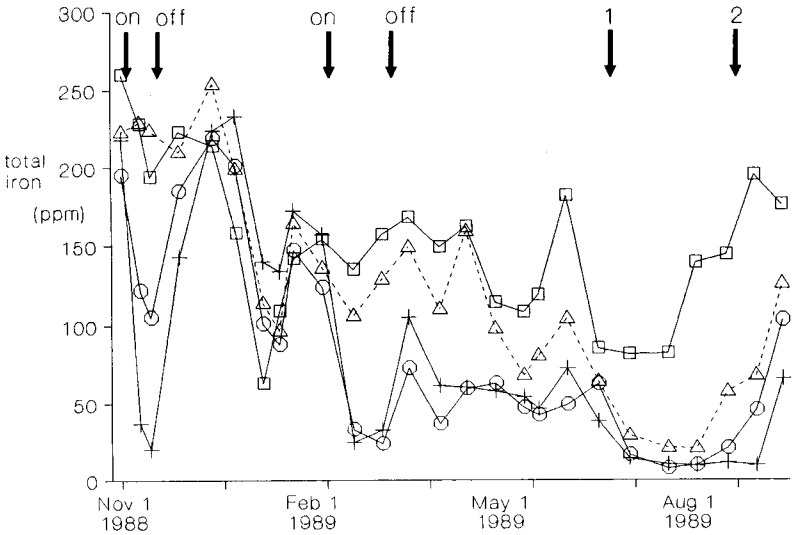



Figure 4. Total iron concentrations in water in the Friendship Hill wetland system during the study period. Symbols are the same as described for Fig. 2.

pipes were turned on, the noncalcareous gravel lanes were just as effective as those containing limestone gravel. Although it was hoped that the anaerobic environment established in the organic substrate of the wetland would prevent the armoring of the gravel with iron oxyhydroxides and/or other deposits, it appeared that this probably occurred anyway. This explanation seems reasonable considering that the surface water introduced to the gravel layers contained both oxygen and ferric iron. These data imply that there is no significant acid neutralization advantage in using limestone gravel in the construction of wetlands employing anaerobic systems. This is in agreement with the laboratory results of Hammack and Hedin (1989).

Soon after the subsurface infusion pipes were turned on for the second time at Friendship Hill, they became clogged with iron oxyhydroxide floc and organic debris. An alternative method of introducing AMD to the anaerobic regions of the wetland without the use of pipes was then devised. This method involved the installation of a sheet of Plexiglas in a test lane of the wetland, placed so that water in the lane was dammed behind the sheet and was forced to flow downward and beneath the dam. This is somewhat analogous to the hay bale dams found in some wetlands. Water downstream of these dams often has a higher pH (Hedin et al. 1988; Stark et al. 1988), although this is in part due to sulfate reduction activity occurring within the hay bales themselves. In the Plexiglas dam method used in the Friendship Hill wetland, the dammed water was forced to make contact with the basal gravel and anaerobic compost layers. Initially, water quality improvements were observed again (Fig. 2, 3, and 4); however, after a few weeks of operation the water quality was no longer affected positively by the dam system, presumably due to the exhaustion of alkalinity in the substrate and the disruption of the anaerobic microbial community.

Measurements of pH in the porewater of the organic substrate at the Friendship Hill wetland showed that the pH generally remained between 6.0 and 7.0, even though the surface water usually had a pH of 2.5 to 3.5. This high porewater pH most likely reflects the influence of alkaline substances present in the compost as well as those produced by the natural bacterial population. When the Plexiglas dam was used, the increased flow of AMD through the relatively small volume of compost located on either side of the dam exhausted the neutralization potential of the compost. Porewater pH values deep in the compost above the dam after its operation were all low (3.2 to 4.5), supporting this hypothesis.

Sulfate Reduction Rates

During the course of this study, the redox potentials measured below 15 cm deep in the organic substrate of the constructed wetland decreased steadily (Table 1). This is an indication of the gradual exhaustion of oxidized chemical species and the accumulation of reduced species.

Table 1. Redox potentials determined with a platinum electrode at depths below 15 cm in the substrate of the Friendship Hill wetland (n = 6 to 20).

| Date | Pt Electrode Potential (mV) Min | Max |
|---|---|---|
| 11/01/88 | 32 | 449 |
| 11/15/88 | -18 | 238 |
| 12/16/88 | -84 | 33 |
| 01/31/89 | -46 | -10 |
| 05/31/89 | -108 | -9 |

Sulfate reduction rates measured in the organic substrate of the Friendship Hill wetland were highly variable, ranging from 2 to 600 nmol

$cm^{-3}$ $day^{-1}$ (Fig. 5). This reflects, in part, the heterogenous nature of the organic compost, which contains materials such as straw, manure, and corncobs. Maximum detectable rates of sulfate reduction activity during the study period are comparable to rates found in coastal marine sediments (0.4 to 3000 nmol $cm^{-3}$ $day^{-1}$; Edenborn et al., 1987). Sulfate reduction rates remained consistently low throughout the winter but increased as the summer progressed, consistent with the predicted overall increase in biological activity with seasonally higher temperatures (Fig. 5). The concentration of total reduced sulfur compounds also increased in the wetland substrate with time (Fig. 6). This confirms that the by-products of bacterial sulfate reduction do accumulate in the wetland substrate and can probably be used to affect the quality of AMD under properly engineered conditions.

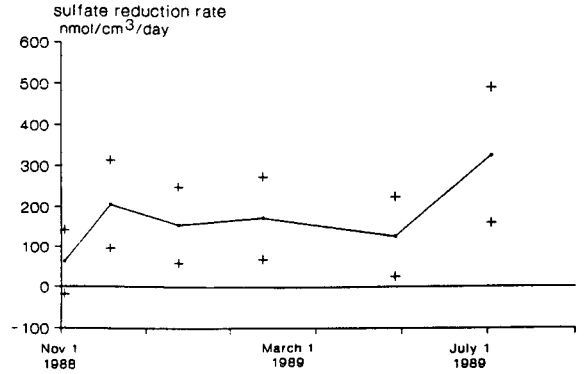



Figure 5. Sulfate reduction rates measured in the organic substrate of the Friendship Hill wetland. $x \pm SD$ is shown for each sampling date (n = 10 to 40).

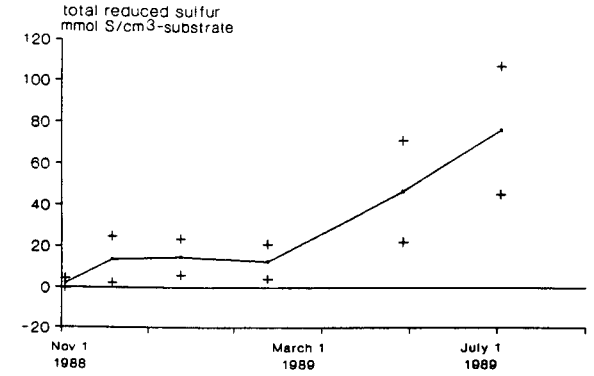



Figure 6. Total reduced inorganic sulfur concentrations in the organic substrate of the Friendship Hill wetland. $x \pm SD$ is shown for each sampling date (n = 10 to 40).

It is of interest to consider the potential role of bacterial sulfate reduction in the neutralization of AMD flowing through a wetland constructed to incorporate this process as a treatment method. If the maximum sulfate reduction rates measured in the Friendship Hill wetland (600 nmol $cm^{-3}$ $day^{-1}$) were maintained throughout the entire volume of organic substrate ($4.16 \times 10^4$ L) for six months, and AMD chemical composition and flow rates were as described previously (acidity = 1000 mg $L^{-1}$, flow rate = 7.5 L $min^{-1}$), 23% of the AMD flowing through the wetland during that time could be neutralized by bacterial activity alone. This percentage could be increased significantly by enhancing the sulfate reduction rate or by lowering the flow rate of AMD through the system. Even low bacterial sulfate reduction rates will lengthen the period of time that AMD can be fully neutralized, until the alkaline components in the substrate are exhausted. It will require additional study to determine if higher rates of sulfate reduction can be stimulated and maintained by improving wetland design and construction practices.

If practical sulfate reduction treatment systems can be developed, they may also be useful in the treatment of wastewater containing heavy metals. Laboratory experiments demonstrated that sulfate-reducing bacteria were active in solutions containing high concentrations of metals (Table 2). The bacteria effectively decreased the concentrations of many of the metals that were added to their growth media, forming insoluble precipitates with Ni, Cu, Zn, Cd, and Pb. These preliminary data indicate that biological systems may have practical application to the treatment of mine effluents containing selected heavy metals that precipitate with hydrogen sulfide.

Table 2. Metal toxicity tests on a mixed culture of sulfate-reducing bacteria.

| | Metal Concentration (ppm) | | | |
|---|---|---|---|---|
| Metal | Toxicity | After Inoculation[1] | After Growth[2] | Percent Removal |
| Cd | 104 | 35 | 0.1 | > 99 |
| Cu | > 14 | 14 | < 0.1 | > 99 |
| Mn | > 223 | 223 | 60 | 73 |
| Ni | > 211 | 211 | 0.4 | > 99 |
| Pb | > 5 | 5 | 0.5 | 89 |
| Zn | 310 | 194 | 2 | 99 |

[1]Metal concentration immediately after inoculation.
[2]Metal concentration at the end of the experiment.

## Conclusions

Although it has been known for several years that bacterial sulfate reduction takes place in wetlands constructed for the treatment of acid mine drainage, the rate measurements reported here represent the first to be published in the literature to our knowledge. The study has found that sulfate reduction occurs at relatively high rates, but that these rates are highly variable in the heterogeneous environment of the wetland organic substrate, and they vary seasonally. It is apparent from laboratory and field studies that this process may prove to be useful in the treatment of acid and metal mine drainage if properly engineered systems can be designed. However, the problem of designing and constructing an effective system incorporating bacterial sulfate reduction for the long-term treatment of mine drainage remains to be solved.

## Literature Cited


Brodie, G.A., D.A. Hammer, and D.A. Tomljanovich. 1988. Constructed wetlands for acid drainage control in the Tennessee Valley. p. 325-331. In 1988 Mine Drainage and Surface Mine Reclamation, Volume I: Mine Water and Mine Waste, BuMines IC 9183.

Edenborn, H.M., N. Silverberg, A. Mucci, and B. Sundby. 1987. Sulfate reduction in deep coastal marine sediments. Mar. Chem. 21:329-345.

Hammack, R.W. and R.S. Hedin. 1989. Microbial sulfate reduction for the treatment of acid mine drainage: a laboratory study. In Proceedings of the ninth annual West Virginia surface mine drainage task force symposium.

Hedin, R.S. 1989. Treatment of coal mine drainage with constructed wetlands. p. 349-362. In Wetlands Ecology and Conservation: Emphasis in Pennsylvania. S.K. Majumdar, R.P. Brooks, F.J. Brenner, and R.W. Tiner, Jr. (eds.). The Pennsylvania Academy of Science, Easton.

Hedin, R.S., R. Hammack, and D. Hyman. 1989. Potential importance of sulfate reduction processes in wetlands constructed to treat mine drainage. p. 508-514. In Constructed Wetlands For Wastewater Treatment. D.A. Hammer (ed.). Lewis Publishers, Chelsea, MI.

Herlihy, A.T. and A.L. Mills. 1985. Sulfate reduction in freshwater sediments receiving acid mine drainage. Appl. Environ. Microbiol. 49:179-186.

Hesslein, R.H. 1976. An in situ sampler for close interval porewater studies. Limnol. Oceanogr. 21:912-914.

Jorgensen, B.B. 1978. A comparison of methods for the quantification of bacterial sulfate reduction in coastal marine sediments. I. Measurements with radiotracer techniques. Geomicrobiol. J. 1:11-27.

Kepler, D.A. 1990. Wetland sizing - design and treatment effectiveness. In Proceedings of the Mining and Reclamation Conference and Exhibition, Charleston, West Virginia.

Postgate, J.R. 1984. The Sulphate-reducing Bacteria. Cambridge University Press, New York, p. 208.

Singer, P.C. and W. Stumm. 1970. Acidic mine drainage: the rate-determining step. Science 167:1121-1123.

Stark, L.R., R.L. Kolbash, H.J. Webster, S.E. Stevens, K.A. Dionis, and E.R. Murphy. 1988. The Simco #4 wetland: biological patterns and performance of a wetland receiving mine drainage. p. 332-344. In 1988 Mine Drainage and Surface Mine Reclamation, Volume I: Mine Water and Mine Waste, BuMines IC 9183.

This page intentionally left blank.

# METAL REMOVAL EFFICIENCIES FROM ACID MINE DRAINAGE IN THE BIG FIVE CONSTRUCTED WETLAND[1]

by

Thomas R. Wildeman, Steven D. Machemer,
Ronald W. Klusman, Ronald R. H. Cohen, and Peter Lemke[2]

Abstract. An experimental constructed wetland in the Idaho Springs-Central City mining district of Colorado is providing data for metal removal efficiencies from acid mine drainage. The wetland is designed to passively remove the heavy metals found in the metal mine drainage flowing from the Big Five Tunnel. A number of different substrate compositions in the wetland have been used. The decrease of metal concentrations and the increase in pH is related to the rate of drainage flow through the wetland. For Fe and Zn, decreases in concentrations generally vary inversely with flow rate as do increases in pH. This information can be used in determining the maximum loading rate for any given metal removal efficiency. Mass removal data (mg/day/$m^2$) indicate that there are some tendencies for greater metal removal rates and pH increases during the summer season than for the winter. No obvious relationships are found between mass removal data and flow rate for Fe, Zn, Cu, and Mn.

Additional Key Words: constructed wetland, low pH, heavy metals pollution.

## Introduction

Research efforts over the past few years have contributed a great deal of knowledge towards designing constructed wetlands for the treatment of acid mine drainage (Hammer 1989). The Big Five wetland is one of the few built specifically to remove heavy metals from acid mine waters. It is located in Idaho Springs, Colorado, a region which produced a significant amount of precious metal ores around the turn of the century. Mine drainages in the region typically have low pH and high metal concentrations.

The mine drainage flowing from the Big Five Tunnel was chosen to investigate the treatment of acid mine drainage with a constructed wetland system (Howard et al. 1989a). The design of the system was based on discussions with and findings of experienced investigators (Holm pers. comm., Kleinmann 1983 and pers. comm., Hiel and Kerins 1988). The system was built for a number of reasons including: to determine the fate of the metals in the system, to identify the organic substrates that function best in such systems, and, the primary focus of this paper, to determine the effectiveness of the system in decreasing the metal concentrations of the mine drainage.

In 1988, the clogging of the mine drainage flow lines inadvertently allowed observations of much larger increases in pH and larger decreases in metal concentrations with reduced flow. These observations led to further association of the metal removal efficiencies and increases in pH with different loading rates.

Initial studies suggested what kinds of metal removal processes may be important in the wetland, such as bacterially mediated sulfate reduction (Wildeman and Laudon 1989). Sulfate reducers were found to be dominant throughout the wetland

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]Thomas R. Wildeman and Ronald W. Klusman are Professors of Chemistry and Geochemistry and Steven D. Machemer is a Graduate Student in the Department of Chemistry and Geochemistry, Colorado School of Mines, Golden, Colorado 80401. Ronald R.H. Cohen is Assistant Professor of Environmental Sciences and Engineering Ecology and Peter Lemke is a Graduate Student in the Department of Environmental Sciences and Engineering Ecology, Colorado School of Mines, Golden, Colorado 80401.

substrates whereas metal-oxidizing bacteria were found only very near the surface (Batal et al. 1989).

The objectives of this paper include (1) examining recent data to determine if processes such as bacterially mediated sulfate reduction are actually occurring, (2) determining if different perspectives on data analysis provide further insights on the metal removal processes taking place and, most importantly, (3) determining if there is a maximum loading rate for which metal removal processes will operate optimally. By relating different loading rates to removal efficiencies, the maximum loading rate for any given metal removal efficiency can be estimated.

## Materials and Methods

Materials, methods and procedures used in building and operating the Big Five treatment system are given in Howard et al. (1989a and 1989b). The treatment system has dimensions of 1.22 m in depth, 3.05 m in width, and 18.3 m in length. The structure is divided into three 6.1 m long cells with an area of 18.6 $m^2$.

Water is piped from the tunnel portal to each of the cells through PVC piping which is fitted with valves to control the flow to each individual cell. Standpipes set at an overflow height of about 1 m act as drains in the cells.

Rock boxes were constructed at the upstream end of each of the cells to allow the mine drainage to contact as much cross-section of the organic substrate as possible. The boxes were filled with 10-15 cm rounded cobbles. The boxes were to force the flow downward into the substrate. These rock boxes, however, did not work well because the accumulation of metal hydroxides appeared to clog the downward flow of water into the substrate.

The cells contain an organic substrate to a depth of about 1 m. Cell A was filled with fresh mushroom compost, which consists of approximately 50 percent animal manure and 50 percent barley mash wastes from a local brewery. Cell B received a mixture of equal parts of peat, aged steer manure, and decomposed wood shavings and sawdust. Cell C was lined to a depth of 10-15 cm with 5-8 cm limestone rock, then filled to a 1 m depth with the same organic mixture as Cell B. Cattail, sedge, and rush species were transplanted from similar locations to each of the cells.

Sampling of the mine drainage water and of the cell outputs was done about twice monthly with more frequent samplings in the summer season. Water samples were filtered in the field and preserved with $HNO_3$. At least two duplicate water samples were taken monthly. Duplicate output water samples were taken from the same sample during the filtering process and duplicate mine drainage water samples were taken as separate samplings from the mine drainage. For reporting purposes, data for the duplicates have been averaged. The relative difference between duplicates for metal determinations are generally within a few percent and within one percent or less for sulfate values.

Metal analyses were done by flame atomic adsorption and sulfate determinations were made gravimetrically by precipitation of $BaSO_4$.

## Water Quality of the Big Five Mine Drainage and Wetland Outputs

The most abundant heavy metals in the Big Five Tunnel mine drainage are Fe, Mn, Zn, and Cu. The analytical results of these metal concentrations and of sulfate in the cell output waters and mine drainage show promising results for the Big Five treatment system (Table 1). Metal concentration decreases range from basically none for Mn to essentially complete removal for Cu. Decreases in concentrations for Fe of up to 63 percent and for zinc of up to 100 percent are found. Decreases in sulfate concentrations of about 10% are typical in Cell A.

## Relationships Between Loading Rates and Metal Concentrations

After about 24 months of operation, the mushroom compost in Cell A shows a much better capability to remove metals and raise the pH than the substrates in the other two cells at similar flow rates (Howard et al. 1989b, Wildeman and Laudon 1989). Because of the higher metal removal efficiency and effluent pH, the focus in this paper is on the results from Cell A.

The Big Five wetland is operated as closely as possible to a passive, natural system. Consequently, events of nature such as freeze-thaw cycles cause fluctuation in the removal processes that produce fluctuations in the output concentrations. Because of this, loading rates should be considered as rough guidelines.

Metal removal data for different loading rates were gathered by adjusting flow rates two days to one week prior to sampling in order to allow for the wetland to equilibrate. Retention time is inversely proportional to the flow rate. Thus, at higher flow rates, less time is required for equilibration. A retention time of about 70 hours is estimated for a flow rate of 4 L/min. Table 2 contains the flow rates of mine drainage through Cell A at the time of sampling. In addition, changes in pH (unit increase) and Eh (decrease in mV) are given as well as the amount of decrease in sulfate concentration.

Table 1. Metal and sulfate concentrations (mg/L) and pH in the Big Five mine drainage and Cell A output water are given below. Percentages of the decrease in metal concentration in the Cell A output water relative to the mine drainage is also given. The Cell A input flow rate is given in liters/minute. The area of Cell A is 18.6 $m^2$.

| | Mn | % dec. | Fe | % dec. | Zn | % dec. | Cu | % dec. | $SO_4^{2-}$ | pH | flow rate |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | December 11, 1987 | | | | | | |
| Mine Drainage | 34 | | 32 | | 10.6 | | 1.02 | | 1750 | 2.8 | |
| Output A | 27 | 21 | 18 | 45 | 7.8 | 27 | 0.44 | 57 | 1560 | 4.6 | 4 |
| | | | | | February 13, 1988 | | | | | | |
| Mine Drainage | 28 | | 28 | | 8.2 | | 0.89 | | 1750 | 3.3 | |
| Output A | 27 | 4 | 18 | 36 | 5.9 | 28 | 0.14 | 84 | 1690 | 4.7 | 4 |
| | | | | | May 31, 1988 | | | | | | |
| Mine Drainage | 25 | | 44 | | 8.1 | | 0.75 | | 1500 | 3.0 | |
| Output A | 25 | 0 | 27 | 39 | 5.4 | 33 | 0.03 | 96 | 1330 | 4.3 | 4 |
| | | | | | August 19, 1988 | | | | | | |
| Mine Drainage | 26 | | 37 | | 8.1 | | 0.91 | | 1460 | 2.9 | |
| Output A | 25 | 4 | 20 | 46 | <0.1 | 100 | 0.17 | 81 | 650 | 5.5 | 1.9 |
| | | | | | December 18, 1988 | | | | | | |
| Mine Drainage | 29 | | 38 | | 9.2 | | 0.80 | | 1710 | 3.0 | |
| Output A | 28 | 3 | 31 | 18 | 8.6 | 7 | 0.62 | 23 | 1710 | 3.4 | 4.6 |
| | | | | | February 21, 1989 | | | | | | |
| Mine Drainage | 27 | | 32 | | 9.3 | | 0.56 | | 1860 | 3.0 | |
| Output A | 22 | 19 | 12 | 63 | 4.5 | 52 | <0.01 | 100 | 1690 | 5.1 | 1.1 |
| | | | | | May 6, 1989 | | | | | | |
| Mine Drainage | 30 | | 42 | | 10.4 | | 0.76 | | 1760 | 3.0 | |
| Output A | 33 | -10 | 28 | 33 | 7.8 | 25 | <0.05 | 100 | 1690 | 3.5 | 3.3 |
| | | | | | August 1, 1989 | | | | | | |
| Mine Drainage | 32 | | 43 | | 9.4 | | 0.75 | | 1740 | 3.0 | |
| Output A | 31 | 3 | 39 | 9 | 5.2 | 45 | <0.05 | 100 | 1700 | 4.1 | 1.1 |
| | | | | | December 3, 1989 | | | | | | |
| Mine Drainage | 29 | | 43 | | 9.0 | | 0.58 | | 1700 | 3.0 | |
| Output A | 30 | -3 | 21 | 49 | 1.3 | 14 | <0.05 | 100 | 1690 | 6.0 | 1.1 |

Table 2. Output concentrations, field measurements, flow rates, and mass removal data for Cell A from November 1988 to August 1989. Missing data represents output and/or mine drainage samples which were not collected on those dates.

| Sample Number | OPA-106 | OPA-108 | OPA-201 | OPA-303 | OPA | OPA-404 |
|---|---|---|---|---|---|---|
| Sampling Date | 19-Nov-88 | 01-Dec-88 | 18-Dec-88 | 21-Jan-89 | 02-Feb-89 | 21-Feb-89 |
| Output Concentration | | | | | | |
| Cu (mg/L) | 0.540 | 0.630 | 0.620 | 0.612 | | <0.05 |
| Fe (mg/L) | 36.9 | 34.1 | 31.3 | 31.6 | | 18.9 |
| Mn (mg/L) | 28.7 | 28.7 | 27.9 | 32.7 | | 27.8 |
| Zn (mg/L) | 8.2 | 8.0 | 8.6 | 10.8 | | 4.86 |
| SO4 (mg/L) | 1690 | 1720 | 1710 | 1670 | | 1660 |
| Field Measurements | | | | | | |
| Output pH | 3.3 | 3.4 | 3.4 | 3.3 | 3.3 | 5.1 |
| Output Eh (mV) | 535 | 625 | 665 | 740 | 633 | 540 |
| Output Temp. (C) | 2.0 | 9.0 | 8.1 | 6.7 | 4.0 | 1.5 |
| Input Flow (L/min) | 4.75 | 3.13 | 4.57 | 4.72 | 1.81 | 1.06 |
| Absolute Decrease | | | | | | |
| Cu (mg/L) | 0.56 | 0.28 | 0.18 | 0.30 | | 0.72 |
| Fe (mg/L) | 7.2 | -6.0 | 6.8 | -0.3 | | 19 |
| Mn (mg/L) | 0.5 | 0.9 | 0.7 | -7.5 | | -4.9 |
| Zn (mg/L) | 1.7 | 1.5 | 0.6 | 0.1 | | 4.1 |
| SO4 (mg/L) | 60 | 10 | 0 | 10 | | 120 |
| Change in pH & Eh | | | | | | |
| pH (unit increase) | 0.4 | 0.5 | 0.4 | 0.5 | 0.4 | 2.1 |
| Eh (mV decrease) | 100 | 65 | 18 | 10 | 35 | 180 |
| Percent Decrease | | | | | | |
| Cu (%) | 51 | 31 | 23 | 33 | | 100 |
| Fe (%) | 16 | -21 | 18 | -1 | | 50 |
| Mn (%) | 2 | 3 | 2 | -30 | | -21 |
| Zn (%) | 17 | 16 | 7 | 1 | | 46 |
| SO4 (%) | 3 | 1 | 0 | 1 | | 7 |
| Mass Removal Data | | | | | | |
| Cu (mg/day/m2) | 210 | 68 | 64 | 110 | | 59 |
| Fe (mg/day/m2) | 2700 | -1500 | 2400 | -110 | | 1600 |
| Mn (mg/day/m2) | 190 | 220 | 250 | -2700 | | -400 |
| Zn (mg/day/m2) | 630 | 370 | 210 | 37 | | 340 |
| SO4 (mg/day/m2) | 22000 | 2400 | 0 | 3700 | | 9900 |

| Sample Number | OPA-407 | OPA-504 | OPA-510 | OPA-604 | OPA-608 | OPA-704 |
|---|---|---|---|---|---|---|
| Sampling Date | 05-Mar-89 | 19-Mar-89 | 01-Apr-89 | 16-Apr-89 | 06-May-89 | 21-May-89 |
| Output Concentration | | | | | | |
| Cu (mg/L) | <0.05 | <0.05 | <0.05 | 0.471 | <0.05 | 0.580 |
| Fe (mg/L) | 18.5 | 23.3 | 17.1 | 31.3 | 28.3 | 31.0 |
| Mn (mg/L) | 27.8 | 24.0 | 30.1 | 31.6 | 32.5 | 41.0 |
| Zn (mg/L) | 4.76 | 0.28 | 4.58 | 8.5 | 7.8 | 9.4 |
| SO4 (mg/L) | 1570 | 1530 | 1670 | 1730 | 1690 | 1700 |
| Field Measurements | | | | | | |
| Output pH | 5.5 | 6.5 | 5.4 | 3.9 | 3.5 | 3.8 |
| Output Eh (mV) | 260 | 130 | 465 | 630 | 600 | 600 |
| Output Temp. (C) | 1.8 | 7.1 | 6.5 | 10.0 | 10.0 | 16.5 |
| Input Flow (L/min) | 0.79 | 0.22 | 1.81 | 2.07 | 3.32 | 2.30 |
| Absolute Decrease | | | | | | |
| Cu (mg/L) | | 0.73 | 0.76 | 0.23 | 0.76 | 0.27 |
| Fe (mg/L) | | 17 | 21 | 7.3 | 14 | 17 |
| Mn (mg/L) | | -1.8 | 0.0 | -2.3 | -2.5 | 2.7 |
| Zn (mg/L) | | 8.0 | 5.2 | 1.3 | 2.6 | 1.1 |
| SO4 (mg/L) | | 180 | 80 | 10 | 70 | 20 |
| Change in pH & Eh | | | | | | |
| pH (unit increase) | 2.6 | 3.5 | 2.4 | 1.2 | 0.5 | 0.5 |
| Eh (mV decrease) | 450 | 510 | 200 | 5 | 40 | 15 |
| Percent Decrease | | | | | | |
| Cu (%) | | 100 | 100 | 32 | 100 | 32 |
| Fe (%) | | 42 | 56 | 19 | 33 | 35 |
| Mn (%) | | -8 | 0 | -8 | -8 | 6 |
| Zn (%) | | 97 | 53 | 13 | 25 | 10 |
| SO4 (%) | | 11 | 5 | 1 | 4 | 1 |
| Mass Removal Data | | | | | | |
| Cu (mg/day/m2) | | 12 | 110 | 36 | 200 | 48 |
| Fe (mg/day/m2) | | 280 | 3000 | 1200 | 3600 | 3000 |
| Mn (mg/day/m2) | | -31 | 0 | -370 | -650 | 480 |
| Zn (mg/day/m2) | | 140 | 730 | 210 | 670 | 200 |
| SO4 (mg/day/m2) | | 3100 | 11000 | 1600 | 18000 | 3600 |

Table 2 (continued). Output concentrations, field measurements, flow rates, and mass removal data for Cell A from November 1988 to August 1989. Missing data represents output and/or mine drainage samples which were not collected on those dates.

| Sample Number | OPA-708 | OPA-804 | OPA-807 | OPA-810 | OPA-814 |
|---|---|---|---|---|---|
| Sampling Date | 01-Jun-89 | 16-Jun-89 | 22-Jun-89 | 29-Jun-89 | 05-Jul-89 |
| Output Concentration | | | | | |
| Cu (mg/L) | <0.05 | <0.05 | <0.05 | 0.470 | <0.05 |
| Fe (mg/L) | 26 | 25.8 | 20.6 | 29.8 | 20.6 |
| Mn (mg/L) | 34.2 | 33.6 | 33.5 | 34.4 | 34.3 |
| Zn (mg/L) | 6.3 | 4.44 | 2.15 | 7.4 | 2.79 |
| SO4 (mg/L) | 1680 | 1620 | 1610 | 1680 | 1600 |
| Field Measurements | | | | | |
| Output pH | 4.3 | 4.6 | 5.1 | 3.4 | 6.0 |
| Output Eh (mV) | 380 | 520 | 285 | 630 | 245 |
| Output Temp. (C) | 16.0 | 18.0 | 16.0 | 25 | 22 |
| Input Flow (L/min) | 2.83 | 1.73 | 1.25 | 1.66 | 0.38 |
| Absolute Decrease | | | | | |
| Cu (mg/L) | 0.88 | 0.76 | | 0.24 | 0.71 |
| Fe (mg/L) | 20 | 22 | | 15 | 20 |
| Mn (mg/L) | 0.9 | -4.2 | | -6.7 | -7.1 |
| Zn (mg/L) | 2.3 | 3.2 | | -0.4 | 3.7 |
| SO4 (mg/L) | 70 | 80 | | 20 | 160 |
| Change in pH & Eh | | | | | |
| pH (unit increase) | 1.4 | 1.6 | | 0.4 | 3.0 |
| Eh (mV decrease) | 250 | 130 | | 10 | 350 |
| Percent Decrease | | | | | |
| Cu (%) | 100 | 100 | | 34 | 100 |
| Fe (%) | 43 | 46 | | 34 | 50 |
| Mn (%) | 3 | -14 | | -24 | -26 |
| Zn (%) | 27 | 42 | | -5 | 57 |
| SO4 (%) | 4 | 5 | | 1 | 9 |
| Mass Removal Data | | | | | |
| Cu (mg/day/m2) | 190 | 100 | | 31 | 21 |
| Fe (mg/day/m2) | 4400 | 3000 | | 2000 | 600 |
| Mn (mg/day/m2) | 200 | -570 | | -870 | -210 |
| Zn (mg/day/m2) | 510 | 430 | | -49 | 110 |
| SO4 (mg/day/m2) | 15000 | 11000 | | 2600 | 4700 |

| Sample Number | OPA-818 | OPA-821 | OPA-904 | OPA-907 | OPA-911 |
|---|---|---|---|---|---|
| Sampling Date | 09-Jul-89 | 11-Jul-89 | 14-Jul-89 | 20-Jul-89 | 01-Aug-89 |
| Output Concentration | | | | | |
| Cu (mg/L) | 0.490 | 0.071 | <0.05 | 0.490 | <0.05 |
| Fe (mg/L) | 25.6 | 21.9 | 17.3 | 29.6 | 39.0 |
| Mn (mg/L) | 29.1 | 28.7 | 30.0 | 30.3 | 30.8 |
| Zn (mg/L) | 6.87 | 3.58 | 0.25 | 7.7 | 5.19 |
| SO4 (mg/L) | 1700 | 1640 | 1630 | 1730 | 1700 |
| Field Measurements | | | | | |
| Output pH | 3.2 | 5.5 | 5.7 | 3.4 | 4.1 |
| Output Eh (mV) | 590 | 385 | 210 | 560 | 225 |
| Output Temp. (C) | 20 | 18 | 17 | 18.5 | 16.5 |
| Input Flow (L/min) | 5.25 | 1.13 | 1.17 | 3.89 | 1.06 |
| Absolute Decrease | | | | | |
| Cu (mg/L) | | 0.68 | 0.72 | | 0.75 |
| Fe (mg/L) | | 25 | 24 | | 3.7 |
| Mn (mg/L) | | 2.8 | -0.3 | | 1.4 |
| Zn (mg/L) | | 5.3 | 8.7 | | 4.2 |
| SO4 (mg/L) | | 60 | 110 | | |
| Change in pH & Eh | | | | | |
| pH (unit increase) | 0.2 | 2.6 | 2.7 | | 1.1 |
| Eh (mV decrease) | 70 | 280 | 390 | | 450 |
| Percent Decrease | | | | | |
| Cu (%) | | 91 | 100 | | 100 |
| Fe (%) | | 53 | 58 | | 9 |
| Mn (%) | | 9 | -1 | | 4 |
| Zn (%) | | 60 | 97 | | 45 |
| SO4 (%) | | 4 | 6 | | 2 |
| Mass Removal Data | | | | | |
| Cu (mg/day/m2) | | 60 | 66 | | 62 |
| Fe (mg/day/m2) | | 2200 | 2200 | | 300 |
| Mn (mg/day/m2) | | 250 | -27 | | 110 |
| Zn (mg/day/m2) | | 470 | 790 | | 350 |
| SO4 (mg/day/m2) | | 5300 | 10000 | | 3300 |

Concentration decreases in the output water relative to the input water can be compared to the inlet flow rate, since the mine drainage concentrations are relatively consistent with time making the flow rate roughly proportional to loading (Table 2). Positive values indicate a decrease in concentration, whereas negative values indicate an increase in concentration.

Fe and Zn generally show a greater removal as the flow rate is decreased but Cu and Mn are not easily related. If one tries to chose an optimum flow below which the removal efficiency appears to be more consistent, the results for Cu and Zn provide the best number (Table 2). Below a flow of about 1.5 L/min, Cu is uniformly removed and below this flow appreciable Zn is always removed from the mine drainage. It appears that below 1.8 L/min, Mn is not removed from solution, and in fact, may be redissolved. If this is the case, the processes affecting Mn in a wetland system require further study. If metal removal occurs through sulfide formation, Cu removal may also occur at higher flow rates because copper sulfides are very insoluble. However, the Cu removal process is not consistent.

In Table 2, pH and sulfate removal values vary with flow. In these cases, there appears a general relationship of better removal of hydrogen ions and sulfate as the flow is decreased. Figure 1 shows the relationship between flow rate and the decrease in sulfate concentration. A linear regression analysis on flow rates below 2.3 L/min and the decrease in sulfate concentrations gives an $R^2$ value of 82%.

In Figure 2, a plot of sulfate removal with Eh shows that as Eh decreases, generally greater amounts of sulfate are removed, presumably due to sulfate reduction. A linear regression on these data gives an $R^2$ value of 77%. The relationships in Figures 1 and 2 also implies that Eh generally decreases with flow rate. Linear regressions on other sets of data result in much lower correlation coefficients.

These data indicate that processes in the substrate affecting pH, Eh, and sulfate concentration operate at much greater efficiencies below a certain flow rate. These three parameters are related to microbiological processes in the wetland. The product of these processes is dissolved sulfide. Dissolved sulfide has been measured at around 5 mg/L in Cell A. Sulfide may precipitate with and remove metals from solution. Microbial sulfate reduction processes are sensitive to the presence of oxygen, tying their efficiencies to lower Eh conditions. Higher flow rates may introduce more oxygen into the system than can be removed by microbial processes leaving Eh values high.

Relationships Between Mass Removal Data and Seasonal Cycles

To take advantage of our flow rates for optimizing treatment efficiency in other wetlands, conversion to metal loading rates should be made. The resulting values apply to a wetland with mushroom compost as described above and a size of 18.6 $m^2$. It is very possible that other substrates will have different loading rates. In addition, It is important to consider how the water is to penetrate, flow through, and be collected from the complete cross-sectional area of the substrate.

Mass removal data was calculated based on discussions with Hedin (pers. comm.) as follows:

$$([M_{md}] - [M_{op}])(FR)(CF)/(A)$$

where:
[M] = concentration of metal or sulfate
md = mine drainage
op = wetland output
FR = flow rate into wetland
CF = conversion factor = (1440 min/day)
A = area of wetland = (18.6 $m^2$)

In Cell A, it has been found that transpiration and evaporation account for at most a 10 % reduction in flow between the inlet and outlet (Dietz, pers. comm. 1989). Consequently, it is assumed that the flow out equals the flow in and the resulting units are: mg/day/$m^2$.

The mass of metal removed from solution per day per square meter of wetland has been calculated for samples analyzed and is also given in Table 2. Mass removal data indicate that there are some tendencies for greater metal removal rates and pH increases during the summer season than for the winter. However, as expected, Mn shows no trend.

These tendencies probably reflect the generally lower temperatures found in the wetland during the winter (Table 2). Lower temperatures should result in lower microbial activity in the substrate. Sulfate reducing bacteria, however, will still function at 3-4°C (Batal et al. 1989). With lower microbial activity, less metal removal is expected due to a slow down of the processes that the microbes mediate.

If mass removal data are compared with flow rates, it is not the case that the greatest removals of a pollutant occur at low flow. This is particularly evident for the Cu data. The reason for this is that the above equation multiplies the decrease by the flow rate, and at flow rates below 1.5 L/min, all of the Cu is essentially removed from solution. For the other metals, this effect of low mass removal at low flow, is not as prominent. This observation implies that a passive wetland system has to be operated with an obvious margin of safety to insure satisfactory removal results.

Other factors may be influencing the mass removal data trend with seasonal cycles besides seasonal changes and loading rate fluctuations. Mass removal

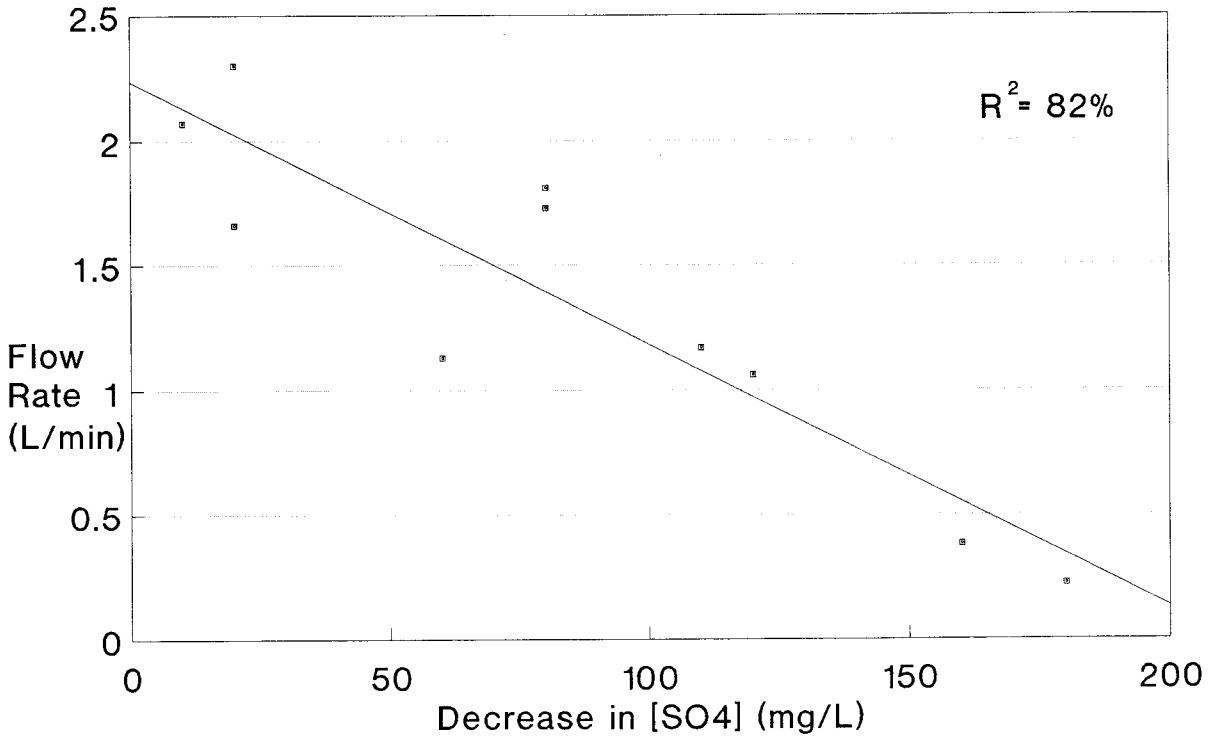



Figure 1. Decease in sulfate concentration versus flow rate for Cell A.

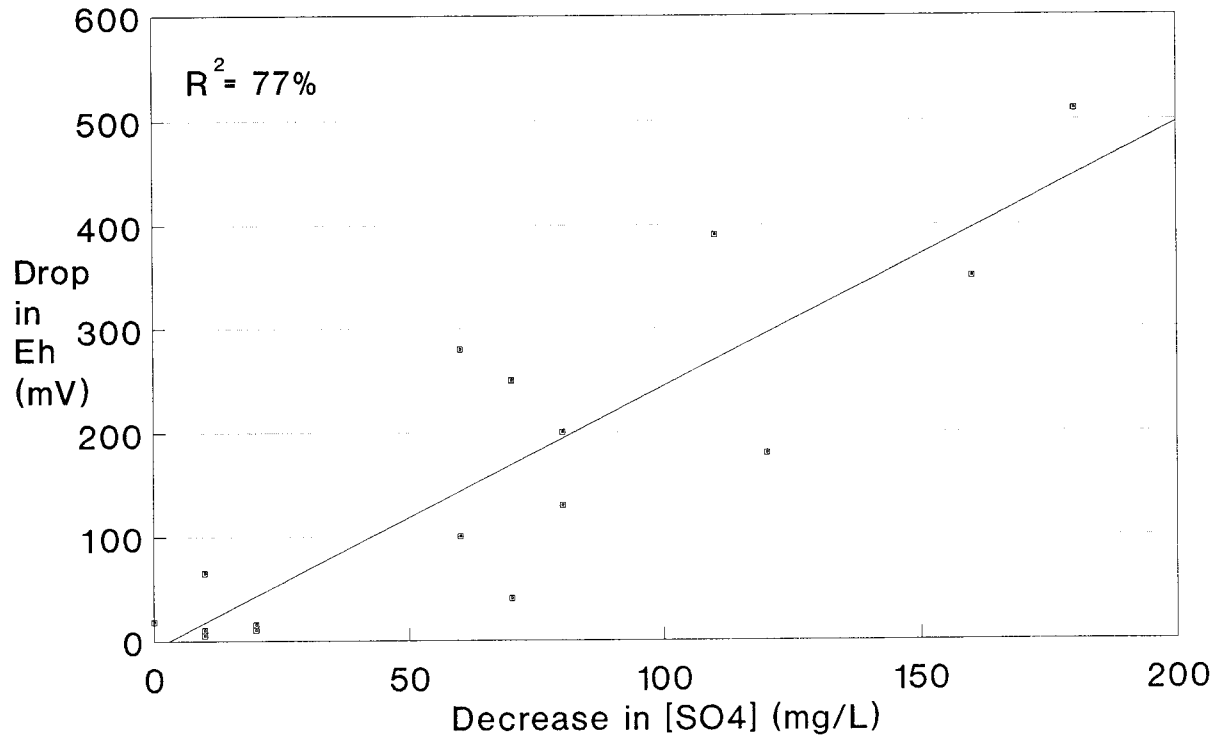



Figure 2. Decrease in sulfate concentration versus decrease in Eh for Cell A.

data for Cu, Zn and sulfate show similar fluctuating trends which are most evident in the spring. Some of the variation may be explained by a drop in metal removal efficiencies at higher loading rates as discussed above.

Another explanation for this variation may be large fluctuations in temperature which occur particularly in the spring. This is when some of the largest fluctuations in the mass removal data occur. Small changes in temperature of 3-4$^{o}$C can cause the interdependent community of microbes operating in a reducing system to go dormant for several days until the microbes are able to readjust to the different temperature (Updegraf, pers. comm.). Such a variable could greatly affect seasonal trends, and it may produce the greatest instability in the fall and spring when temperature fluctuations are the largest. Determining if these factors can cause temporary failures certainly is a future research objective.

## Summary

Research conducted at the Big Five wetland shows that the Eh of the effluent and the decrease in sulfate concentration is very responsive to changes in flow rate. Data indicate that processes in the substrate affecting pH, Eh, and sulfate concentration operate at much greater efficiencies below flow rates of about 1.5 L/min. These three parameters are related to microbial sulfate reduction in the wetland where the dissolved sulfide produced may precipitate with and remove metals from solution.

An important result of these tests is the delineation of metal removal efficiencies and pH increases with flow rate. Consistently good results for Cu, Fe and Zn removal from the Big Five mine drainage are obtained using a substrate of fresh mushroom compost with flow rates less than 1.5 L/min in a system with an area of 18.6 m$^2$. This corresponds roughly to a loading rate of 100, 1000, and 4000 mg/day/m$^2$ for Cu, Fe and Zn, respectively. For Cu, decrease in concentration is not as strongly linked to flow rate as is found for sulfate and pH changes. This may be due to some other process, such as complexation by humic acids, that is also important to copper removal. For Mn, no removal with flow is detected and it appears that at low flows Mn is added to solution. This implies that if Mn removal is an objective it will have to be accomplished in later stages in the wetland after the pH is raised and the other heavy metals have been removed.

Mass removal data does not appear to be as good an indicator of how well a wetland is removing pollutants. This appears to be due to the need to maintain the loading rate at a value that is low enough to allow a margin of safety for the natural misfortunes that occur.

## Literature Cited


Batal, W., Wildeman, T. R., Mohdnoordin, N. and L. S. Laudon. 1989. Bacteriological tests from the constructed wetland of the Big Five Tunnel, Idaho Springs, Colorado, p. 550-557. In Constructed Wetlands for Wastewater Treatment, D. A. Hammer, ed. Lewis Publishers, Ann Arbor, MI.

Hammer, D.A., ed. 1989. Constructed Wetlands for Wastewater Treatment, Lewis Publishers, Ann Arbor, MI., 831 p.

Hiel, M. T. and F. J. Kerins, Jr. 1988. The Tracy wetlands: a case study of two passive mine drainage treatment systems in Montana, p. 352-358. In Proceedings of a conference on Mine Drainage and Surface Mine Reclamation Vol. I: Mine Water and Mine Waste. U.S. Department of the Interior, Bureau of Mines Information Circular IC 9183.

Howard, E. A., Emerick, J. E. and T. R. Wildeman. 1989a. The design, construction and initial operation of a research site for passive mine drainage treatment in Idaho Springs, Colorado, p. 761-764. In Constructed Wetlands for Wastewater Treatment, D. A. Hammer, ed. Lewis Publishers, Ann Arbor, MI.

Howard, E. A., T. R. Wildeman, L.S. Laudon, S.D. Machemer. 1989b. Design considerations for the passive treatment of acid mine drainage, p. 651-660. In Proceedings of the Conference "Reclamation, A Global Perspective". Alberta Land Conservation and Reclamation Council Report # RR-TAC 89-2.

Kleinmann, R. L. P. 1983. A low-cost, low maintenance treatment system for acid mine drainage using sphagnum moss and limestone. Symposium on Surface Mining, Hydrology, Sedimentology and Reclamation. (Lexington, KY: University of Kentucky).

Wildeman, T. R. and L. S. Laudon. 1989. The use of wetlands for treatment of environmental problems in mining: non-coal mining applications, p. 221-231. In Constructed Wetlands for Wastewater Treatment, D.A. Hammer, ed. Lewis Publishers, Ann Arbor, MI.

# TESTS OF ORGANIC MATTER ADDITIONS FOR THE NEUTRALIZATION OF ACID MINE DRAINAGE INFLUENCED LAKES[1]

by

Richard B. Brugam, James Gastineau, and Eric Ratcliff[2]

Abstract. Many investigators have proposed that dissimulatory sulfate reduction by anaerobic bacteria should result in declines in acidity in sulfuric acid-polluted lakes. Because these bacteria use organic matter as a substrate, increased organic matter loading of an acid lake should encourage bacterial sulfate reduction resulting in the loss of sulfuric acid from the water column.

To test this hypothesis, we constructed six cylindrical polyethylene enclosures in an acid (pH=3.1) lake on an abandoned coal surface mine. Tops of the enclosures were open to the air and bottoms were sunk in the mud. Organic matter in the form of anaerobic digester sludge or straw was added to two of the enclosures. Two other enclosures received additions of digester sludge or straw with $Ca(OH)_2$. Two enclosures were reserved as controls. One received only $Ca(OH)_2$ to control for hydrated lime additions with the organic matter. The last enclosure received no additions and served as a control for enclosure. Experiments lasted six weeks.

Alkalinity and pH increased in enclosures treated with straw. Stabilized digester sludge did not reduce acidity because it contained little readily oxidizable organic matter. In the straw treatments oxygen levels declined and sulfide was produced. More sulfide was produced where $Ca(OH)_2$ and straw were added together suggesting that dissimulatory sulfate reduction is enhanced at higher pH.

Additional key words: Acid mine lakes, sulfate reduction, acid lake neutralization.

## Introduction

Acid mine drainage contaminates lakes in many parts of the country but it causes especially severe problems in the midwest where both acid mine drainage and lakes are a product of surface mining for coal (Brugam and Lusk 1986). Neutralization of acid contaminated final cut impoundments is a major problem for midwestern mining companies. Under normal circumstances inorganic bases are added to the impoundments by mining companies to reduce acidity, but a number of investigators (King et al. 1974; Campbell and Lind 1968; Turbak et al. 1979) have suggested that additions of organic matter might also enhance neutralization.

King et al. (1974) argued that the added organic matter would encourage the reduction of sulfate to sulfide within a lake removing sulfuric acid from the water column. The organic matter is a substrate for sulfate reducing bacteria of the genus Desulfovibrio (Pfennig and

[1]Paper presented at the 1990 Mining and Reclamation Conference April 23-26 held in Charleston, West Virginia, USA.

[2]R.B. Brugam is a Professor of Biological Sciences, Southern Illinois University, Edwardsville, IL 62026. J. Gastineau is an Environmental Assessment Specialist, Illinois Geological Survey, Carbondale, IL and E. Ratcliff is a Graduate Student in the Environmental Studies Program, Southern Illinois University, Edwardsville.

Widdel 1982). These bacteria survive only under anaerobic conditions where they use the oxygen from sulfate to oxidize organic matter (a process called dissimulatory sulfate reduction). Anderson and Schiff (1985) describe the oxidation of organic matter to $CO_2$ by sulfate-reducing bacteria (equation 1).

$$2CH_2O + 2H^+ \rightarrow 2CO_2 + H_2S + H_2O \quad (1)$$

The result of this reaction is the removal of hydrogen ion from the water. King et al. (1974) suggested that the $H_2S$ produced during dissimulatory sulfate reduction would be outgassed thereby removing hydrogen ion from the lake.

Workers with acid precipitation-influenced lakes (Schindler et al. 1980) have also suggested that sulfate reduction plays a role in the neutralization of these lakes. In contrast to the investigators of acid mine drainage, Schindler et al. (1980) have suggested that the major loss of sulfide from the lake is by precipitation as FeS.

Anderson and Schiff (1985) propose a mechanism for lake neutralization by FeS precipitation. They argue that sulfate reduction alone is inadequate to cause permanent neutralization without either precipitation of FeS or outgasing of $H_2S$. Although sulfate reduction produces buffer capacity in the form of $H_2S$ (equation 1), the reduced sulfur can be quickly reoxidized to sulfuric acid under aerobic conditions (equation 2) if the $H_2S$ gas has not escaped into the atmosphere.

$$H_2S + 2O_2 \rightarrow SO_4^{--} + 2H^+ \quad (2)$$

Although precipitation of sulfide with $Fe^{++}$ will prevent the reoxidation of the sulfide, it also releases $H^+$ (equation 3).

$$Fe^{++} + H_2S \rightarrow FeS + 2H^+ \quad (3)$$

For sulfide reduction and FeS precipitation to be effective agents of neutralization, the iron in equation 3 must originate from the reduction of $Fe^{+++}$ (equation 4).

$$CH_2O + 4FeOOH + 8H^+ \rightarrow CO_2 + 4Fe^{++} + 7H_2O \quad (4)$$

The source of the FeOOH is likely to be the lake sediment. The neutralization of a lake by precipitation of FeS must, therefore, include both the reduction of $Fe^{+++}$ to $Fe^{++}$ and the reduction of $SO_4^{--}$ to $S^{--}$.

King et al. (1974) originally hypothesized that organic matter addition would aid in the neutralization of acid mine drainage contaminated lakes by providing a substrate for sulfate-reducing bacteria. We have tested this hypothesis by adding organic matter in the form of straw and sewage sludge to polyethylene enclosures in an acidic surface mine lake.

## Materials and Methods

### Location

Our experimental location was pit #8, a 58 ha. lake on Peabody Coal Company's Will Scarlet Mine which is located south of Crab Orchard in Saline County, Illinois (description in Jones et al. 1985). Our experimental apparatus was placed in the "middle incline", a former inclined haul road now under water. Because the acidity of the middle incline is higher than that of the open lake, it was chosen for our work. The maximum depth of the middle incline is 6m.

### General Experimental Design

Experiments were performed in cylindrical polyethylene enclosures 6m long and 1 m diameter (Fig. 1). The polyethylene was held open with large plastic hoops. Six cylinders were suspended from a floating dock in the middle incline. The tops of the cylinders were open to the air and the bottoms were open and sunk in the mud of the lake bottom.

The advantage in using enclosures for limnological experiments is that small regions of a lake can be partitioned off from the main lake for experimentation. These "enclosures" allow the establishment of controls to the experimental manipulations. They also reduce the expense of experiments by reducing the amount of reagents needed. Finally, in the case of experiments where the possibility of temporary environmental degradation exists, a smaller portion of the environment is placed at risk. A disadvantage of the enclosure technique is that, although they reduce exchange of water with the open lake, they don't completely eliminate it. There is some leakage of open lake water into the enclosures.

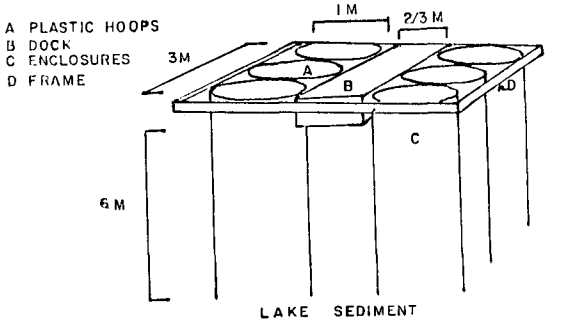



**Figure 1: Experimental Apparatus**

In this paper we discuss an enclosure experiment conducted at Pit #8 between August 24 and September 27, 1988. Additions to the enclosures are presented in Table 1. The control enclosure received no additions and was used as a test of the enclosure process. The lime enclosure allowed comparisons between organic matter additions and the inorganic bases normally employed by mining companies to neutralize lakes. Organic matter used as additives were dried straw (straw enclosure) or stabilized anaerobic digester sludge (sewage enclosure). Combinations of lime and organic matter (lime+straw and lime+sewage enclosures) were used to test the hypothesis of Schuurkes and Kok (1988) that dissimulatory sulfate reduction is limited by pH.

Table 1: Additions to enclosures

| Treatment | Date of Addition 8/24/88 | 8/31/88 | 9/7/88 |
|---|---|---|---|
| Control (C) | Nothing | Nothing | Nothing |
| Lime | 3.6 kg Limestone | 1.8 kg Limestone | .9kg Hydrated Lime |
| Straw | 10 kg Straw | 5 kg Straw | 5 kg Straw |
| Sewage (Sew) | 20 kg Sludge | 10 kg Sludge | 10 kg Sludge |
| Lime+Straw (LStraw) | 3.6 kg Limestone + 10 kg Straw | 1.8 kg Limestone + 5 kg Straw | .9kg Hydrated Lime + 5 kg Straw |
| Lime+Sewage (LSew) | 3.6 kg Limestone + 10 kg Sludge | 1.8 kg Limestone + 5 kg Sludge | .9kg Hydrated Lime + 5 kg Sludge |

No special calculations of organic loading were made in advance of adding sewage sludge or straw to the enclosures. Organic matter additions were simply continued at a high level until sulfide was detected in the organically loaded enclosures.

## Field Sampling

Water samples were taken from 0, 3, and 6m in each enclosure and from the open lake using a Kemmerer sampler (American Public Health Association 1985). We will discuss only the samples taken from the 3m layer in the enclosures because that depth was representative of the whole water column during the experiment. Although at other times of the year the middle incline is strongly chemically stratified below 4m, our measurements show that this stratification had broken down by the time we started our experiment.

Separate water samples were taken for general chemical analysis, for oxygen and for sulfide. Temperature and pH were measured in the field with an alcohol thermometer and a Corning portable glass electrode pH meter.

## Laboratory Methods

In the laboratory, samples were analysed using the techniques of Standard Methods (American Public Health Association 1985). Analysis began immediately after our return from the field. Alkalinity was measured using a potentiometric titration. Acidity was measured using the hot peroxide method. Sulfate was measured turbidimetrically. Oxygen was determined using the Alsterberg modification of the Winkler technique. Sulfide was measured by iodimetric titration. Iron was measured in 1% nitric acid-preserved aliquots of the original sample by atomic absorption spectrophotometry.

## Statistical Analysis

We compared treatments between enclosures using the Wilcoxon Signed Rank Test (Sokal and Rohlf 1981). For the analysis we paired measurements at each depth between treatments. Results of statistical analysis are presented as comparisons between treatments (for example, straw versus control). The Z statistic is the Wilcoxon Signed Rank Test statistic. The number of sample pairs used for comparisons was 18 (3 depths X 6 weeks).

The advantage of the Wilcoxon Signed Rank Test is its lack of assumptions about statistical distributions of the samples. An alternative test would be the paired sample ANOVA, but this assumes that the data are normally distributed and have equal variances -- assumptions that our

data do not necessarily fulfill.

## Results and Discussion

The sewage enclosures showed no changes that could be attributed to the organic matter additions. Apparently, the stablized sewage sludge that we employed was dominated by recalcitrant compounds with little of the simple organic compound that form substrates for sulfate reduction.

Sulfide appeared in the lime+straw and straw enclosures by the second week (Fig. 2 ). The amounts of sulfide in the lime+straw enclosure were always higher than those in the straw enclosure (lime+straw versus straw, Z=-2.61, P=.0092). As sulfide increased in the lime+straw and straw enclosures dissolved oxygen decreased until it disappeared. Likewise, pH (Fig. 3 ) and alkalinity (Fig. 4 ) increased in the enclosures containing straw. As expected pH and alkalinity also increased in the lime enclosures.

Acidity declined in all enclosures throughout the experiment (Fig. 5 ). It is unclear why acidities were so high at the beginning of the experiment. The usual acidity of surface water of the lake was approximately 50 mg/l $CaCO_3$ -- the level to which the control enclosure declined by week 3. The acidity of the straw treatment declined even more than the controls (straw versus control, Z=-3.10, P=.0019). Acidity declined most in enclosures which received lime additions (lime+straw versus control, Z=-3.52, P=.0004; lime versus control, Z=-2.95, P=.0032). Acidity in the straw enclosure only briefly declined to zero even though alkalinity and pH rose. Otherwise, significant acidity remained in the straw enclosure. The acidity in the straw enclosure was less than in the control enclosure, however (straw vs. control Z=-3.10, P=.002).

The straw treatment increased Fe concentrations (straw vs. control Z=-2.44, P=.02, Fig 6 ). When inorganic neutralizing agents were added with the straw, iron remained unchanged (lime+straw vs. control Z=-.46, P=.65).

Sulfate concentrations remained unchanged in all enclosures during the course of the experiment. The conversion of sulfate to sulfide in the straw enclosures seems not to have had any effect on sulfate concentrations (Fig. 7). The sulfate equivalent to the sulfide we found is small (25 mg/l $S^{--}$= 75mg/l $SO_4^{--}$) in relation to the usual concentration of sulfate present in the lake (2,000 mg/l). This means that approximately 4% of the sulfate was converted to sulfide -- a change unlikely to be detected using the turbidimetric method of sulfate analysis.

We have found that organic additions do neutralize acidic lakes by supporting the production of sulfide. Our experiment shows a rise in pH to approximately the $pK_a$ of hydrogen sulfide (6.95).

Because combined additions of $CaCO_3$, $Ca(OH)_3$ and straw produced larger amounts of sulfide than straw alone, dissimulatory sulfate reduction seems to be limited by pH as Schuurkes and Kok (1988) suggest. The inorganic neutralizing agents raise pH to optimal levels for sulfate reduction supported by the added organic material. Straw alone also allows sulfate reduction, but rates of reduction are inhibited by the lower starting pH. The accumulating $H_2S$ eventually raises pH sufficiently to allow accelerated sulfide production.

A major problem with our organic matter additions is that neutralization was not permanent. High pH and alkalinity existed only as long as sulfide existed. As soon as the sulfide was reoxidized, pH dropped and alkalinity disappeared. Our measurements of iron show that there was no significant precipitation of FeS from the water column. Water drawn from the enclosures was grey in color turning black with FeS only on standing in sampling bottles. This phenomenon has also been seen by Stahl (1978). It suggests that FeS precipitation is prevented in situ.

It is unclear what prevents the precipitation of FeS in the enclosures. The concentrations of iron and sulfide in the straw enclosure on week 3 are high (total Fe= 76 mmolar, total sulfide = 313 mmolar). The pH at the same time is 5.5. It would seem that these conditions would be conducive to strong FeS precipitation.

One possible mechanism preventing metal sulfide precipitation could be chelation by organic compounds. In naturally acidic lakes organic material from decaying plants are strong iron chelators (Hutchinson 1957). The decay of straw probably also produces chelators which make Fe unavailable for FeS precipitation.

The practical result of metal chelation is the prevention of the loss of mineral acidity that we might expect from metal sulfide precipitation. This result limits the usefulness of direct application of organic matter to acid lakes. Organic addition might be more successful if it were done in the hypolimnion (deep water) of a lake. These waters are sealed off from the atmosphere and are anaerobic for a longer time period than is possible in

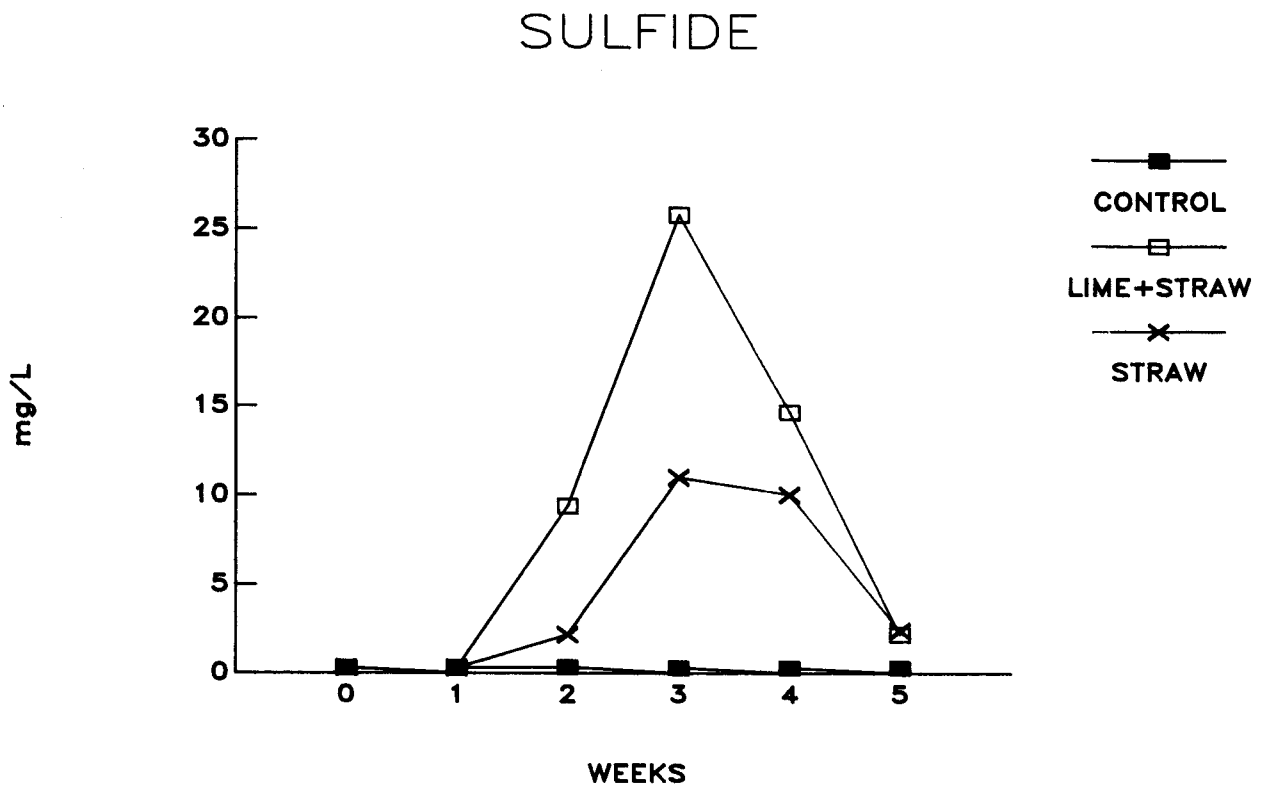



Figure 2 : Sulfide concentrations at the 3 m level in the experimental enclosures

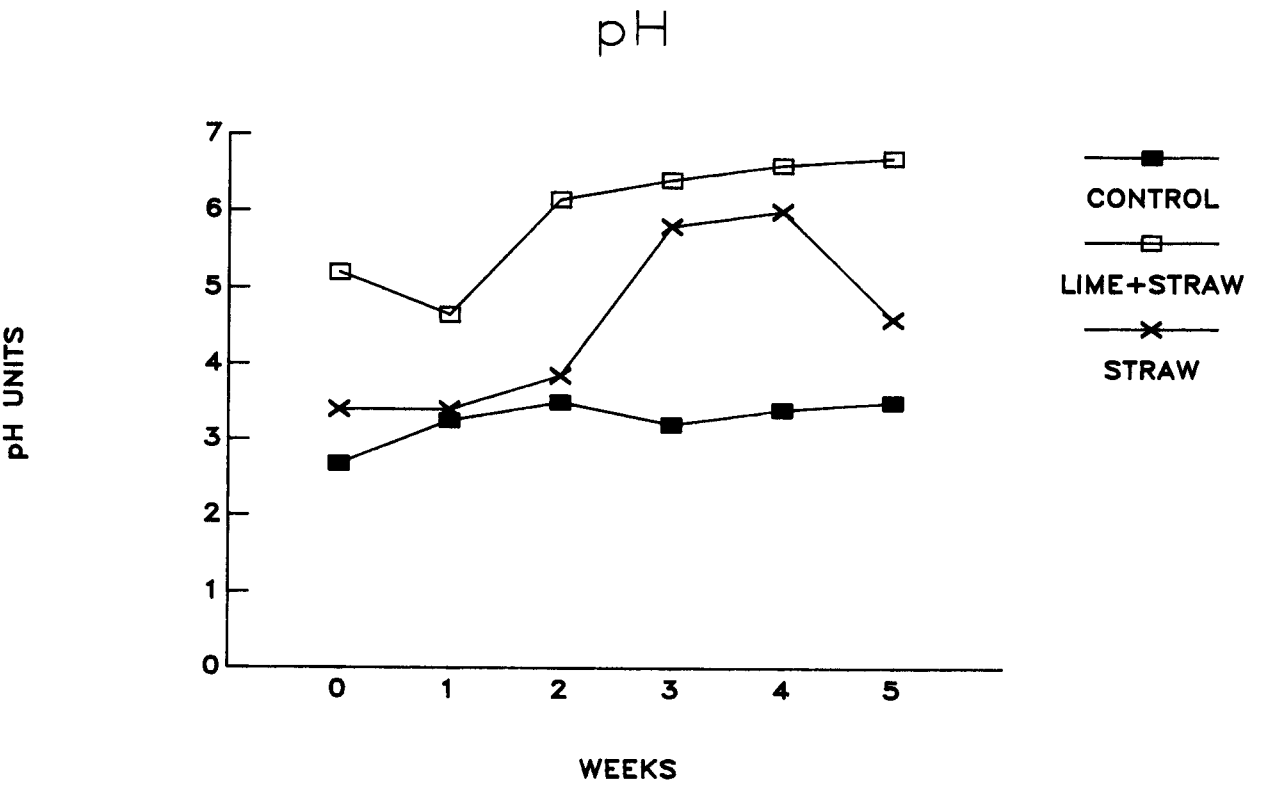



Figure 3: pH at the 3 m level in the experimental enclosures.

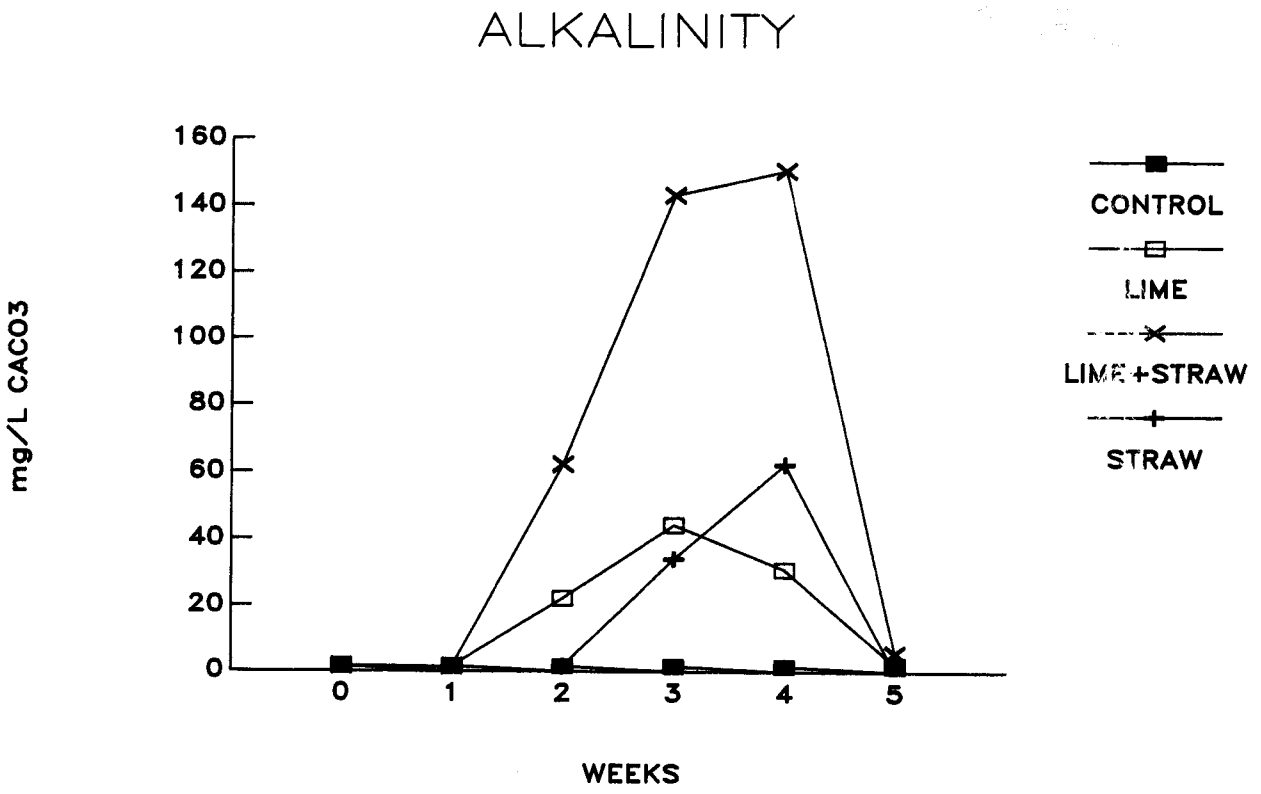



Figure 4: Alkalinity at the 3 m level in the experimental enclosures

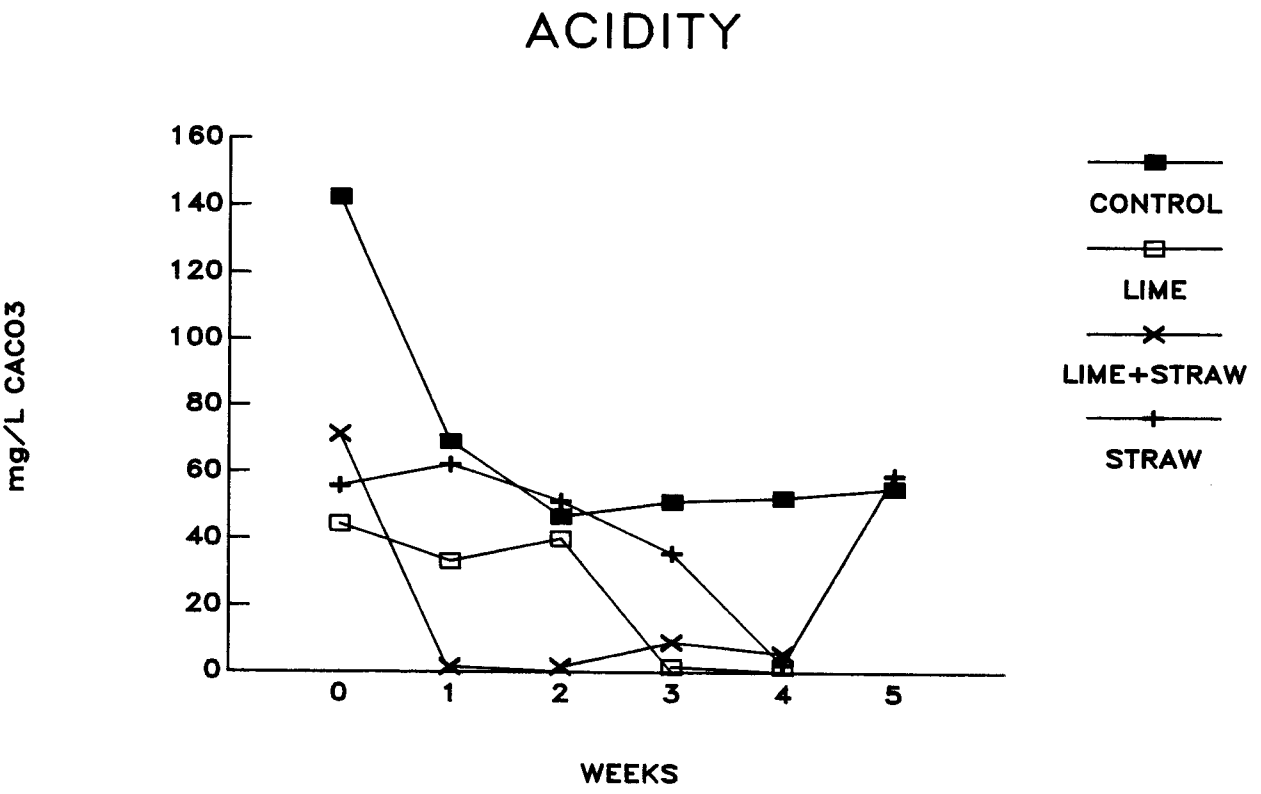



Figure 5: Acidity at the 3 m level in the experimental enclosures.

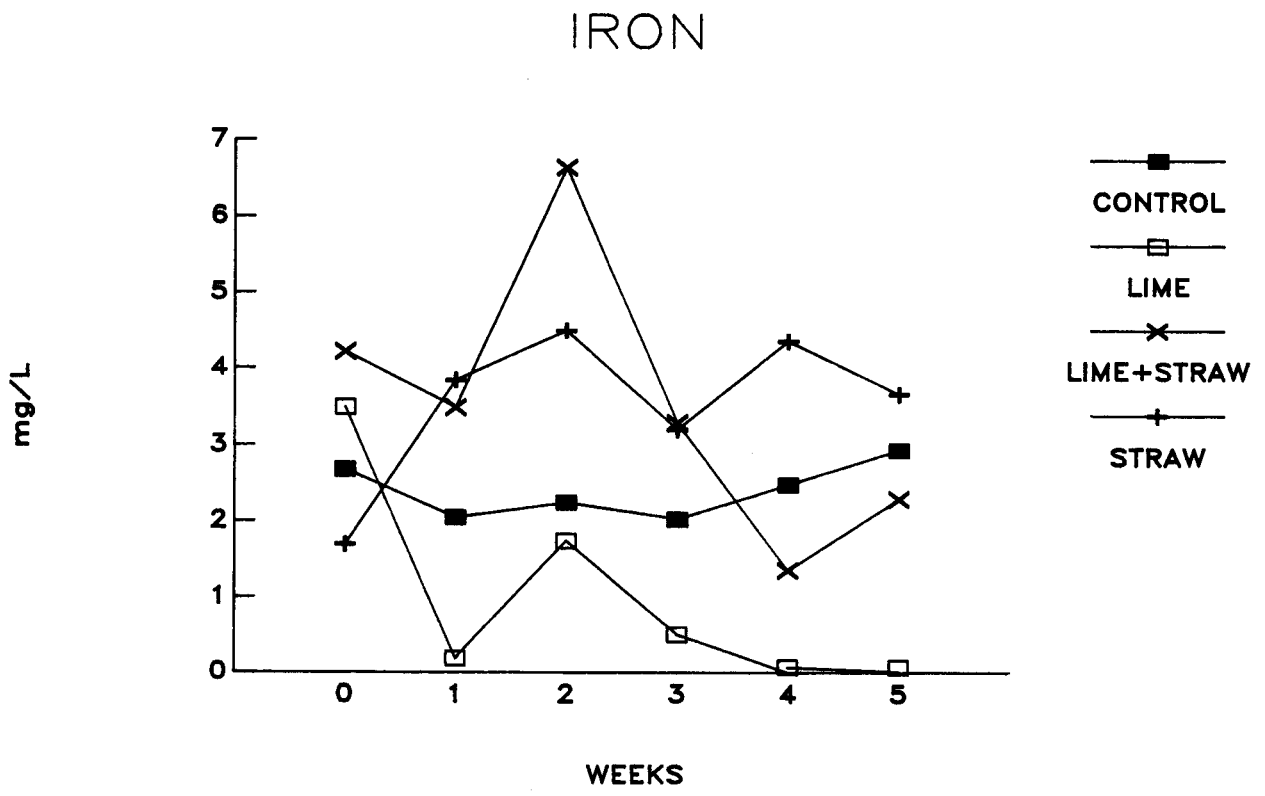



Figure 6: Iron concentrations at the 3 m level in the experimental enclosures.

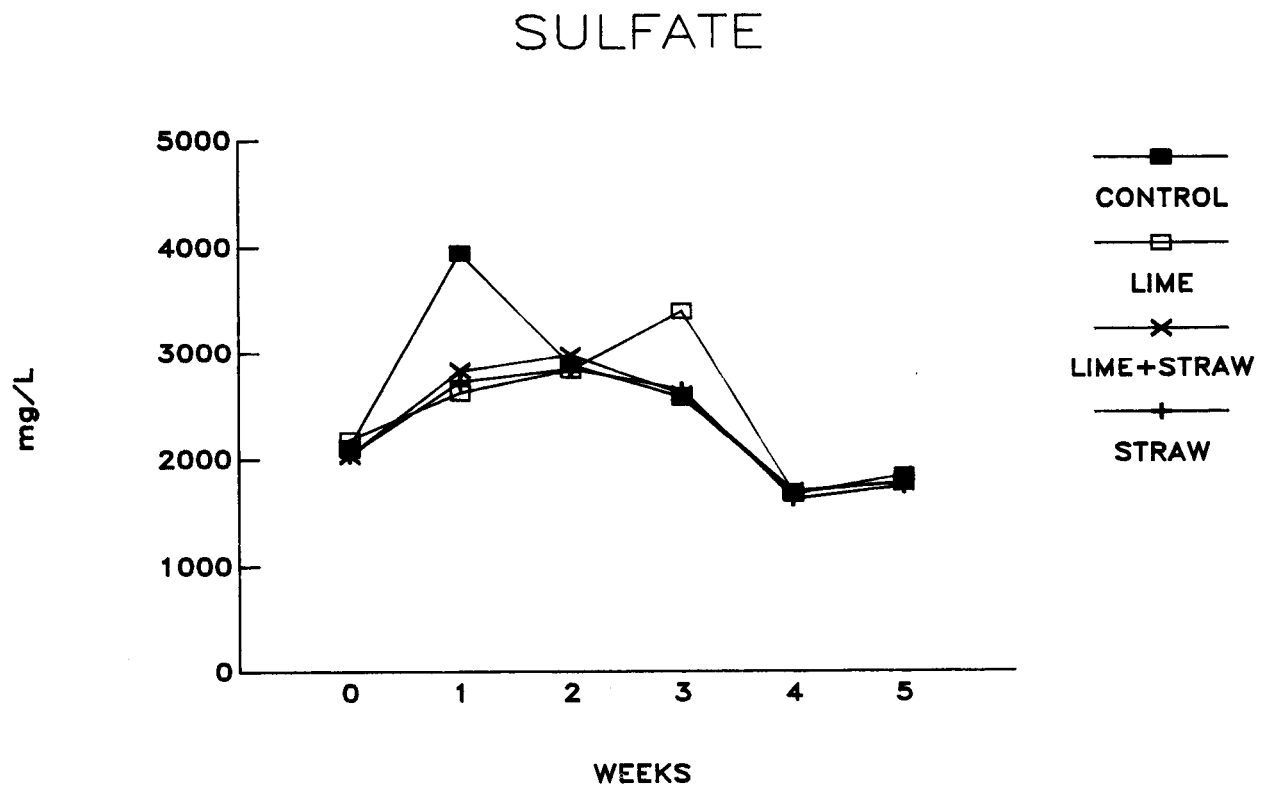



Figure 7: Sulfate concentrations at the 3 m level in the experimental enclosures.

our enclosures. The longer time available for decay might allow more complete decomposition of potential chelating substances generated from the straw.

If the organic matter is added to the hypolimnion of a lake one might expect that $H_2S$ would be brought into the epilimnion and oxidized at turnover. For the process to be successful, sufficient FeS must be entombed in the sediment to prevent the reoxidation of sulfide. Herlihy et al. (1987) found that the deep water sediment of an acid mine drainage-influenced lake was a location of strong sulfate reduction which permanently removed acidity and sulfate from the overlying water. It is unclear why there should be a permanent loss of acidity in the lake Herlihy et al (1987) studied but not in Pit 8.

Our results suggest that additions of mineral bases along with the organic matter might also increase metal sulfide precipitation by increasing pH enough to make such precipitation more probable. Davison (1986) used a combination of organic matter with hydrated lime to neutralize an acidic sand quarry in England. He found that neutralization occurred but provided no test of whether the combination of materials was more effective than the hydrated lime alone. Our results suggest that this might be true.

## Conclusions

Additions of organic matter to acid lakes to encourage dissimulatory sulfate reduction does neutralize these lakes. At present, however, it is unclear whether neutralization is permanent.

## Acknowledgements

This work was supported by the U.S. Bureau of Mines contract No. J0178025. We thank Patricia Erickson and Pamela MacIntire from the Bureau of Mines for thoughtful discussions about this project. I am especially grateful to the Illinois Division of Peabody Coal Company for allowing me access to the Pit #8 site and to Robert Hill and Phil Stafford of Peabody Coal for help and guidance at the mine.

## Literature Cited

American Public Health Association. 1985. Standard Methods for the Examination of Water and Wastewater. 874 p.

Anderson, R.L. and S.L. Schiff. 1987. Alkalinity generation and the fate of sulfur in lakes sediments. Can. J. Fish. Aquat. Sci. 44(Suppl. 1): 188-193

Baas-Becking, L.G.M., I.R. Kaplan, and D. Moore. 1960. Limits of the natural environment in terms of pH and oxidation-reduction potentials. Journal of Geology 68:243-283.

Brugam, R.B. and M. Lusk. 1986. Chapter 10: Diatom evidence for neutralization in acid surface mine lakes. In Diatoms and Lake Acidity J. Smol, R.W. Battarbee, R.B. Davis, J. Merilainen eds. Dr. W. Junk, Dordrecht p. 115-129.

Campbell, R.S., and O.T. Lind 1969. Water quality and aging of strip-mine lakes. J. Wat. Pollut. Cont. Fed. 41:1943-1955.

Davison, W. 1986. Sewage sludge as an acidity filter for groundwater-fed lakes. Nature 322:820-822.

Jones, D.W., M.J. McElligott, and R.H. Mannz. 1985. Biological, Chemical and Morphological Characterization of 33 Surface Mine Lakes in Illinois and Missouri. Peabody Coal Co. 248 p.

Herlihy, A.T., A.L. Mills, G.M. Hornberger, and A.E. Bruckner. 1987. The importance of sediment sulfate reduction to the sulfate budget of an impoundments receiving acid mine drainage. Water Resources Research 23:287-292.

Hutchinson, G.E. 1957. Treatise on Limnology. Vol I. Geography, Physics and Chemistry. Wiley Interscience, New York. 1000 p.

King, D.L., J.T. Simmler, C.S. Decker, and C.W. Ogg. 1974. Acid strip mine lake recovery. Journal of the Water Pollution Control Federation 42:2301-2316.

Pfennig, N. and F. Widdel. 1982. The bacteria of the sulphur cycle. Phil. Trans. R. Soc. B 433-441.

Schindler, D.W. and R. Wagemann, R.B. Cook, T. Ruszcynski, and J. Prokopowich. 1980. Experimental acidification of Lake 223, Experimental Lakes Area: Background data and the first three years of acidification. Can. J. Fish. and Aquat. Sci. 37:342-354.

Schuurkes, J.A.A.R., and C.J. Kok. 1988. In vitro studies on sulphate reduction and acidification in sediments of shallow soft water lakes.

Freshwater Biology 19:417-426.

Sokal, R.B. and F.J. Rohlf. 1981. Biometry: Principles and Practice of Statistics in Biological Research. W.H. Freeman & Co. San Francisco.875p.

Stahl, J.B. 1978. Water and peculiar stratification in an organically loaded strip-mine lakes. Water Research 13: 467-471.

Turbak, S.C., G.J. Olson, and G.A. McFeters. 1979. Impact of western coal mining. I. Chemical investigations of a surface coal mine sedimentation pond. Water Research 13: 1013-1031.

This page intentionally left blank.

# Revegetation - General

This page intentionally left blank.

# RECLAMATION OF SOIL STERILANT CONTAMINATED SOILS[1]

S. Landsburg and A.W. Fedkenheuer[2]

Abstract. The Alberta Gas Transmission Division of NOVA Corporation of Alberta has more than 14,000 km of pipeline rights-of-way to operate and maintain. Above ground facilities associated with the pipeline system include approximately 850 meter stations, 36 compressor stations, numerous valve sites and several office and shop facilities. Total vegetation control at above ground facilities is an attempt to eliminate all vegetation under and within the fenced areas. In the past, total vegetation control was accomplished through the use of "soil sterilants" such as Hyvar-XL and Calmix. With the discovery of migration of soil sterilants outside fenced areas, vegetation control has recently been attempted using non-residual products such as Roundup. Mechanical methods involve using mowers, weed-eaters or hand-picking. Reclamation problems associated with vegetation management in AGTD are directly related to the use of soil sterilants. Twenty-five above ground facilities have been monitored annually as a result of bromacil contamination. Two research projects were initiated a number of years ago to determine the best methods of bromacil dissipation in the field as well as the best methods of rehabilitating bromacil contaminated topsoils in the laboratory and greenhouse. The two methods of bromacil dissipation occurring in the field were leaching and microbial degradation. Recommendations for the rehabilitation and reclamation of soils contaminated by bromacil include the use of activated carbon, manure, wet-dry soil cycles, and plant seedlings to enhance adsorption, absorption, and degradation of bromacil.

Additional Key Words: bromacil, weed control, soil degradation, soil pollution.

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]Senior Soil Scientist and Senior Specialist, respectively, NOVA Corporation of Alberta, P.O. Box 2535, Station M, Calgary, Alberta, Canada, T2P 2N6.

Publication in this proceedings does not preclude authors from publishing their manuscripts, whole or in part, in other publication outlets.

## Introduction

In the Province of Alberta, the Electrical Protection Act has regulations governing electrical installation and equipment in the oil and gas industry. As well, the Weed Control Act requires industry to conduct a weed control program to prevent the introduction and spread of noxious and nuisance weeds. These Acts, along with an operating policy of safety and fire and explosion prevention, have resulted in the removal of all potentially combustible material within the fencelines of Alberta Gas Transmission Division (AGTD) facilities of NOVA Corporation of Alberta (NOVA).

Total vegetation control is the suppression or control of all vegetation in order to maintain a vegetation-free or barren area. Within the AGTD, total vegetation control has been used within the fencelines of 36 compressor stations, 850 meter stations, off-line sales stations, yards of maintenance and storage facilities, most valve sites and some rectifiers.

Prior to 1987, the AGTD used non-selective residual herbicides commonly referred to as soil sterilants for total vegetation control at above ground facilities. The two soil sterilants most commonly used were Calmix and Hyvar XL, both having bromacil as the major active ingredient. Soil sterilants were chosen for the total vegetation control program due to the long-term control provided (minimum of two to five years). This resulted in reduced labour requirements for weed control, as one major application was required every two to five years with touch-up applications applied as required. Both products were effective against most plants when correctly used. Due to problems with off-site migration, the AGTD suspended use of soil sterilants in 1987. Since then, total vegetation control has been conducted primarily using the non-selective, non residual herbicide known as Roundup.

## Bromacil

Bromacil ($C_9H_{13}BrN_2O_2$) is a residual herbicide which is selective, or non-selective, depending on the application rate. It is used at rates of up to 27 kg active ingredient/ha on non-cropland areas for non-selective control of a wide range of grasses and broadleaf weeds, and certain woody species. It is used for selective control at rates of 2 to 7 kg active ingredient/ha for weeds in orange, grapefruit and lemon orchards, and for seedling weeds in pineapple (Beste et al. 1983). Bromacil can be applied by spraying or spreading it dry on the soil surface, preferably just before or during a period of active growth of weeds. Bromacil has a high solubility and stability in water, whereas most other soil sterilants are less soluble and stable (Landsburg and Reinl-Dwyer 1989).

Bromacil may persist at phytotoxic levels for up to six months when applied at "normal" rates of 2 to 5 kg/ha (Gardiner 1975). According to Hassall (1982) bromacil applied at a rate of 1.7 kg/ha can be expected to persist at toxic levels for six months to a year. Phytotoxic levels in Israeli soils have been reported by Angemar et al. (1984) as being less than or equal to 0.1 mg bromacil/kg soil for mineral soils and 0.4 mg/kg for peat soils. Phytotoxicity values in ED50 (the dose that reduces test plants fresh weight by 50%) were found to be 0.4, 0.08, 0.06, and less than 0.01 mg/kg soil for an organic, clay loam, loess and a sandy loam soil, respectively (Angemar et al. 1984).

Bromacil can be applied at either the pre- or post-emergent stage of plant growth. It is most readily absorbed through the plant root system with lesser amounts entering through the foliage and stems. Once in the roots, bromacil is transferred from root to shoot into the chloroplasts of the leaves (Shriver and Bingham 1973).

Bromacil is a powerful and selective inhibitor of photosynthesis (Hilton et al. 1964; Hoffman et al. 1964; Hoffman 1971). It inhibits photosynthesis in vascular plants, algae and blue-green bacteria by blocking a step in the electron transport chain of photosystem II (Scott 1987).

The movement of bromacil in soil seems to be closely related to soil water (Landsburg and Reinl-Dwyer 1989). Increased amounts of water may increase the degree of downward movement (Reed and Holt 1982). Weber (1972) showed that bromacil moved laterally over the soil surface in surface waters and that it leached vertically into the soil profile. Bromacil is also much less subject to adsorption on soil colloids than many other herbicides. The lower adsorption may reflect the high solubility (815 ppm in water at 25 degrees celsius) and leaching characteristics discussed

earlier. Increased clay and organic matter concentrations have been found to increase bromacil adsorption in soil (Scott 1987; Hague and Coshow 1971; Angemar et al. 1984).

There are several possible methods of bromacil dissipation including plant uptake, adsorption, microbial degradation and leaching. Volatilization and photo-decomposition are thought to be of minor importance (Bingeman et al. 1962; Gardiner 1975).

## Field Reclamation Program

### Bromacil Monitoring Program

Bromacil has become a problem in reclamation within AGTD due to its use for total vegetation control in Hyvar (spray) and Calmix (pellets) on above ground facilities. The first AGTD bromacil contaminated site was identified in 1983 and more have since been identified as having phytotoxic concentrations of bromacil (0.1 ppm) both on and off-site. Concentrations have ranged from 0.1 to 16 ppm. Bromacil contaminated sites have been identified through various departments within AGTD, by landowners and Alberta Environment.

The AGTD Bromacil Monitoring Program involves the annual monitoring of those sites identified as having phytotoxic concentrations of bromacil as described above. Each year soil samples are taken from the facility pad as well as the affected off-site area. Samples are taken in 10 to 15 cm increments from the soil surface down to a minimum depth of 30 cm, or lower depending upon the concentration of bromacil. The soil sampling design is site specific and depends upon the: mode of contamination; size and shape of the affected area; topography of the area; vegetative growth; and the possibility of additional contamination from a neighbouring facility. Samples are collected and then sent to a laboratory for analyses for total bromacil, pH and electrical conductivity (EC). When results of the analyses are received, the data is interpreted and recommendations are made for handling the site and these are sent to the appropriate land agent. Each site is sampled yearly until there is no longer a problem and/or reclamation attempts have been successful.

There are a number of reclamation options available. The first is to do nothing to the site but pay crop compensation until the problem is gone. The second is to remove the contaminated soil and replace it with uncontaminated soil from elsewhere. Problems with this technique include finding a landfill that will accept bromacil contaminated soil, finding a good quality source of uncontaminated soil, and county by-laws concerning the removal of topsoil.

The third reclamation option and one used most widely within AGTD is to treat the contaminated soil in place. Treatments include: ditching and berming around the facility station pad to prevent further off-site contamination; applying activated charcoal to the affected off-site area to adsorb the bromacil; applying manure and fertilizers to the affected area to promote the adsorption and microbial decomposition of the bromacil; and seeding the site to barley or wheat, letting the crop germinate and take up some bromacil and when the crop dies, removing it and reseeding the area to promote more plant uptake and deactivation of bromacil.

The summer of 1989 should see the completion of the Bromacil Monitoring Program within AGTD. Those sites requiring reclamation will be amended with the suitable materials and plant growth monitored as an indication of reclamation success.

### Facility Retirements

In 1988 three meter stations were assessed for retirement or decommissioning. Field investigations included a site description, soil classification and sampling both on and off the meter station. Samples were analyzed for both organic and inorganic components. Follow-up included interpretation of the field data and analytical results. For each of the three sites, a large portion of the retirement plan involved the reclamation of bromacil contaminated soil.

Final recommendations for retirement included: removing the surface 20 cm of the meter station pad due to bromacil contamination followed by the addition of organic matter to the remaining soil to promote residual bromacil degradation; mixing 30 kg of activated charcoal into the surface 30 cm of a 0.2 ha site to promote the adsorption of bromacil; and incorporating 20 kg of activated charcoal into the surface 30cm of a 0.2 ha site to adsorb bromacil before replacing previously removed topsoil.

## Dissipation Of Bromacil In Humic Luvic Gleysols In Northwestern Alberta

On August 24, 1983 bromacil in the form of Hyvar XL was sprayed on the Mulligan Creek Meter Station in northwestern Alberta. The following year crops failed to grow on portions of the adjacent farm field. The affected area was in a discontinuous strip parallel to Highway 681 extending approximately 500 m east from the meter station. A preliminary investigation in July 1984 indicated that soil from the affected area was contaminated with bromacil.

In April 1985, two sites were selected within the bromacil contaminated area east of the meter station. Each site was selected to ensure location within the contaminated area. Site 1 was 12 m long and 12 m wide, and Site 2 was 4 m wide and 36 m long. Each was divided up into nine, four by four metre sampling plots or replicates.

In May 1985, approximately 20 cm of well-decomposed cow manure was applied and disced into Site 1 and Site 2. Anhydrous ammonia at a rate of 56 kg/ha and 112 kg/ha of the fertilizer 11-51-0 was also added to both sites. Each subsequent year the sites were fertilized with anhydrous ammonia at a rate of 70 kg/ha and 11-51-0 at 30 kg/ha. The cropping pattern consisted of a wheat-alfalfa rotation.

In November 1984, eighteen locations were sampled for background soils information: nine from an area surrounding and including Site 1, and nine from an area surrounding Site 2. Soil samples were collected in August 1985, 1986 and 1987 from each of the eighteen sampling plots, nine from Site 1 and nine from Site 2. At each sampling location a pit was excavated. Soil samples were taken in 4 cm increments from the vertical face of a soil pit from the soil surface down to 40 cm. Each sample was placed in a plastic bag and stored frozen until analyzed.

Inorganic analyses were performed using techniques listed in McKeague (1978) as follows: pH (3.14); electrical conductivity (3.21); particle size analysis (2.12); and total carbon (3.611). Total bromacil determinations were done using a shake/sonicate method developed for this project.

For the statistical analyses, Sites 1 and 2 were treated as two separate experiments as preliminary statistical evaluations showed the background soil samples from each site to be significantly different in terms of pH, electrical conductivity (EC), and organic carbon (OC) (Landsburg and Reinel-Dwyer 1989). Significant differences between means at each site were determined using confidence intervals according to the procedure outlined in Snedecor and Cochran (1980) at the 95% confidence level. This procedure was used to determine significant differences between each study year, at each depth, for each parameter monitored.

Through both the addition of cow manure and root turnover, OC contents increased significantly ($p \leq 0.05$) at both sites over the three year study (Landsburg and Reinel-Dwyer 1989). The increase from 1984 to 1985 ranged from 0.8 to 3.4 percent in the surface 0 to 12 cm. The percent OC decreased in 1986 from the 1985 level by anywhere from 0 to 2.0 percent in the same depth interval. In 1987, OC levels increased by 0.2 to 3.3 percent in the 0 to 16 cm depth intervals, probably due to the sporadic alfalfa, wheat and weed growth on the sites. Interestingly, the largest increases in 1987 occurred at the 8 to 16 cm depth.

The addition of manure in 1985 resulted in a significant ($p \leq 0.05$) increase in pH levels from 1984 to 1985 (1.2 to 2.2 units change in the 0 to 12 cm depth and 0.3 to 0.4 units at 12 to 16 cm) (Landsburg and Reinel-Dwyer 1989). In 1986, pH values decreased in the surface 0 to 16 cm by 0.2 to 1.2 pH units. This was likely due to the impact of microbial decomposition of the added organic material. In 1987, pH values increased by 0.2 to 1.3 pH units in the surface 0 to 16 cm. This change was probably due to the re-establishment of the soil equilibrium following large hydronium ion additions from the microbial decomposition activity in 1986. These trends are similar to those reported for organic carbon.

Electrical conductivity values were significantly increased ($p \leq 0.05$) from 1984 to 1985 in the 0 to 12 cm layer at Site 1 (range of 0.5 to 2.9 mS/cm increase) and in the 0 to 16 cm depth at site 2 where the increases varied from 1.3 to 6.0 mS/cm (Landsburg and Reinel-Dwyer 1989). The EC values below 20 cm significantly decreased at both sites

over this time by anywhere from 0.2 to 2.1 mS/cm. By the time of the 1987 sampling, EC values had significantly decreased in the surface 0 to 12 cm at Site 1 (by 0.1 to 2.1 mS/cm) and in the 0 to 20 cm depth at Site 2 (by 0.2 to 5.5 mS/cm). The 24 cm and lower depth at Site 1 showed significant increases in the range of 0.3 to 0.4 mS/cm while at Site 2 significant increases occurred at all sampling intervals below 20 cm. The range of the increases was between 0.6 to 1.2 mS/cm. These trends in EC levels suggest that leaching of salts is predominant over upward migration through capillary movement of the soil solution. This also suggests that water soluble bromacil was also moving primarily downward.

Since the laboratory technique used in this project analyzed for total bromacil, that amount adsorbed by organic matter was not determined. Only those dissipation methods resulting in complete removal of bromacil from the soil system will be discussed. Plant uptake, photo-decomposition and volatilization are assumed to be negligible in terms of the dissipation of bromacil at Sites 1 and 2.

Between 1985 and 1987, bromacil concentrations decreased within the surface 12 cm at Site 1. Below 12 cm, concentrations decreased in 1986 and increased in 1987. These trends for bromacil coincided with those exhibited by the other parameters monitored. The addition of manure to the surface 12 cm of the soil at Site 1 correlated well with the decrease in bromacil within the same depth over time. The decrease and subsequent increase in bromacil below 12 cm can be attributed to leaching. At Site 2 between 1985 and 1987, bromacil concentrations decreased at all depths monitored. Microbial degradation and leaching were probably responsible: microbial degradation within the surface 24 cm where OC percentages were increased by the manure, and leaching below 24 cm.

Conclusions reached as to the dissipation of bromacil in Humic Luvic Gleysols in northwestern Alberta were as follows:

- OC promoted the microbial degradation of bromacil at Sites 1 and 2. Degradation occurred within the same depths as the increased OC percentages.

- Bromacil concentrations decreased within the surface 12 cm at Site 1 and throughout the entire soil profile at Site 2. Site 1 had restricted dissipation to 12 cm due to the presence of a high water table which reduced leaching.

- The two apparent methods of bromacil dissipation in Humic Luvic Gleysols in northwestern Alberta were microbial degradation and leaching.

Laboratory Degradation Of Bromacil

The rehabilitation of bromacil contaminated topsoil from Chernozemic and Luvisolic soils is being studied in greenhouse and laboratory experiments as part of a Ph.D. research project being conducted at the University of Texas (Scott 1989). The purpose of the study is to determine methods for the rehabilitation of soil contaminated by bromacil.

Topsoil from a Calcareous Eluviated Black Chernozem and an Orthic Gray Luvisol was used for the study. Uncontaminated soils were collected in 1985, packed in separate copolymer plastic containers and shipped to the Controlled Ecosystem Lab at the Brackenridge Field Laboratory of the University of Texas in Austin.

Orthic Gray Luvisols have a well developed eluviated A horizon of 0 to 25 cm; may or may not have an Ah; they have a Bt horizon; and usually have a mean annual soil temperature of <8°C. The Orthic Gray Luvisol study soil was silt loam in texture with a pH (1:1 soil/water) of 5.4 and a cation exchange capacity (CEC) of 24.5 mg/100 gm. Eluviated Black Chernozems have well developed Ah horizons of 15 to 40 cm in thickness with an organic carbon content of 1 to 17 percent. They also have a Bt horizon and a mean annual soil temperature of 0 to 5.5°C. This Eluviated Black Chernozem was clay loam in texture with a pH of 7.5 and a CEC of 44.3 mg/100 gm.

Amendments used for the rehabilitation of the contaminated topsoil included commercially available cow manure, organic fiber, chemically modified peat and activated carbon powder. All experiments were conducted using 98% purity bromacil.

Adsorption experiments showed the Chernozem to have adsorbed more bromacil than the Luvisol. The clay and organic matter content of a soil are generally regarded to be the components which

adsorb most of the herbicide (Appleby 1985). The results from the adsorption experiments show this to be the case when comparing the Chernozem and Luvisol. The Chernozem, which had a high clay and high organic matter content compared to the Luvisol, adsorbed more bromacil (Scott 1989).

It is well known that the major method of bromacil degradation in the soil is by microbial organisms (Nemec and Tucker 1983; Pancholy and Lynd 1969). Preliminary Controlled Ecological Life Support System (CELSS) experiments showed that bromacil did not affect respiration, therefore most soil fungi and bacteria were not directly inhibited (Scott 1989). As a result, enhancing the soil microbial community should assist in the adsorption and degradation of bromacil. Enhancing a microbial community could be done by adding amendments required for microbial growth, and providing a suitable environment for growth (i.e. appropriate temperature and moisture). The addition of manure to bromacil contaminated soils in these experiments showed a slight decrease of biologically available bromacil compared to the no amendment contaminated control after a one year incubation period (Scott 1989).

Extensive adsorption experiments showed that powdered activated carbon amended to soils at rates as low as 0.1 g/kg soil doubled the adsorptive capacity for bromacil of the Chernozem and increased the adsorptive capacity of the Luvisol by eight times (Scott 1989). Therefore, the phytotoxicity of bromacil in soils can be reduced by the addition of activated carbon powder. Rates of activated carbon to be added to soil depend on the soil type, specific soil parameters, and the amount of bromacil in the soil.

Scott's (1989) results showed that activated carbon assists in the rehabilitation of bromacil contaminated soils by first adsorbing the bromacil and second, providing conditions conducive to the microbial growth which is responsible for degrading bromacil. Therefore after adding activated carbon as a slurry, the soil should be cultivated and dried to provide proper conditions for bromacil molecules to be adsorbed to the activated carbon. The contaminated area should then be supplemented with manure after the soil dries and then re-cultivated and watered (and possibly fertilized) to provide the proper environment for soil microbial growth.

Adding activated carbon to the soil not only increases the soil's adsorptive capacity, but alters other soil properties as well. The moisture holding capacity and pH are properties affected by carbon (Scott 1989). The soil pH was found to decrease with the addition of activated carbon while the moisture holding capacity increased. Therefore, the levels of activated carbon used for rehabilitation of bromacil contaminated soils should take into account a given soil's field capacity and its soil-moisture dynamics. Information concerning the soil type and texture, as well as site location and habitat, and climatic conditions must be taken into account when determining levels of activated carbon to be used as a rehabilitation amendment.

Results of this research project included recommendations for the rehabilitation and reclamation of soils contaminated by bromacil. These included the use of activated carbon, manure, wet-dry soil cycles, and plant seedlings to enhance adsorption, absorption and degradation of bromacil.

## Summary

The AGTD of NOVA Corporation of Alberta has more than 14,000 km of pipeline rights-of-way to operate and maintain. Above ground facilities associated with the pipeline system include approximately 850 meter stations, 36 compressor stations, numerous valve sites and several office and shop facilities.

Total vegetation control at above ground facilities is an attempt to eliminate all vegetation under and within the fenced areas. In the past, total vegetation control was accomplished through the use of "soil sterilants" such as Hyvar-XL and Calmix. With the discovery of migration of soil sterilants outside fenced areas, vegetation control has recently been attempted using non-residual products such as Roundup. Mechanical methods involve using mowers, weed-eaters or hand-picking.

Reclamation problems associated with vegetation management in AGTD are directly related to the use of soil sterilants. Twenty-five above ground facilities have been monitored annually as a result of bromacil contamination. These sites will be reclaimed in 1989.

Two research projects were initiated a number of years ago to determine the best methods of bromacil dissipation in the field as well as the best methods of rehabilitating bromacil contaminated topsoils in the laboratory and greenhouse. The two methods of bromacil dissipation found occurring in the field were leaching and microbial degradation. Recommendations for the rehabilitation and reclamation of soils contaminated by bromacil included the use of activated carbon, manure, wet-dry soil cycles, and plant seedlings to enhance adsorption, absorption, and degradation of bromacil. In our experience, plants will grow to maturity on the site when bromacil reaches a concentration of <0.1 ppm, but it is species dependent.


## Acknowledgements

The authors acknowledge C. Cloutier for her work on the Bromacil Monitoring Program, E. Reinl-Dwyer for her work on bromacil at Mulligan Creek and K. Scott for his work on bromacil in the lab and greenhouse.


## Literature Cited


Angemar, Y., M. Rebhun and M. Horowitz. 1984. Adsorption, phytotoxicity and leaching of bromacil in some Israeli soils. J. Environ. Qual. 13:321-326.

Appleby, A.P. 1985. Factors in examining fate of herbicides in soil with bioassays. Weed Sci. 33(2):2-6.

Beste, C.E., N.E. Humburg, H.M. Kempen, R.O. Radke, J.D. Riggleman, J.F. Stritzke and G.R. Miller. 1983. Herbicide handbook of the weed science society of America. Champaign, Illinois. pp.65-71.

Bingeman, C.W., R.W. Hill, R.W. Varner and TuA. Weidenfeller. 1962. Substituted uracils for industry weed control. Proc. N. Cent. Weed Centr. Conf. 19:42-43.

Gardiner, J.A. 1975. Substituted uracil herbicies. In: P.C. Kearney and D.D. Kaufman, eds. Herbicides: chemistry, degradation and mode of action. Marcel Dekker, Inc. New York.

Hague, R. and W.R. Coshow. 1971. Adsorption of isocil and bromacil from aqueas solution onto some mineral surfaces. Environ. and Sci. Techn. 5(2):139-141.

Hassall, K.A. 1982. The chemistry of pesticides: their metabolism, mode of action and uses in crop production. MacMillan Press Ltd., London.

Hilton, J.L., T.J. Monaco, D.E. Moreland and W.A. Gentner. 1964. Mode of action of substituted uracil herbicides. Weeds 12:129-131.

Hoffman, C.E., J.W. McGahen and P.B. Sweetser. 1964. Effect of substituted uracil herbicides on photosynthesis. Native 202: 577-578.

Hoffman, C.E. 1971. The mode of action of bromacil and related uracils. p.65-85. In: Proceedings: 2nd Int. Conf. on Pest. Chem.

Landsburg, S. and E. Reinl-Dwyer. 1989. Dissipation of bromacil in humic luvic gleysols in northwestern Alberta. NOVA Corporation of Alberta. Calgary, Alberta.

McKeague, J.A. 1978. Manual on soil sampling and methods of analysis. 2nd ed. Subcommittee on Methods of Analysis. Canadian Society of Soil Science. Ottawa, Ontario.

Nemec, S. and D. Tucker. 1983. Effects of herbicides on the endomycorrhizal fungi in Florida citrus (Citrus spp.) soils. Weed Sci. 31:427-431.

Pancholy, S.K. and J.Q. Lynd. 1969. Bromacil interactions in plant bioassay, fungi cultures and nitrification. Weed Sci. 17(6):460-463.

Reed and Holt. 1982. Leaching of selected total vegetation control herbicides. Proc. Northeast Weed Sci. Soc. 36:39-63.

Scott, K.W. 1987. Laboratory and greenhouse studies for the rehabilitation of bromacil contaminated soils: annual report 1986/87. Prepared for NOVA Corporation of Alberta. Calgary, Alberta. File report.

Scott, K. 1989. Laboratory and greenhouse studies for the rehabilitation of bromacil contaminated soils: annual report 1988. NOVA Corporation of Alberta. Calgary, Alberta. File report.

Shriver, J.W. and S.W. Bingham. 1973. Physiological effects of bromacil in Kentucky bluegrass and orchard grass. Crop Science. 13:45-49.

Snedecor and Cochran. 1980. Statistical methods. 7th ed. Am. Soc. Qc. Milwaukee, Wi.

Weber, J.B. 1972. Interaction of organic pesticides with particulate matter in aquatic and soil systems. p.55-120. In: R.F. Gould (ed). Fate of organic pesticides in the aquatic environment. Advan. Chem. Ser. III:55-120. Amer. Chem. Soc. Washington. D.C.

# COMPARISON OF DIFFERENT MULCHES, LIME AND FERTILIZER APPLICATIONS ON THE ESTABLISHMENT AND GROWTH OF VEGETATION ON SURFACE COAL MINING LAND[1]

by

Fred J. Brenner[2]

Abstract. The objective of the current study was to determine the effect of different mulches, lime and fertilizer applications on the establishment and growth of vegetation on two surface mine sites in western Pennsylvania. One site was reclaimed under the current regulations (1977) while the second site was mined and reclaimed prior to the passage of PL 95-87. Experimental quadrats were established in a checkerboard manner and all but one quadrat, which served as a control, received either mulch, lime and/or fertilizer alone or in combinations of mulch, lime and/or fertilizer. Quadrats were seeded with either annual rye (Lolium multiform), winter wheat (Triticum aestivum), K-31 fescue (Fescue aurndinacea), alfalfa (Medicago sativa) or buckwheat (Fagopyrum sagittatum). Generally, those quadrats that were mulched with either straw or bark enhanced the establishment of vegetation to a greater degree than newsprint, which in some cases was detrimental, when compared to the control. However,considerable variation existed among the different species in their response to mulch and/or lime fertilizer applications. Generally the response of vegetation to all treatments was better on the site with top soil replacement than it was on the older site which was reclaimed in the early 1960's.

Additional key words: Mulch, Lime, Fertilizer, Establishment, Growth, and Biomass of Winter Wheat, Annual Rye, Fescue, Alfalfa, Buckwheat.

## Introduction

The application of mulch, lime and fertilizer has been a standard practice in reclaiming mine lands for over two decades. Previous studies have shown that mulching will enhance the establishment and growth of vegetation on disturbed lands (Dyer et al. 1984). These authors found that bark was slightly superior to hay as a mulch but that no single mulch was best for all soil types. These authors further indicated that considerable variability existed in the response of vegetation to different mulches and that some mulches could have an adverse effect on vegetation. In general, the use of mulches which are high in organic matter, such as bark chips, hay and straw, were better for the establishment of legumes than for grasses. In the southwest, Elkins et al. (1983) found that bark as an organic amendment to minesoils was as effective in developing a productive soil profile as the more expensive applications of topsoil and mulch currently recommended for surface mine reclamation.

Lime and fertilizer applications prior to seeding is a common practice in re-vegetating mined lands. DePuit and Conenbert (1979) found that a light fertilizer application may be best to enhance species diversity while prolonged fertilizer applications will increase biomass production.However,fertilizer applications at maintenance levels had little long-term impact on soil fertility. In another study, Hanson et al.(1982) found that lime application increased yield for one year but not the following year and that there was not a significant difference between sites receiving lime applications and those which did not. McFee et al. (1981) found that the application of 10-10-10 fertilizer improved the growth of both wheat and alfalfa on all types of overburden except black and gray shales. It appears therefore, that lime fertilizer applications may be important for the initial establishment of vegetation but may not be important in maintaining long-term fertility and biomass production unless fertilizer application is continued on an annual basis.

## Materials and Methods

Experimental $3m^2$ quadrats were established in a 4x4 checkerboard design on two mine sites (3 replicas per site) located in the glaciated plateau providence of northwestern Pennsylvania. The overburden overlaying the Brookeville coal of the Clarion Formation consisted of glacial till along with shale and sandstone directly

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston,West Virginia,April 23-26,1990.

[2]Fred J. Brenner is a Professor of Biology Grove City College, Grove City, PA 16127.

above the coal seam. The glacially derived soils on both sites had a pH between 6.0 and 6.6 and the organic content between 3.0% and 4.8% with a moisture content at saturation between 15 and 22%.

The first site, hereafter referred to as site one, was mined after the passage of PL 95-87, whereas the second site, hereafter referred to as site two, was mined in the early 1960's.

All quadrats were established at the same time and were located on a level contour. A wood frame buried at a depth of 15cm separated the individual quadrats. Quadrats, which served as controls, received no treatment while the others were mulched at a rate equivalent to 3 tons/acre of either straw, bark or newsprint as mulch alone, or received various combinations of lime, fertilizer or lime and fertilizer in addition to mulching. The quadrats were limed at a rate equivalent to 2 tons/acre and 10-20-20 fertilizer applications were equivalent to 200 lb/acre as recommended by the Pennsylvania State Agronomy Guide (1987-88).

Quadrats were seeded with either annual rye (Lolium multiforum), winter wheat (Triticum aestivum), K-31 fescue (Fescue aurndinacea) and alfalfa (Medicago sativa) or buckwheat (Fagopyrum sagittatum). Winter wheat was seeded in September and all other seedlings were completed prior to 31 May. The seeding rates were based on pounds of live seeds/acre (PLS/A) as recommended in the Pennsylvania State Agronomy Guide (1987-88) for seeding on disturbed lands and establishing foliage and small grains (Table 1). All quadrats were seeded with a cyclone seeder to insure as uniform a seeding as possible.

At the end of the first growing season, the number of stems and height of all individual plants on each quadrant was determined for winter wheat, annual rye and buckwheat and the number of stems, height and the plant and root biomass/$m^2$ was also determined for buckwheat. After the second growing season, the biomass, chlorophyll a and plant height and root length of K-31 fescue and alfalfa was determined on each quadrat. The amount of chlorophyll a was also determined according to the procedures described by Richards and Thompson (1952), Parson and Strickland (1963), and Brenner et al. (1975). Chlorophyll a has been used as a factor in determining the importance of different species occurring on reclaimed mine lands by Brenner et al. (1975) and, thus, was used as another index of the effect of different treatments on the establishment and growth of vegetation on the different quadrats. In addition to measuring the vegetation, a soil sample was obtained from each quadrat to determine the effect of the different treatments on organic matter and moisture on pH. Since these parameters have been shown to be important in the establishment of vegetation on mined lands (Brenner et al. 1984).

## Results and Discussion

Considerable variation existed among the different treatments in both soil characteristics as well as in the response of vegetation. The use of mulch had no significant effect on the amount of organic matter or the moisture content of the surface mine spoil, either among treatments or between the two mine sites as indicated by a two-way analysis of variants ($P > 0.75$).

Variation also existed among the different species of vegetation in their response to the application of the various mulches and lime-fertilizer treatments between the two mine sites. For example, with winter wheat, the number of plants per $m^2$ on the first mine site was greater when straw was used as a mulch than it was when quadrats were mulched with either bark or newsprint. However, on site two the number of plants/$m^2$ to a greater extent on site one than they did on site two. On site one, the greatest number of plants occurred when straw was used in combination with fertilizer, but there was not a significant difference in the numbers of plants/$m^2$ between the use of fertilizer and the use of lime-fertilizer ($P > 0.50$). However, when bark was used as a mulch, the number of plants/$m^2$ was greater when lime-fertilier was used, but there was not a significant difference ($P > 0.50$) between no lime fertilizer application and when

Table 1. Seeding rates of five different species on two surface coal mines in Pennsylvania receiving different treatments of mulch and lime and fertilizer applications.

| SPECIES | SEEDING RATES - PLS (LB/A) |
|---|---|
| Annual Rye (Lolium multiforum) | 40 |
| Winter Wheat (Triticum aestivum) | 60 |
| K-31 Fescue (Fescue aurndinacea) | 20 |
| Alfalfa (Medicago sativa) | 20 |
| Buckwheat (Fagopyrum sagittatum) | 60 |

fertilizer was used (Table 2). On site two except for newsprint, the number of plants/$m^2$ was greater on the control quadrat and on those receiving mulch than it was among the different lime and fertilizer treatments.

In contrast to winter wheat annual rye had a better response at site two to all treatment categories with the possible exception of when newsprint was used as a mulch. On site two, the greatest number of plants/$m^2$ occurred when no treatment was used. The greatest number of plants/$m^2$ among different treatments occurred when either straw or bark was used as a mulch in combination with lime (Table 2). When both lime and fertilizer or fertilizer alone was used, the number of plants/$m^2$ increased when straw was used as a mulch, but not on quadrats mulched with bark or newsprint. On one site, the number of plants/$m^2$ were greater when straw was used as a mulch in combination with lime and fertilizer, with the exception of the control quadrats or when newsprint was used as a mulch. On quadrats mulched with straw or bark, the number of plants/$m^2$ varied among the different treatments.

On mine site one, the use of straw along with the combinations of lime and fertilizer increased the growth of both winter wheat and rye compared to the control quadrats. But when bark was used as a mulch, winter wheat showed a greater growth response than annual rye. On site two, the use of straw with the addition of either lime or fertilizer resulted in a greater increase in the plant growth than when bark was used as a mulch. The least amount of growth on both sites occurred on quadrats mulched with newsprint, regardless of whether fertilizer or lime were applied to the site (Table 2).

K-31 fescue exhibited considerable variation among the different treatments both within and between the two mine sites. On site one, the greatest root biomass occurred on quadrats which were not mulched but received applications of lime and fertilizer, while the greatest above-ground biomass with the smallest root biomass occurred when the quadrats receiving straw along (Table 3). There was considerable variation in the height and biomass of vegetation among the other mulches as well as between treatments with lime and fertilizer (Table 3). The tallest plants occurred on quadrats mulched with newsprint in conjunction with lime, but considerable variation occurred among the other treatments. On site two, the greatest above-ground biomass occurred when bark alone was used as mulch, but the greatest root biomass occurred when bark was used in conjunction with lime and fertilizer. Overall, there was no significant difference in the

Table 2. The number of stems/$m^2$ and height of winter wheat (*Triticum aestivum*) and annual rye (*Lolium multiforum*) on quadrats receiving different mulch, lime and fertilizer applications.

| | Number/$m^2$ | | | | Height mm | | | |
|---|---|---|---|---|---|---|---|---|
| | Winter Wheat | | Annual Rye | | Winter Wheat | | Annual Rye | |
| | Site | | | | Site | | | |
| Treatment | 1 | 2 | 1 | 2 | 1 | 2 | 1 | 2 |
| No mulch | 130 | 48 | 49 | 116 | 618 | 482 | 522 | 505 |
| Lime | 98 | 10 | 48 | 55 | 522 | 313 | 499 | 513 |
| Fertilizer | 53 | 8 | 24 | 42 | 261 | 250 | 485 | 324 |
| Lime-Fertilizer | 50 | 21 | 30 | 53 | 315 | 478 | 491 | 345 |
| Straw | 153 | 49 | 28 | 54 | 989 | 511 | 788 | 498 |
| Lime | 178 | 8 | 47 | 80 | 1012 | 486 | 791 | 987 |
| Fertilizer | 235 | 9 | 53 | 46 | 1058 | 361 | 496 | 965 |
| Lime-Fertilizer | 235 | 5 | 58 | 62 | 1062 | 483 | 601 | 503 |
| Bark | 105 | 78 | 25 | 47 | 510 | 281 | 302 | 266 |
| Lime | 153 | 11 | 48 | 77 | 768 | 260 | 328 | 278 |
| Fertilizer | 102 | 3 | 52 | 25 | 518 | 272 | 343 | 485 |
| Lime-Fertilizer | 155 | 1 | 54 | 48 | 783 | 291 | 331 | 472 |
| Newsprint | 53 | 3 | 2 | 8 | 260 | 262 | 452 | 183 |
| Lime | 52 | 3 | 3 | 28 | 260 | 105 | 448 | 101 |
| Fertilizer | 25 | 1 | 15 | 8 | 125 | 245 | 435 | 114 |
| Lime-Fertilizer | 50 | 3 | 4 | 12 | 266 | 251 | 410 | 256 |

height and among the biomass of vegetation different mulches or lime and fertilizer applications a indicated by a two-way analysis of variance ($P > 0.50$).

As with K-31 fescue, alfalfa also showed a considerable variation among the different treatment but there was not as great a difference between the two sites as occurred with K-31 fescue or winter wheat and annual rye. In terms of biomass, site one had a slightly greater biomass per $m^2$ among the different treatments than site two (Table 4). Overall, on site one quadrats with mulch had a greater biomass/$m^2$ than the control quadrats, with the possible exception of the control quadrats that received lime and fertilizer applications but differences occurred in the amount of above-ground and root biomass on site one among the different treatments (Table 4). On site two plant and root length, was greatest on the quadrats that received bark mulch without lime and fertilizer and there was not a significant difference among the other treatments in either the plant height or root length. Also on site two, the greatest biomass occurred on the control quadrats receiving the lime applications (Table 4). Generally the plants obtained their greatest height and bioma s/$m^2$ when bark and newsprint were used as a mulch rather than straw.

As with biomass/$m^2$ and plant height, the mg chlorophyll a kg varied among the different treatments on the two sites. In general, among the different treatments the chlorophyll a/kg was greater in fescue than it was in alfalfa. On site one, the greatest concentrations of chlorophyll a occurred in fescue on the quadrats mulched with bark or newsprint along with receiving lime and fertilizer application ($P > 0.50$). As with site one, the greatest concentration of chlorophyll a occurred in plants on the control quadrat receiving lime and fertilizer applications (Table 3,4).

The number of buckwheat plants/$m^2$ on both sites tended to be greater on quadrats mulched with straw or bark than it was on the control quadrat or those mulched with newsprint (Table 5). In regards to the effects of lime and fertilizer treatments on the number of plants/$m^2$ were significantly greater on quadrats receiving straw along or straw and lime that it was on the other quadrats ($P > 0.05$). However, there was not a significant difference in the number of plants/$m^2$ on the quadrats mulched with straw and those mulched with straw along with applications of lime and fertilizer ($P > 0.50$). When bark was used as a mulch, the number of plants/$m^2$ was essentially the same between the quadrats receiving fertilizer applications and those receiving no additional treatments. On site two, quadrats receiving straw and bark with no addition of lime and fertilizer had a greater number of plants/$m^2$ than those receiving various combinations of lime and fertilizer. Generally, the number of plants/$m^2$ on those plots mulched with newsprint did not differ significantly from the control quadrats ($P > 0.50$).

On site one, the greatest total biomass of buckwheat occurred on quadrats mulched with straw or bark in combination with either lime or fertilizer. While on site two, the greatest total bio-

Table 3. Height, Biomass, Dry Weight and Chlorophyll a concentration of K-31 fescue (Festucue aurndinacea) grown on quadrat receiving different mulch, lime and fertilizer applications.

| | Length mm | | | | Biomass g/$m^2$ | | | | Chlor. a | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Plant Site | | Root Site | | Plant Site | | Root Site | | mg/kg Site | |
| Treatment | 1 | 2 | 1 | 2 | 1 | 2 | 1 | 2 | 1 | 2 |
| No mulch | 70 | 145 | 50 | 85 | 10.5 | 12.7 | 2.5 | 4.8 | 23 | 28 |
| Lime | 61 | 97 | 99 | 8 | 8.3 | 8.7 | 3.0 | 1.5 | 27 | 27 |
| Fertilizer | 135 | 148 | 65 | 50 | 6.0 | 20.2 | 4.5 | 2.3 | 23 | 49 |
| Lime-Fertilizer | 100 | 168 | 90 | 25 | 9.7 | 14.8 | 19.8 | 3.2 | 24 | 48 |
| Straw | 100 | 95 | 50 | 98 | 29.0 | 18.0 | 2.0 | 5.0 | 48 | 74 |
| Lime | 97 | 70 | 53 | 53 | 9.8 | 17.8 | 8.0 | 10.1 | 26 | 49 |
| Fertilizer | 80 | 135 | 30 | 12 | 6.7 | 13.9 | 3.8 | 3.1 | 49 | 73 |
| Lime-Fertilizer | 79 | 105 | 31 | 98 | 17.0 | 15.8 | 7.5 | 4.7 | 53 | 72 |
| Bark | 117 | 152 | 63 | 47 | 10.3 | 35.1 | 2.2 | 9.7 | 26 | 77 |
| Lime | 149 | 186 | 51 | 9 | 5.7 | 18.0 | 5.8 | 3.8 | 75 | 50 |
| Fertilizer | 183 | 205 | 15 | 15 | 1.8 | 9.3 | 2.0 | 1.2 | 23 | 49 |
| Lime-Fertilizer | 135 | 120 | 75 | 98 | 6.8 | 33.5 | 3.1 | 17.5 | 27 | 75 |
| Newsprint | 80 | 120 | 40 | 30 | 8.2 | 14.8 | 1.5 | 2.2 | 10 | 49 |
| Lime | 125 | 144 | 75 | 52 | 5.4 | 12.5 | 4.5 | 5.6 | 12 | 47 |
| Fertilizer | 135 | 169 | 75 | 51 | 3.8 | 2.7 | 1.8 | 2.1 | 53 | 77 |
| Lime-Fertilizer | 147 | 159 | 73 | 51 | 3.3 | 17.0 | 1.7 | 2.2 | 25 | 76 |

Table 4. Height, biomass, dry weight and chlorophyll a concentration of alfalfa (Medicago sativa) grown on quadrats receiving different mulch, lime and fertilizer applications.

| | Length mm | | | | Biomass g/m$^2$ | | | | Chlor. a | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Plant Site | | Root Site | | Plant Site | | Root Site | | mg/kg Site | |
| Treatment | 1 | 2 | 1 | 2 | 1 | 2 | 1 | 2 | 1 | 2 |
| No mulch | 244 | 180 | 201 | 155 | 16.5 | 10.6 | 11.0 | 15.0 | 26 | 26 |
| Lime | 284 | 290 | 155 | 185 | 12.4 | 18.9 | 8.1 | 30.2 | 24 | 25 |
| Fertilizer | 256 | 136 | 199 | 50 | 9.7 | 7.8 | 10.5 | 2.2 | 26 | 27 |
| Lime-Fertilizer | 205 | 140 | 210 | 110 | 26.9 | 15.0 | 19.9 | 10.0 | 50 | 50 |
| Straw | 170 | 108 | 180 | 180 | 20.7 | 13.8 | 16.1 | 15.7 | 26 | 27 |
| Lime | 135 | 107 | 175 | 178 | 20.1 | 14.2 | 12.7 | 16.3 | 13 | 11 |
| Fertilizer | 225 | 187 | 174 | 100 | 18.6 | 29.0 | 17.9 | 10.0 | 14 | 24 |
| Lime-Fertilizer | 223 | 143 | 176 | 156 | 21.6 | 19.4 | 18.2 | 16.3 | 27 | 26 |
| Bark | 305 | 192 | 230 | 188 | 29.5 | 16.5 | 22.5 | 19.7 | 23 | 24 |
| Lime | 240 | 295 | 205 | 105 | 21.4 | 29.1 | 19.8 | 10.8 | 22 | 23 |
| Fertilizer | 320 | 269 | 135 | 130 | 28.3 | 17.8 | 11.5 | 11.2 | 15 | 13 |
| Lime-Fertilizer | 253 | 245 | 145 | 165 | 30.1 | 20.0 | 11.7 | 21.0 | 25 | 25 |
| Newsprint | 278 | 285 | 120 | 120 | 22.4 | 22.6 | 16.8 | 15.0 | 12 | 23 |
| Lime | 288 | 175 | 110 | 157 | 34.0 | 16.5 | 17.2 | 16.3 | 13 | 24 |
| Fertilizer | 210 | 237 | 113 | 158 | 15.9 | 21.6 | 16.1 | 16.4 | 27 | 27 |
| Lime-Fertilizer | 335 | 210 | 110 | 200 | 24.6 | 20.0 | 17.4 | 19.9 | 25 | 24 |

Table 5. Number of stmes/m, height and biomass and dry weight of buckwheat (Fagopyrum sagittatum) grown on quadrats receiving different mulch, lime and lime and fertilizer applications.

| | Number of stems/m$^2$ | | Length mm | | | | Biomass g/m$^2$ | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Site | | Plant Site | | Root Site | | Plant Site | | Root Site | |
| Treatment | 1 | 2 | 1 | 2 | 1 | 2 | 1 | 2 | 1 | 2 |
| No mulch | 77 | 61 | 345 | 260 | 35 | 15 | 5.0 | 93.5 | 17.0 | 16.5 |
| Lime | 78 | 60 | 352 | 259 | 28 | 17 | 79.0 | 47.0 | 22.0 | 16.8 |
| Fertilizer | 101 | 62 | 453 | 282 | 35 | 18 | 103.0 | 146.5 | 22.0 | 22.4 |
| Lime-Fertilizer | 49 | 59 | 380 | 580 | 25 | 50 | 82.5 | 85.9 | 16.0 | 15.2 |
| Straw | 155 | 105 | 263 | 390 | 25 | 15 | 21.5 | 39.8 | 22.0 | 15.8 |
| Lime | 161 | 75 | 367 | 372 | 28 | 13 | 99.4 | 32.8 | 101.1 | 16.0 |
| Fertilizer | 248 | 78 | 377 | 499 | 28 | 16 | 200.9 | 115.8 | 21.9 | 22.7 |
| Lime-Fertilizer | 53 | 79 | 371 | 589 | 24 | 10 | 22.7 | 170.3 | 16.1 | 10.0 |
| Bark | 125 | 140 | 227 | 271 | 18 | 14 | 14.0 | 110.5 | 8.5 | 8.9 |
| Lime | 75 | 76 | 350 | 387 | 28 | 15 | 189.4 | 178.8 | 100.5 | 22.0 |
| Fertilizer | 126 | 80 | 370 | 563 | 29 | 15 | 160.8 | 178.7 | 135.0 | 22.1 |
| Lime-Fertilizer | 52 | 85 | 403 | 627 | 12 | 18 | 18.0 | | 17.0 | 19.8 |
| Newsprint | 54 | 48 | 194 | 286 | 11 | 14 | 6.0 | 80.9 | 16.0 | 18.9 |
| Lime | 51 | 51 | 383 | 599 | 35 | 16 | 7.5 | 178.7 | 16.2 | 20.1 |
| Fertilizer | 20 | 52 | 451 | 573 | 34 | 17 | 7.3 | 254.1 | 15.0 | 45.8 |
| Lime-Fertilizer | 78 | 55 | 374 | 520 | 36 | 85 | 6.7 | 267.9 | 15.3 | 22.1 |

mass of buckwheat occurred when bark alone was used as a treatment followed by quadrats mulched with bark along with lime and fertilizer. Overall, on site one, there was not a significant difference among the various treatments ($P > 0.50$). However, those quadrats receiving mulch and/or lime-fertilizer applications tended to have a greater biomass/$m^2$ than those that received only mulch (Table 5). On site two, the tendency was for the greatest biomas /$m^2$ to occur on those quadrats mulched with newsprint. The tallest plants occurred on quadrats receiving lime and fertilizer applications but differences occurred among quadrats receiving different treatments.

As in previous studies, considerable variation existed in the response of the vegetation to the various treatments. Overall, traw and bark appeared to stimulate both the establishment and growth of vegetation on surface mine lands but the response varied among the different species. Likewise, the addition of lime and fertilizer either with or without the use of mulch tended to enhance the establishment and growth of vegetation, but variations also occurred with all species tested. In some instances, the growth was equal to or greater on the control plots which did not receive any additional treatment. In general, with all species tested, the response of vegetation among the various treatments was greater on site one which was reclaimed in accordance with the PL 95-87 regulations as to the placement of top soil than it was on the older mine site which was mined in the early 1960's.

## Acknowledgements

I would like to thank the Mercer County Conservation District for their assistance both financially and with personnel during the course of the study. I would also like to than Mr. Richard H. Crowley, retired District Conservationist, U.S.D.A. Soil Conservation Service, for his helpful suggestions during the study.

## Literature Cited

Brenner, F.J., R.H. Crowley, M.J. Musaus and J.H. Goth. 1975. Evaluation and recommendations of strip-mine reclamation procedures for maximum sediment-erosion control and wildlife potential. pp. 3-23. Third Symposium Surface Mining and Reclamation. Louisville, KY.

Brenner, F.J., M.W. Werner, and J. Pike. 1984. Ecosystem development and natural succession in surface coal mine reclamation. Minerals and the Environment 6: 10-22.

DePuit, E.J. and J.G. Conenberg. 1979. Responses of revegetated coal strip mine spoils to variable fertilization rates, longevity of fertilization programs and season seeding. Research Report 150. Montana Agricultural Experiment Station. Montana State University, Bozeman, Mont. 75p.

Dyer, K.L., W.R. Curtis, and J.T. Crews. 1984. Response of vegetation to various mulches used in surface mine reclamation in Alabama and Kentucky-7 year case history. Gen. Tech. Report NE 93. U.S. Depart. Agric. Forest Service, N.E. Exp. Station. 11p.

Elkins, N.Z., L.W. Parker, E. Aldon, and W.G. Whitford. 1983. Responses of soil biota to organic amendments in strip mine spoil in northwestern New Mexico. J. Environ. Qual.13:215-219.

Hanson, R.G., M. Tucker, A.D. Cable, and D.A. Steper. 1982. Yield, quality and K/(Ca - Mg) ratio of tall fescue breeding lines on amended and non-amended mine soil. Commun. in Soil Sci. Plant/Anal. 13: 1081-1094.

McFee, W.W., W.R. Byrnes, and J.G. Stockton. 1981. Characteristics of coal mine overburden important in plant growth. J. Environ. Qual. 10: 300-308.

Parsons, J.R. and J.D.H. Strickland. 1963. Discussions of spectrophotometric determinations of marine plant pigments with equations for ascertaining chlorophylls and carotoids. J. Mar. Res. 21: 155-163.

Richards, R.A. and T.G. Thompson. 1952.The estimation and characterization of plankton community by pigment analysis II. A spectrophotometric method for estimation of plankton pigments. J. Mar. Res. II: 156-172.

The Penn State Agronomy Guide. 1987-1988. The Pennsylvania State University. College of Agriculture Extension Service. Univ. Park. PA 148 p.

# ESTABLISHMENT OF NATIVE PLANT SPECIES FROM FOREST TOPSOIL SEEDBANKS ON A BORROW AREA IN KENTUKCY[1]

by

Gary L. Wade and Ralph L. Thompson[2]

Abstract. The seed banks in forest topsoils were used to introduce native species to an unreclaimed, xeric borrow area in eastern Kentucky. The results of using eight treatments are reported: (1) 1 cm topsoil + mulch, (2) 1 cm topsoil tilled into the substrate + mulch, (3) 2 cm topsoil + mulch, (4) 1 cm topsoil without mulch, (5) 4 cm topsoil in strips covering 25% of the area + mulch, (6) 1 cm topsoil + a grass-legume reclamation mix + mulch, (7) the reclamation mix + mulch without topsoil, and (8) mulch without topsoil or reclamation mix. The seed bank produced 90 species from 34 families including seven tree species, seven shrubs, 16 graminoids, and 50 forbs. Forest topsoil use introduced 57 native or naturalized species during the first growing season and 82 species were present during the second growing season. Average ground cover of native species totaled only about 5% after ten weeks of the first growing season. Wheat and timothy seed contained in the hay mulch increased mean ground cover to over 70%. The reclamation mix produced 96% cover during the first year and it completely suppressed establishment of native species during the first two years. Different methods of topsoil use without the reclamation mix did not significantly affect the number of native species established in the first and second growing seasons, cover in the first growing season, the number of species increase in the second year, or the number of established trees and shrubs.

Additional key words: Reclamation, revegetation

## Introduction

Drastic land disturbances such as surface mines, road banks, quarries, and construction projects with fills or borrow areas produce significant areas requiring revegetation and reclamation. Techniques for simple revegetation of these areas are well developed, but exotic and monotonous, highly competitive plant communities are the frequent result. Also, establishment and succession of desirable communities of native species may be slowed by current revegetation practices (Brenner 1984). Communities of native species can be established with commercially available seed and plant-int stock, but such materials tend to be expensive, and propagules of many native species are often not available.

Topsoil seed banks have been used effectively to introduce native species on reclaimed mined lands in the western United States (Beauchamp et al. 1975, Howard and Samuel 1979, Iverson and Wali 1982). However, the amount of cover produced by the seed bank species has been less than that obtained with exotic reclamation species. DePuit (1984) has synthesized some potentially successful strategies for enhancing vegetation and species diversity on reclaimed mined lands using topsoil seed banks.

Use of topsoil seed banks to introduce native species to reclaimed land has been successful in Australia. Tacey (1979) used the top 5 cm of topsoil to introduce native species of jarrah (Eucalyptus marginata) forest to mined lands in Western Australia. Dominance-diversity curves in reestablished vegetation approached the log-normality distribution found in more diverse communities after four years (Tacey and Glossop 1980). Topsoil seed banks have been used to reestablish many high dune forest and coastal wetland species after surface mining on the Australian east coast (Brooks 1988).

In the United Kingdom, chopped turf from old mined sites has been used to introduce native

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]Gary L. Wade is Botanist, Northeastern Forest Experiment Station, USDA Forest Service, Berea, KY 40403; Ralph L. Thompson is Associate Professor of Biology, Biology Department, Berea College, Berea, KY 40404.

species to newly mined sites. A 30 cm layer of topsoil was also successful in native species introduction, but a "few millimeters thick" and "small molehill" amounts were not effective (Wathern and Gilbert 1978, Gilbert and Wathern 1980).

Farmer et al. (1982) and Wade (1986) found that seed banks in eastern United States deciduous forests produced many native species in pioneer communities with high biomass and cover when forest topsoils were spread 2.5 to 5.0 cm thick over mine spoils in 1.3 $m^2$ microplots. Moreover, the growing conditions in these last two studies were optimal because of supplemental irrigation.

Quality of the seed bank is important for species introductions. Quality can be influenced by source soil types, vegetation, disturbance history, aspect, and season of collection. A soil may contain several types of seed banks that range from transient to persistent based on seed-dormancy characteristics of its component species. Different types of seed banks may be able to exploit certain types of disturbances at only certain times of the year (Grime 1981).

Many reclamation/revegetation projects occur on sites where stressful conditions of high temperatures, low fertility, and low available water potential exist. Large quantities of topsoils for large-scale projects can be difficult to obtain without disturbance to other areas on an unacceptable scale. The borrow area of Cowbell Hollow provided an opportunity to investigate: (1) what methods of application of minimal amounts of forest topsoil would establish pioneer communities from the contained seed bank, (2) whether exotic reclamation ground cover species would inhibit the invasion and establishment of desired native successional species, and (3) which species could be established after two growing seasons.

## Materials and Methods

The study area is a 7.0 ha borrow area within the Red Lick Watershed Control Basin below Cowbell Hollow Reservoir in the Berea College Forest, Madison County, Kentucky. The forest topsoils and part of the subsoils were removed for use as fill in a large earthen dam in 1985. The original soils were classified as Shelocta gravely silt loams (Newton et al. 1973). Only the very strongly acid subsoils remain with a depth of approximately 30 cm over sandstone and shale. Three study areas were established on a broad second terrace above Red Lick Creek below a long east-facing slope with an open southern exposure. The site can be classified as xeric because of its exposure and shallow soil parent materials, although water from upper slope positions sometimes reaches the plots during and after high precipitation events.

In March 1986, the borrow area was chisel-plowed to a depth of 25 cm and 10-10-10 fertilizer was applied at a rate of 1210 kg $ha^{-1}$. Three study areas were subdivided into eight 3 m x 3 m plots in line with a 1 m buffer strip separating the plots. Forest topsoil to a depth of about 15 cm was collected from a nearby undisturbed Virginia pine-red maple (Pinus virginiana-Acer rubrum) stand using a farm tractor with a front-mounted end loader and deposited adjacent to two of the study areas. Topsoil was similarly collected for use in a third area from an adjacent American beech-white oak (Fagus grandifolia-Quercus alba) stand.

The study was laid out as four independent experiments to study 1) effectiveness of topsoil for introducing native species, 2) effect of the reclamation species on native species, 3) effects of different methods of topsoil use, and 4) effects of mulch. Eight treatment plots were randomly assigned and replicated once in each of three blocks in the barrow area. Treatments were as follows:

- T1. 1 cm topsoil plus mulch.
- T2. 1 cm topsoil lightly tilled into soil with a rototiller plus mulch.
- T3. 2 cm topsoil plus mulch.
- T4. 1 cm topsoil minus mulch.
- T5. 1 cm topsoil equivalent applied in two strips 4 cm deep across the plot plus mulch.
- T6. 1 cm topsoil plus a reclamation species mix plus mulch.
- T7. No topsoil, reclamation species mix only plus mulch.
- T8. No topsoil, no reclamation mix, mulch only.

The required amounts of topsoils were spread April 4 on each plot by filling buckets with the appropriate volumes of topsoil, transferring the soil to the plot, and raking it evenly or stripping it over the surface. The reclamation species mix was seeded on April 9. It consisted of Japanese lespedeza (Lespedeza striata) seeded at 9 kg $ha^{-1}$, Korean lespedeza (Lespedeza stipulacea) at 8 kg $ha^{-1}$, white clover (Trifolium repens) at 5 kg $ha^{-1}$, perennial ryegrass (Lolium perenne) mixed with Italian ryegrass (L. multiflorum) at 9 kg $ha^{-1}$, orchard grass (Dactylis glomerata) at 11 kg $ha^{-1}$, and timothy (Phleum pratense) at 4 kg $ha^{-1}$. A common wheat (Triticum aestivum) hay mulch was spread at the rate of 2 t $ha^{-1}$ on the day of seeding. The surrounding borrow area was reclaimed at the same time in the same manner as the reclamation mix only plots but tall fescue (Festuca arundinacea) and red clover (Trifolium pratense) were added in some areas. Plots in which reclamation species were not desired were covered with plastic sheeting during the short time that seed was broadcast near the experimental areas.

Vegetation development was monitored and species presence in each plot was recorded at three to four week intervals. Cover after ten weeks was determined on June 20, 1986, using a pindrop method through a 20 point frame placed at random five times in each plot. Woody species were counted in each plot in late September, 1986. Vegetation development was recorded at various times using black and white or color photography. Nomenclature for all species follows Strausbaugh and Core (1978).

Data were analyzed on a Data General MV-7800 computer using the SYSTAT statistical program package of Wilkinson (1988). Data for numbers of native species, number of trees and number of shrubs were log transformed (n=log(n+1)) to reduce positive skewness, but data for other variables did not require this procedure. Comparisons of treatment means of vegetation variables were made using one-way analysis of variance (ANOVA) and Fisher's (protected) LSD (Steel and Torrie 1980). Independent variables tested by ANOVA were 1) use of topsoil,

2) use of reclamation mix, 3) use of mulch, and 4) method of topsoil use. Topsoil source was not considered as a variable. We decided a priori to make multiple comparisons of vegetation variables only among certain treatments as follows:

Absolute effectiveness of topsoil on vegetation variables:

T1 vs. T8,

Effects of reclamation mix on vegetation variables linked to topsoil:

T1 vs. T6 vs. T7

Effects of different methods of topsoil use on vegetation variables:

T1 vs. T2 vs. T3 vs. T5

Effects of mulch on vegetation variables:

T1 vs. T4

Significance level for all statistical tests was $\alpha = 0.05$. Analysis of variance tables are presented in Appendix 1.

Rainfall amounts were measured and recorded at the Northeastern Forest Experiment Station, Forestry Sciences Laboratory, about 3 km from the study site.

## Results and Discussion

This study revealed that small quantities of forest topsoils and their seed banks can be used successfully to introduce native species to borrow areas. However, the most obvious differences in plots were linked to use of the reclamation species mix and the hay mulch at ten weeks after establishment and afterwards. Where the reclamation mix was employed, an aggressive cover of planted grasses dominated the plots with lesser amounts of annual legumes. Where mulch was employed without the reclamation mix, the plots were dominated by wheat and timothy. Scattered native species were established in all topsoiled plots that did not have the seeded reclamation mix.

Forest topsoils significantly increased the number of native or naturalized plant species in the plots during the first and second growing seasons (one-way ANOVA of Treatment 1 and Treatment 8 data, $\alpha = 0.05$) (Table 1). Most of these species were few in number and occurred in only a few plots. In all topsoiled plots, 57 native species were found during the first growing season. The mean number of native or naturalized species per topsoiled plot without the reclamation mix was 11 with a range of 6 to 20. During the second year, a total of 87 species were documented with a mean of 19 and a range of 7 to 27 native or naturalized species per topsoiled plot. Some species found the first year were not present during the second growing season. Nonplanted native or naturalized species totaled 90 taxa from 34 families present in either or both years (Table 2). This number of established species is comparable to the 199 species found in a greenhouse study of eight Appalachian forest seed banks (Dobberpuhl 1981), 134 species from three Appalachian forest topsoils spread over mine spoils (Farmer et al. 1982), and 84 species in one forest topsoil seed bank spread on mine spoil in irrigated microplots (Wade 1986, 1989).

Native species composition of topsoiled plots was characteristic of the early seral stages of secondary forest succession, but species composition and frequency were not comparable to the standing forests (intermediate or late seral stages) of the topsoil donor sites.

Topsoiling resulted in a significantly greater increase in the mean numberof species in the plots (ANOVA) of Treatment 1 and Treatment 8 data, $\alpha = 0.05$ (Table 1). This may have been because additional dormant seed in the applied soils did not germinate until the second year. Other possibilities are that invaders were more successfully established after modification of environmental factors, or the presence of symbiotic soil microflora, especially mycorrhizal fungi. Farmer et al. (1982) observed that many native species from forest topsoil seed banks on newly exposed mine spoils had endomycorrhizal associates. Korean and Japanese lespedezas in the reclamation mix were observed to be growing more luxuriantly on and near the topsoil strips used in Treatment 5 than in the rest of the plots. This may have been because of native strains of Rhizobium spp. in the topsoils. Although the amounts of added forest soils were small, fertility factors might also have been influential in facilitating greater establishment of native species on topsoils.

Topsoil application significantly increased the number of shrubs but not the number of trees (ANOVA of Treatment 1 and Treatment 8 data, $\alpha = 0.05$) (Table 1). A total of seven tree species and seven shrub or woody vine species were present in all topsoiled plots (Table 2) with a mean total number of woody stems equivalent to 11,000 $ha^{-1}$. Of the trees, black locust (Robinia pseudoacacia) was the most numerous and fastest growing species and it was more frequent in the plots that received topsoil from the Virginia pine-red maple stand.

Native species from the seed bank did not provide a significant amount of ground cover in the plots after twn weeks in the first growing season (ANOVA of Treatment 1 and Treatment 8 data, $\alpha = 0.05$) (Table 1). Mean cover of native species in all topsoiled plots was 5 percent, but this was highly variable. The pin drop method of cover estimation is conservative since the fine pins as used in this study frequently drop through the canopies of some plants without actually touching leaves or stems. The amount of cover produced by seed bank species alone would not have met the standards for mined-land reclamation in the United States. Other forest topsoil seed bank studies in the eastern United States have produced acceptable cover during the first growing season under optimal growing conditions (Farmer et al. 1982; Wade 1986, 1989).

The reclamation mix species effectively eliminated wheat and timothy from the mulch and the native species in the topsoil (Table 3). ANOVA and Fisher's (protected) LSD of vegetation variable means from Treatment 1 (1 cm topsoil plus mulch), Treatment 6 (1 cm topsoil plus mulch and reclamation mix significantly ($\alpha = 0.05$) reduced the number of native species during the first two growing seasons, the number of trees, and the number of shrubs (Table 1). Cover of reclamation species was significantly higher than cover of wheat and timothy from the mulch. There were no significant differences in the amount of native species cover because of the low but highly variable numbers of native species in topsoiled plots. The increase in native species between the first and second years was significantly less in plots containing the reclamation mix. This suggests that the reclamation mix community could retard plant succession.

Table 1. Effects of various factors on vegetation variables in Cowbell Hollow plots. Within each factor group, means followed by the same letter are not significantly different, ANOVA followed by LSD where appropriate, $\alpha = 0.05$.

| Factor | Treat. | Year 1 Native Species | Year 2 Native Species | Incr. Native Species | Number Trees | Number Shrubs | % Cover | % Cover Native |
|---|---|---|---|---|---|---|---|---|
| Topsoil | T1 | 8.7a | 22.0a | 13.3a | 11.3a | 3.7a | 71.3a | 4.0a |
| | T8 | 0.7b | 5.0b | 4.3b | 0.7a | 0.0b | 76.0a | 0.3a |
| Rec Mix | T1 | 8.7a | 22.0a | 13.3a | 11.3a | 3.7a | 71.3b | 4.0a |
| | T6 | 0.3b | 3.3b | 3.0b | 0.0b | 0.3b | 100.0a | 0.0a |
| | T7 | 0.3b | 3.0b | 2.7b | 0.3b | 0.0b | 96.3a | 0.0a |
| Topsoil Method | T1 | 8.7a | 22.0a | 13.3a | 11.3a | 3.7a | 71.3a | 4.0a |
| | T2 | 8.3a | 14.7a | 6.3a | 2.0a | 0.3a | 72.3a | 7.0a |
| | T3 | 11.0a | 20.0a | 9.0a | 15.0a | 3.7a | 81.0a | 5.3a |
| | T5 | 8.7a | 18.0a | 9.3a | 5.3a | 2.7a | 66.0a | 5.0a |
| Mulch | T1 | 8.7a | 22.0a | 13.3a | 11.3a | 3.7a | 71.3a | 4.0a |
| | T4 | 16.0a | 19.0a | 3.0b | 16.0a | 4.3a | 9.0b | 4.7a |

Table 2. Frequency of species occurrence in plots of each treatment in 1986 and 1987.

| | T1 TS | | T2 TS til | | T3 TS 2x | | T4 TS-mul | | T5 TSstri | | T6 TS+RM | | T7 RMonly | | T8 No TSRM | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| SPECIES | 86 | 87 | 86 | 87 | 86 | 87 | 86 | 87 | 86 | 87 | 86 | 87 | 86 | 87 | 86 | 87 |
| Acalypha gracilens Gray | .3 | | .3 | | .3 | | .6 | | .3 | | | | | | | |
| Acalypha rhomboidea Raf. | | .7 | | | | .7 | | | | | | | | | | |
| Acer rubrum L. | | .7 | | .7 | | | | .3 | .7 | | | .3 | | .3 | | |
| Agrostis perennans (Walt.) Tuckerm. | | | | | | | | .3 | | | | | | | | |
| Aster pilosus Willd. | | | | | | .7 | | | | .3 | | | | | | |
| +Bromus japonicus Thunb. | | | | | | | | | | | | | | .3 | | .3 |
| +Cardamine hirsuta L. | | .3 | | | | | | | | | | | | | | |
| Carex digitalis Willd. | | .3 | | .3 | | | | | | .3 | | | | | | |
| +Cerastium viscosum L. | | .3 | | | | | | | | | | | | | | |
| Chenopodium album L. | | | | | | | | | | | | | .3 | | | |
| +Chrysanthemum leucanthemum L. | | | .3 | | | | | | | | | | | | | |
| +Cirsium arvense L. | | | | .7 | | .7 | | | .3 | | | | | | | |
| +Cirsium vulgare (Savi) Ten. | | | .3 | .3 | | | | | | | | | | | | |
| Cyperus esculentus L. | | | | | | | | | .3 | | | | | | | |
| +*Dactylis glomerata L. | | | | .3 | | .3 | | | | | | .3 | | .3 | | .3 |
| Danthonia spicata L. | | | | | | | | .3 | | | | | | | | |
| Desmodium paniculatum (L.) DC. | .3 | | | | | | | | | | | | | | | |
| +Digitaria ischaemum (Schreb.) Muhl. | | | .3 | | .3 | | | .3 | | | | | | | | |
| +Digitaria sanguinalis L. | | .3 | .3 | | | | | | | | | | | | | |
| Diodia teres Walt. | | | | | | | .3 | | | | | | | | | |
| Eragrostis spectabilis (Pursh) Steud. | | .3 | | | | | | | | | | | | | | |

Table 2. Frequency of species occurrence in plots of each treatment in 1986 and 1987. (continued)

| SPECIES | T1 TS | | T2 TS til | | T3 TS 2x | | T4 TS-mul | | T5 TSstri | | T6 TS+RM | | T7 RMonly | | T8 No TSRM | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 86 | 87 | 86 | 87 | 86 | 87 | 86 | 87 | 86 | 87 | 86 | 87 | 86 | 87 | 86 | 87 |
| Erechtites hieracifolia (L.) Raf. | .7 | | .3 | | .7 | | .7 | | | | | | | | | .3 |
| Erigeron annuus (L.) Pers. | | 1. | | 1. | | .3 | .3 | .7 | | 1. | | .7 | | .3 | .3 | .7 |
| Erigeron canadensis L. | | 1. | | 1. | | 1. | .3 | .7 | | 1. | | .3 | | .3 | | .7 |
| Erigeron philadelphicus L. | | .3 | | | | | .3 | | | .3 | | | | | | |
| Erigeron strigosus Muhl. | | .7 | | | | | | | | | | | | | | |
| +Erucastrum gallicum (Willd.) O. E. Shulz | | | .7 | | | | .7 | .3 | .3 | .3 | | | | | | |
| Eupatorium serotinum Michx. | .7 | .7 | | .3 | .7 | .7 | .7 | 1. | .7 | .7 | | | | | | |
| +*Festuca arundinacea Schreb. | .7 | 1. | 1. | .7 | 1. | 1. | .7 | 1. | 1. | 1. | 1. | 1. | 1. | 1. | .7 | 1. |
| Fragaria virginiana Duchesne. | | .3 | | | .3 | .3 | | | | | | | | | | |
| Geranium caroliniana L. | | | | .3 | | .3 | | .3 | | | | | | | | .3 |
| Gnaphalium purpureum L. | | .7 | | .7 | | .3 | | .3 | | | | | | | | |
| Hedeoma pulegioides (L.) Pers. | | | .3 | .3 | | | | | .3 | | | | | | | |
| Helianthus microcephalus T. & G. | .3 | | | | .3 | .3 | | | | | | | | | | |
| Hieracium venosum L. | | | | | | | .3 | | | | | | | | | |
| Houstonia purpurea L. | | .3 | | | | | | .3 | | | | | | | | |
| Hypericum gentianoides (L.) BSP. | | .3 | | | .3 | | | | | .3 | | .3 | | | | |
| Hypericum punctatum Lam. | | .3 | | | | | | .3 | | | | | | | | |
| +Ipomoea hederacea (L.) Jacq. | | | | | | | | | | .3 | | | | | | |
| Iva ciliata Willd. | | | | | | | | | | .3 | | | | | | |
| Juncus tenuis Willd. | .3 | 1. | | .3 | | 1. | .7 | .7 | .3 | | | | | | | |
| Lactuca canadensis L. | .3 | | | .3 | | .7 | | | | | | | | | | |
| +Lactuca serriola L. | | .7 | .3 | .7 | | .3 | | .3 | | .7 | | | | | | |
| Lespedeza intermedia (S. Wats.) Britton | | | | | | | .3 | | | | | | | | | |
| Lespedeza repens (L.) Bart. | | | | | | .3 | | | | | | | | | | |
| +*Lespedeza stipulacea Maxim. | 1. | 1. | 1. | .7 | 1. | 1. | 1. | 1. | 1. | 1. | 1. | 1. | 1. | 1. | 1. | 1. |
| +*Lespedeza striata (Thunb.) H. & A. | | .3 | | .7 | | | | .7 | .3 | | | | | | | |
| Lindernia dubia (L.) Pennell | | | | | | | | | .3 | | | | | | | |
| Liriodendron tulipifera L. | .3 | .3 | | | | .3 | .3 | .3 | | .3 | | | | | | |
| Lobelia inflata L. | | | | | | .7 | .7 | | | | | | | | | |
| +*Lolium multiflorum Lam. | .7 | 1. | .7 | 1. | 1. | 1. | .3 | 1. | 1. | 1. | 1. | 1. | 1. | 1. | 1. | 1. |
| +*Lolium perenne L. | 1. | .7 | .3 | .7 | 1. | 1. | .3 | 1. | 1. | 1. | 1. | 1. | 1. | 1. | 1. | 1. |
| Lysimachia quadrifolia L. | | | | | | .3 | | .3 | | | | | | | | |

Table 2. Frequency of species occurrence in plots of each treatment in 1986 and 1987. (continued)

| SPECIES | T1 TS | | T2 TS til | | T3 TS 2x | | T4 TS-mul | | T5 TSstri | | T6 TS+RM | | T7 RMonly | | T8 No TSRM | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 86 | 87 | 86 | 87 | 86 | 87 | 86 | 87 | 86 | 87 | 86 | 87 | 86 | 87 | 86 | 87 |
| +Medicago lupulina L. | .3 | .3 | .3 | .3 | | .3 | | | .7 | 1. | | .7 | | | | |
| +Microstegium vimineum (Trin.) A. Camus | | | | | | | .3 | | | | | | | | | |
| Panicum clandestinum L. | | | | | | | | .3 | | | | | | | | |
| Panicum commutatum Schultes | | | | | .7 | | | .3 | | | | | | | | |
| Panicum dichotomum L. | | | | | .3 | | .3 | | | | | | | | | |
| Panicum lanuginosum Ell. | | | | .3 | .3 | .7 | .3 | .7 | | | | | | | | |
| Panicum polyanthes Schultes | | | | | | | | .3 | | | | | | | | |
| Paronychia fastigiata (Raf.) Fernald | .3 | | .3 | | .3 | | | | | | | | | | | |
| +*Phleum pratense L. | .7 | .7 | .7 | .7 | .7 | .3 | .3 | .7 | .3 | .3 | .3 | | .3 | | .7 | .3 |
| Physalis heterophylla Nees. | | | | | | | | .3 | | | | | | | | |
| Phytolacca americana L. | .7 | .7 | .7 | | 1. | .7 | 1. | .7 | .7 | .7 | | | | | .3 | |
| Pinus virginiana Mill. | | .3 | | | | .3 | | | | | .3 | | | | | .3 |
| Plantago rugelii Dcne. | | | | .3 | | | .3 | .7 | | | | | | | | .3 |
| Platanus occidentalis L. | | | | | | | .3 | .3 | | | | | | | | |
| +Poa compressa L. | | | | | | | | .3 | | | | | | | | |
| Polygonatum biflorum (Walt.) Elliott | | .3 | | | | .3 | | | | .3 | | | | | | |
| Polygonum scandens L. | 1. | .7 | .3 | .3 | .7 | .3 | | | .7 | .7 | | | | .7 | .3 | .3 |
| Populus deltoides Marsh. | | | | | | | | .3 | | | | | | | | |
| Potentilla norvegica L. Ann. | | | | | | | | | | .3 | | | | | | |
| +Potentilla recta L. Per. | | | | .3 | | | | | | .3 | | | | | | |
| Potentilla simplex Michx. | | | | | | | | | | .3 | | | | | | |
| Rhus copallina L. | .3 | 1. | | | .7 | .3 | 1. | 1. | .7 | .7 | | | | | | |
| Rhus glabra L. | .7 | .7 | .3 | .3 | .3 | .3 | .7 | .3 | .3 | .3 | | | | | | |
| Robinia pseudoacacia L. | .7 | .7 | .7 | .7 | 1. | 1. | 1. | 1. | 1. | 1. | .3 | | | | | |
| Rubus allegheniensis T. C. Porter | | .3 | .3 | .3 | | .3 | .3 | .7 | .3 | .3 | | | | | | |
| +Rumex crispus L. | | | .3 | .3 | | | | | .3 | .3 | .3 | .3 | | | | |
| Sassafras albidum (Nutt.) Nees | | | | .3 | | .3 | .3 | .3 | .3 | .7 | .3 | .3 | | | | |
| Scutellaria leonardii Epling | | | .3 | .3 | | | .3 | .3 | | | | | | | | |
| +Sida spinosa L. | | | .7 | .3 | | | .3 | | | | | | | | | |
| Smilax glauca Walt. | | | | | | | | | | .3 | | .3 | | | | |
| Smilax bona-nox L. | | | | | | | | | | | | .3 | | | | |
| Solanum carolinense L. | | .3 | | .7 | .3 | .3 | | | | | | | | | | |
| Solanum nigrum L. | .3 | .3 | | | .3 | | | | .3 | .3 | | | | | | |
| Solidago canadensis L. | .3 | .3 | | | | | | .3 | | .3 | | | | | | |
| +Sonchus asper (L.) Hill. | | | | | | .3 | | .3 | | | | | | | | |
| +Sonchus oleraceus L. | | | .7 | | .3 | .3 | .3 | .3 | .3 | .3 | | | | | | |
| +Taraxacum officinale Weber. | | .3 | .3 | .7 | | .3 | .3 | .7 | | 1. | | | | .3 | | |
| +Trifolium campestre Schreb. | | | | .3 | | | | | | | | | | | | |

Table 2. Frequency of species occurrence in plots of each treatment in 1986 and 1987. (continued)

| SPECIES | T1 TS | | T2 TS til | | T3 TS 2x | | T4 TS-mul | | T5 TSstri | | T6 TS+RM | | T7 RMonly | | T8 No TSRM | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 86 | 87 | 86 | 87 | 86 | 87 | 86 | 87 | 86 | 87 | 86 | 87 | 86 | 87 | 86 | 87 |
| +*Trifolium pratense L. | .3 | | .3 | | .3 | | | | .3 | | | | | | | |
| +*Trifolium repens L. | | .3 | | | .3 | .7 | | .3 | .3 | .3 | | | | | | |
| +*Triticum aestivum L. | 1. | 1. | 1. | 1. | 1. | 1. | | .7 | 1. | 1. | 1. | 1. | .7 | .7 | 1. | .7 |
| +Verbascum blattaria L. | | | | | | | | | | .3 | | | | | | |
| +Verbascum thapsus L. | | | .3 | .3 | | .3 | .3 | | | | | | | | | |
| +Veronica arvensis L. | .3 | | | | | | | | | | | | | | | |
| Viola affinis Le Conte. | | | | | .3 | .7 | | | | | | | | | | |
| Vitis aestivalis Michx. | .3 | | | | | | .3 | .3 | | | | | | | | |

* Planted in reclamation mix plots and in the surrounding area.

+ Naturalized species.

Table 3. Mean number of native species present per plot in the first and second years, total number of trees, total number of shrubs, percent cover (at 10 weeks), and percent cover by native or naturalized species, planted grasses, planted legumes, and wheat. Standard deviations are in parentheses.

| | TREATMENT | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | TS 1 cm | TS 1 cm Tilled | TS 2 cm | TS no mulch | TS strip | TS + RecMix | RecMix only | no TS or RecMix |
| # native spp. year 1 | 8.7 (3.8) | 8.3 (2.1) | 11.0 (4.4) | 16.0 (6.1) | 8.7 (4.6) | 0.3 (0.6) | 0.3 (0.6) | 0.7 (0.6) |
| # native spp. year 2 | 22.0 (2.0) | 14.7 (7.1) | 20.0 (9.6) | 19.0 (8.5) | 18.0 (3.5) | 3.3 (2.5) | 3.0 (2.7) | 5.0 (1.7) |
| 2nd year ssp. increase | 13.3 (4.0) | 6.3 (5.0) | 9.0 (7.2) | 3.0 (3.5) | 9.3 (4.2) | 3.0 (3.0) | 2.7 (2.9) | 4.3 (2.1) |
| # trees | 11.3 (12.9) | 2.0 (1.0) | 15.0 (19.1) | 16.0 (19.2) | 5.3 (3.5) | 0.0 (0.0) | 0.3 (0.6) | 0.7 (0.6) |
| # shrubs | 3.7 (3.1) | 0.3 (0.6) | 3.7 (3.1) | 4.3 (3.5) | 2.7 (2.5) | 0.3 (0.6) | 0.0 (0.0) | 0.0 (0.0) |
| % cover | 71.3 (11.7) | 72.3 (25.7) | 81.0 (12.3) | 9.0 (9.8) | 66.0 (6.6) | 100.0 (0.0) | 96.3 (4.0) | 76.0 (4.6) |
| % cover native species | 4.0 (5.3) | 7.0 (3.0) | 5.3 (4.0) | 4.7 (2.3) | 5.0 (8.7) | 0.0 (0.0) | 0.0 (0.0) | 0.3 (0.6) |
| % cover planted grasses | 4.0 (6.9) | 0.7 (1.2) | 0.0 (0.0) | 0.0 (0.0) | 0.7 (1.2) | 100.0 (0.0) | 96.3 (4.0) | 0.0 (0.0) |
| % cover planted legumes | 0.3 (0.6) | 0.3 (0.6) | 0.7 (1.2) | 0.0 (0.0) | 0.7 (1.2) | 7.3 (5.8) | 6.0 (2.6) | 0.0 (0.0) |
| % cover wheat | 60.7 (12.2) | 68.3 (32.3) | 75.7 (19.0) | 5.0 (7.8) | 52.0 (9.5) | 7.3 (5.1) | 4.3 (5.9) | 75.7 (4.5) |

Comparisons of different methods of topsoil use: 1 cm, 1 cm tilled into the substrate, 2 cm, and a 1 cm equivalent piled in deeper strips, showed no significant differences among methods of topsoil use for any vegetation variables (ANOVA, $\alpha = 0.05$) (Table 1).

Use of mulch resulted in a significantly greater increase in native species and cover (ANOVA of Treatment 1 and Treatment 4 data, $\alpha = 0.05$) (Table 1). The cover difference was largely because of wheat and timothy from seed input with the mulch. Greater increase in native species in mulched plots probably occurred because mulch conserved soil moisture, lowered surface temperatures, provided a trapping effect on wind-blown seeds, and reduced erosion. Mulch did not significantly affect the number of native species in the first two years, cover in the first year, or the total number of trees or shrubs. These results suggest that with seed-free straw mulches a light seeding of wheat or timothy may be desirable to provide additional cover in the first growing season of native species introduced via seed banks.

Native species were established during a severe drought on an exposed site with shallow soils and without protective cover except the hay mulch. Total precipitation was 0.7 and 1.9 standard deviations below normal in 1986 and 1987, respectively. Water deficits during most months of the growing seasons were even greater (Table 4). While the wheat and timothy from the mulch provided some shading and amelioration of surface temperatures, they were also competitors for soil water.

In a study using a forest topsoil seed bank spread over two surface-mine spoils, the number and biomass of established native species was significantly reduced when a reclamation mix of Italian ryegrass (Lolium multiflorum), perennial ryegrass (L. perenne), tall fescue (Festuca arundinacea), weeping lovegrass (Eragrostis curvula), and sericea lespedeza (Lespedeza cuneata) was seeded into the forest topsoil, but the native species were not eliminated (Wade 1986, 1989). Ground cover developed fastest in the reclamation mix and slowest in the seed bank community, but there were no significant differences in ground cover 16 weeks after establishment. Doerr and Redente (1983) also observed that a mixture of seeded grasses repressed native forb biomass production in Colorado. The lower cover and populations of native species in the Cowbell Hollow borrow area may be attributed to the harsher environmental conditions, especially drought, low soil moisture, and possibly a reduced number of propagules in the smaller amount of forest topsoil used.

## Conclusions

Forest topsoil seed banks can be used to introduce a variety of native species under harsh environmental conditions to drastically disturbed lands such as borrow areas. Many of the species contained in forest topsoil seed banks are not available from seed suppliers. For the reclamationist interested in native species establishment, topsoil seed banks provide a low cost, easily available propagule source for the revegetation of disturbed sites for enhancement of wildlife or forest habitats.

The coemployment of common, aggressive herbaceous ground cover species can significantly reduce success of native species establishment from forest topsoils. The reclamation species mix effectively eliminated all species from the forest topsoil seed banks during the first year; 82 were present during the second year. Different methods of forest topsoil use did not significantly affect results. Effects of hay mulch were not significant except that presence of mulch significantly enhanced native species establishment during the second year.

## Literature Cited


Beauchamp, H., R. Lang, and M. May. 1975. Topsoil as a seed source for reseeding strip mine spoils. Wyo. Agr. Exp. Sta. Res. J. 90, Univ. Wyo., Laramie, WY.

Brenner, F. J. 1984. Restoration of natural ecosystems on surface coal mine lands in the northeastern United States. p. 211-255. In: T. N. Veziroglu (ed.) The Biosphere: Problems and Solutions. Elsevier Science Publishers B. V., Amsterdam.

Brooks, D. R. 1988. Rehabilitation following heavy mineral sands mining in Australia. p. 129-147. In: Environmental Workshop--1988 Proceedings, Volume I. Darwin, September 1988. Australian Mining Industry Council, Dickson ACT.

Conner, G., T. Boyer, and L. Dilgard. 1981. Climate of Berea Kentucky. Climatology of Kentucky Series. Kentucky Climate Center, Department of Geology and Geography, Western Kentucky University, Bowling Green.

Dobberpuhl, J. M. 1981. Seed banks of forest soils in East Tennessee. M. S. Thesis, The University of Tennessee, Knoxville.


Table 4. Growing season and yearly precipitation (mm) at Berea, Kentucky, for 1986, 1987, and the 1951-1980 mean and standard deviation (Conner et al. 1981).

| Year | Feb | Mar | Apr | May | Jun | Jul | Aug | Sep | Year Total |
|---|---|---|---|---|---|---|---|---|---|
| 1986 | 110 | 71 | 29 | 67 | 45 | 102 | 86 | 156 | 1043 |
| 1987 | 90 | 73 | 49 | 58 | 73 | 82 | 32 | 11 | 813 |
| mean | 85 | 122 | 115 | 106 | 109 | 122 | 102 | 95 | 1193 |
| s.d. | 56 | 56 | 61 | 48 | 48 | 55 | 61 | 60 | 201 |

DePuit, E. J. 1984. Potential topsoil strategies for enhancement of vegetation diversity on mined lands. Minerals and the Environment 6:115-120.

Doerr, T. B., and E. F. Redente. 1983. Seeded plant community changes on intensively disturbed soils as affected by cultural practices. Reclam. Reveg. Res. 2:13-24.

Farmer, R. E., Jr., M. Cunningham, and M. A. Barnhill. 1982. First-year development of plant communities originating from forest topsoils placed on southern Appalachian minesoils. J. Appl. Ecol. 19:282-294.

Gilbert, O. L., and P. Wathern. 1980. The creation of flower-rich swards on mineral workings. Reclamation Review 3:217-221.

Grime, J. P. 1981. The role of seed dormancy in vegetation dynamics. Ann. Appl. Ecol. 98: 555-558.

Howard, G. S., and M. J. Samuel. 1979. The value of fresh-stripped topsoil as a seed source of useful plants for surface mine revegetation. J. Range Manage. 32:76-77.

Iverson, L. R., and M. L. Wali. 1982. Buried, viable seeds and their relation to revegetation after surface mining. J. Range Manage. 35:648-652.

Newton, J. H., H. P. McDonald, D. G. Preston, A. J. Richardson, and R. P. Sims. 1973. Soil survey of Madison County, Kentucky. USDA Soil Conservation Service, Washington, DC.

Steele, R. G. D., and J. H. Torrie. 1980. Principles and Procedures of Statistics, 2nd Ed. McGraw-Hill Book Company, New York.

Strausbaugh, P. D., and E. L. Core. 1978. Flora of West Virginia, Second Ed. Seneca Books, Inc., Grantsville, West Virginia. 1079 p.

Tacey, W. H. 1979. Landscaping and revegetation practices used in rehabilitation after bauxite mining in Western Australia. Reclamation Review 2:123-132.

Tacey, W. H., and B. L. Glossop. 1980. Assessment of topsoil handling techniques for rehabilitation of sites mined for bauxite within the jarrah forest of Western Australia. J. of Appl. Ecol. 17:195-201.

Wade, G. L. 1986. Forest topsoil seed banks for introducing native species in eastern surface-mine reclamation. p. 155-164. In: New Horizons for Mined Land Reclamation: 1986 National Meeting of the American Society for Surface Mining and Reclamation; Jackson, MS: March 17-20; American Society for Surface Mining and Reclamation, Princeton, WV.

Wade, G. L. 1989. Grass competition and establishment of native species from forest soil seed banks. Landscape and Urban Planning 17:135-149.

Wathern, P., and O. L. Gilbert. 1978. Artificial diversification of grassland with native herbs. J. Environ. Manage. 7:29-42.

Wilkinson, L. 1988. SYSTAT: The System for Statistics. SYSTAT, Inc., Evanson, Illinois.

Appendix 1. Analysis of variance tables for effects of topsoil use, reclamation mix, method of topsoil use, and mulch.

| Source | SS | DF | MS | F | Sig. | Data |
|---|---|---|---|---|---|---|
| $\log_{10}$ number of species, 1st year | | | | | | |
| topsoil | 4.643 | 1 | 4.643 | 31.437 | ** | T1 T8 |
| error | 0.591 | 4 | 0.148 | | | |
| rec mix | 7.924 | 2 | 3.962 | 26.090 | ** | T1 T6 T7 |
| error | 0.911 | 6 | 0.152 | | | |
| method | 0.122 | 3 | 0.041 | 0.324 | ns | T1 T2 T3 T5 |
| error | 1.004 | 8 | 0.125 | | | |
| mulch | 0.470 | 1 | 0.470 | 3.054 | ns | T1 T4 |
| error | 0.615 | 4 | 0.154 | | | |
| $\log_{10}$ number of species, 2nd year | | | | | | |
| topsoil | 2.830 | 1 | 2.830 | 50.541 | ** | T1 T8 |
| error | 0.224 | 4 | 0.056 | | | |
| rec mix | 6.779 | 2 | 3.389 | 12.552 | ** | T1 T6 T7 |
| error | 1.620 | 6 | 0.270 | | | |
| method | 0.330 | 3 | 0.110 | 0.687 | ns | T1 T2 T3 T5 |
| error | 1.281 | 8 | 0.160 | | | |
| mulch | 0.065 | 1 | 0.065 | 0.549 | ns | T1 T4 |
| error | 0.473 | 4 | 0.118 | | | |

Appendix 1. Analysis of variance tables for effects of topsoil use, reclamation mix, method of topsoil use, and mulch. (continued)

| Source | SS | DF | MS | F | Sig. | Data |
|---|---|---|---|---|---|---|
| species increase | | | | | | |
| topsoil | 121.500 | 1 | 121.500 | 11.758 | * | T1 T8 |
| error | 41.333 | 4 | 10.333 | | | |
| rec mix | 220.667 | 2 | 110.333 | 9.832 | * | T1 T6 T7 |
| error | 67.333 | 6 | 11.222 | | | |
| method | 75.000 | 3 | 25.000 | 0.901 | ns | T1 T2 T3 T5 |
| error | 222.000 | 8 | 27.750 | | | |
| mulch | 160.167 | 1 | 160.167 | 11.306 | * | T1 T4 |
| error | 56.667 | 4 | 14.167 | | | |
| $\log_{10}$ number of trees | | | | | | |
| topsoil | 4.090 | 1 | 4.090 | 5.893 | ns | T1 T8 |
| error | 2.776 | 4 | 0.694 | | | |
| rec mix | 8.064 | 2 | 4.032 | 8.713 | * | T1 T6 T7 |
| error | 2.776 | 6 | 0.463 | | | |
| method | 2.622 | 3 | 0.874 | 1.108 | ns | T1 T2 T3 T5 |
| error | 6.308 | 8 | 0.789 | | | |
| mulch | 1.014 | 1 | 0.104 | 0.080 | ns | T1 T4 |
| error | 5.181 | 4 | 1.295 | | | |
| $\log_{10}$ number of shrubs | | | | | | |
| topsoil | 2.883 | 1 | 2.883 | 12.000 | * | T1 T8 |
| error | 0.961 | 4 | 0.240 | | | |
| recmix | 3.3310 | 2 | 1.655 | 7.750 | * | T1 T6 T7 |
| error | 1.281 | 6 | 0.214 | | | |
| method | 2.677 | 3 | 0.892 | 1.781 | ns | T1 T2 T3 T5 |
| error | 4.008 | 8 | 0.501 | | | |
| mulch | 0.019 | 1 | 0.019 | 0.037 | ns | T1 T4 |
| error | 2.110 | 4 | 0.528 | | | |
| cover (percent) | | | | | | |
| topsoil | 32.667 | 1 | 32.667 | 0.413 | ns | T1 T8 |
| error | 316.667 | 4 | 79.167 | | | |
| rec mix | 1460.007 | 2 | 730.003 | 14.252 | ** | T1 T6 T7 |
| error | 307.33 | 6 | 51.222 | | | |
| method | 347.333 | 3 | 115.778 | 0.468 | ns | T1 T2 T3 T5 |
| error | 1979.333 | 8 | 247.417 | | | |
| mulch | 5828.167 | 1 | 5828.167 | 49.743 | ** | T1 T4 |
| error | 468.667 | 4 | 117.167 | | | |
| native species cover (percent) | | | | | | |
| topsoil | 20.167 | 1 | 20.167 | 1.424 | ns | T1 T8 |
| error | 56.667 | 4 | 14.167 | | | |
| rec mix | 31.947 | 2 | 15.973 | 1.711 | ns | T1 T6 T7 |
| error | 56.00 | 6 | 9.333 | | | |
| method | 14.00 | 3 | 4.667 | 0.145 | ns | T1 T2 T3 T5 |
| error | 256.667 | 8 | 32.083 | | | |
| mulch | 0.667 | 1 | 0.667 | 0.040 | ns | T1 T4 |
| error | 66.667 | 4 | 16.667 | | | |

Recolonization of Vesicular-Arbuscular Mycorrhizae on a Reclaimed Strip Mine in North Dakota. John Reed Cockrell, Graduate Student, Dept. of Animal and Range Sciences, North Dakota State University, Fargo, ND 58105.

Vesicular-arbuscular mycorrhizae (VAM) are an integral component of North Dakota grassland ecosystems. Stockpiling topsoil in the process of mining lignite coal can drastically reduce the number of viable VAM propagules; reestablishment of VAM is therefore an important measure of reclamation success. Numerous soil parameters as well as topography and age of the reclaimed soil may all influence VAM recolonization. This study investigates the relationship of VAM infection to numerous soil properties, basal cover and time since reclamation. Thirteen transects of ten points each have been laid out on the Glenharold Mine near Stanton, North Dakota. Four transects have been placed on undisturbed sites, four on a site reclaimed in 1979 and five on a site reclaimed in 1984. Topographic data have been collected at each transect point, along with percent basal cover of plant species and numerous soil properties: pH; electrical conductivity; saturation percentage; calcium magnesium, sodium and sodic adsorption ratio (SAR); and sand, silt and clay percentages. Root samples of Agropyron smithii (a cool season grass) and Bouteloua gracilis and B. curtipendula (warm season grasses) were excavated at each transect point, and stained, and analyzed under a microscope to determine percent mycorrhizal infection. Preliminary data indicate that VAM infection is significantly higher on the reclaimed sites than it is on undisturbed sites, for both cool season and warm season species. Undisturbed sites exhibit VAM infection levels of 7.60 ± 0.52% for Agropyron and 6.58 ± 0.50% for Bouteloua, while the site reclaimed in 1979 exhibits 24.43 ± 0.88% infection for Agropyron and 17.43 ± 1.20% for Bouteloua. The site reclaimed in 1984 exhibits 15.57 ± 0.66% VAM infection for Agropyron. There appears to be little, if any correlation between infection level and basal cover or soil properties. No trends in infection level due to topography have been noted.

This page intentionally left blank.

# Importance of Indigenious VAM Fungi for the Reclamation of Coal Refuse Piles[1]

by

D. C. Dove, W. L. Daniels and D. J. Parrish[2]

Abstract. During the initial phases of vegetation establishment on drastically disturbed soils, vesicular-arbuscular mycorrhiza (VAM) can enhance plant establishment and growth. A similar beneficial response would be very advantageous in efforts to reclaim coal refuse piles. Coal refuse, a by-product of eastern coal cleaning plants, presents a particularly difficult problem for reclamationists. In preliminary field trials conducted in Mercer Co. WV, results indicated that survival of containerized white clover (Trifolium repens) was enhanced after outplanting when VAM colonized plant roots. Roots extending into the coal refuse were found to be devoid of VAM. A subsequent growth chamber study was undertaken to evaluate various white clover-VAM combinations to increased concentrations of coal refuse in plant growth media. Five combinations of coal refuse and soil were used. Treatments included two VAM species and a non-VAM control. After 60 days of growth a significant increase in root and shoot dry weights was observed in VAM colonized seedlings. As the percentage of coal refuse in the growth media increased, those plants colonized by an indigenous VAM species had improved S:R ratios, leaf water relations and leaf areas. Tissue concentration of P, K, Ca and Mg were higher for VAM colonized seedling, with indigenous VAM infections providing increases as the percentage of coal refuse in the growth media increased. These data suggest that VAM do enhance plant growth and establishment in coal refuse; however ecotypic variations in VAM must be studied further.

Additional Key Words: vesicular-arbuscular, mycorrhiza, coal refuse, reclamation.

[1] Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, WV. April 23-26, 1990.
2 D.C. Dove, W.L. Daniels and D.J. Parrish are Research Assistant, Assistant Professor and Assistant Professor, respectively in the Dept. of Crop and Soil Environmental Sciences, Virginia Polytechnic Institute and State University, Blacksburg, VA.

## Introduction

Coal refuse piles present a unique environment for reclamation efforts. Modern coal cleaning plants produce, as a by-product, thousands of tons of coal refuse annually. This material consists primarily of rock inclusions and other contaminating, often phytotoxic materials that are removed from coal before it is burned. Fragments ranging from 20 cm in diameter to sand-sized and smaller are separated from the coal using various sedimentation processes. After separation, the refuse material is transported to a permanent disposal site. As a result of the inherent mineralogical variability in coal seams and the different compounds used in the cleaning processes, coal refuse

exhibits a wide range of chemical characteristics. Such materials often represent an environmental hazard because of toxic substances that can develop in and leach from them. Recently enacted laws (Surface Mining Control Act of 1977) require that all permanent refuse disposal sites be vegetated with a permanent self-sustaining plant cover.

The inherent low fertility and recent deposition of coal refuse presents a set of growth-limiting factors similar to those encountered by pioneer plant species. Advancing on this premise, several studies (Schramm 1966; Daft and Hacskaylo 1976; Khan 1978) have been conducted to determine possible adaptations by successful colonizing species to overcome the stresses presented.

Schramm (1966) concluded that plants successfully colonizing bare, nutrient-deficient coal wastes are generally nitrogen-fixing, mycorrhizal, or both. A majority of plants growing on coal refuse piles in Scotland (Daft and Hacskylo 1974) and Australia (Khan 1978) was found to be infected by mycorrhiza-forming fungi.

The benefit to the "host plant" in a mycorrhizal association is widely believed to be due to improved phosphorus nutrition of the host (Daft and Hacskylo 1976; Lambert and Cole 1980). The possible alteration of other physiological processes that would enhance colonization of stressful sites has also been suggested (Schramm, 1966; Khan 1978).

A major purpose of the studies presented in this work was to examine the possible links between successful plant colonization of coal-refuse piles and the presence of vesicular-arbuscular mycorrhizae (VAM). In many situations, the presence of VAM does not confer an advantage to the host plant (Hetrick et al. 1984); i.e., VAM infection cannot always be viewed as beneficial. While the VAM-host-plant interaction may be highly integrated both structurally and physiologically, better growth of the host is not automatically a consequence of the relationship. Host-plant responses vary widely when conditions of nutrition, soil, and other environmental factors are changed. Isolates (ecotypes) of the same VAM species collected at locations with differing rainfall patterns have been shown to have substantially different effects on host-plant water relations (Stahl and Smith 1986). Especially at the species level, the presumption that differing host-plant responses would occur seems valid.

The ability of a VAM association to enhance host plant growth has been called symbiotic efficiency (Smith and Gianinazzi-Pearson 1988). Several growth parameters have been used to quantify the symbiotic efficiency of a VAM isolate with a particular host species. In situations of environmental stress good indicators of symbiotic efficiency are relative growth rate and root:shoot ratio (Smith and Gianinazzi-Pearson 1988). Under nutritional stress, possible determinants of symbiotic efficiency of a VAM isolate or species are host nutrient uptake and deployment as measured by tissue analysis (Smith and Gianinazzi-Pearson 1988).

A central (implicit) hypothesis in these studies is that VAM species isolated from environments most closely matching the experimental conditions of a study, will exhibit a higher degree of symbiotic efficiency. Commonly, VAM isolates are selected for their ability to promote growth under non-stressful conditions. By selecting ecotypes that have evolved on or near refuse piles the possibility of finding a VAM-producing fungus that will enhance its hosts tolerance of these stresses is much improved.

## Experimental Methods

### Seedling Establishment

White clover (Trifolium repens) seedlings were established in plastic cone-tainers that had been filled with various mixtures of soil and coal refuse. The soil, a Groseclose silt loam, was mixed with coal refuse to achieve 0, 25, 50, 75 or 100 % coal refuse content. Nutrient status and pH for the five soil:refuse combinations is shown in Tables 1 and 2. Prior to mixing, soil was fumigated with methyl bromide and thoroughly aerated. Coal refuse samples taken from several seams in the Pocahontas formation of West Virginia were used after being autoclaved at 180°C for 30 minutes, allowed to cool for 18 hour and re-autoclaved. In preparation for this experiment, the coal refuse was treated with lime at 2 Mg/ha 18 months prior to experimentation.

At planting, soil preinoculated with either Glomus etunicatium (Ge) or Glomus sp. (Gs) or filtrate from sievings of the two spore cultures (none) was added to each cone-tainer. After seedling emergence, but prior to the appearance of the first trifoliolate leaf, plants were thinned to one per pot.

Table 1. Chemical characteristics of soil: coal refuse mixtures.

| Growth Media | pH | E.C. |
|---|---|---|
| % Refuse | | dS/cm |
| 0 | 5.6 | 0.22 |
| 25 | 6.8 | 0.23 |
| 50 | 7.3 | 0.34 |
| 75 | 7.4 | 0.42 |
| 100 | 7.4 | 0.69 |

Table 2. Plant available nutrient content of soil:coal refuse mixtures.

| Growth Media | NH4OAC Extractable P | K | Ca |
|---|---|---|---|
| % Refuse | ------- mg/kg ------- | | |
| 0 | 3.8 | 47.4 | 216.0 |
| 25 | 4.8 | 46.8 | 412.0 |
| 50 | 7.6 | 57.4 | 648.0 |
| 75 | 7.4 | 64.4 | 777.6 |
| 100 | 9.4 | 68.2 | 871.2 |

Plants were grown in a growth chamber under a 14 h daylength, temperatures of 25° C day /18oC night and relative humidities of 70 to 80 %. All plants were irrigated with 5 ml of 50% Hoaglands nutrient solution minus P - containing salts (Hoagland and Arnon, 1950) daily. At biweekly intervals, all mycorrhizal treatments were fertilized with 10 ml of 50% Hoaglands nutrient solution with P salts added, non-mycorhizal treatments received 10 ml weekly.

Water Relations Studies

When the plants were 55 days old, all pots were brought to field capacity and wrapped to prevent evaporative water loss, and water was withheld for five days. Relative water content (RWC)(Matin et al. 1989) was determined for an excised leaf at day five. Determination of RWC was performed using the formula:

$$RWC = [(FW - DW)/(TW - DW)]* 100$$

where FW = fresh weight
DW = dry weight
TW = turgid weight ( after rehydration in distilled water).

Post-Harvest Measurements

Plants were removed from all pots 60 days after planting, and the soil:refuse materials were washed from roots. Root length was estimated using the line intersect method (Tennant 1975). Roots and shoots were dried separately at 70 C for 18 hours and weighed. Rehydrated roots were stained (Kormanik et al. 1980) and the VAM colonization assessed by a modified line-intersect method (Biermann and Lindemann 1981). Concentrations of P in dried leaves was determined using inductive coupled plasma spectroscopy (Donahue 1986). Statistical analysis was conducted using a split-plot design. Data compilation was performed using GLM (General Linear Models) procedure in SAS (SAS Institute, Inc. 1985).

Results

When compared to the non-VAM control, total biomass production of white clover (Tables 3 and 4) was enhanced by the presence of VAM at all soil:refuse combinations. Top growth was particularly enhanced by VAM. The isolate Ge, a ubiquitous mycorrhizal species, produced higher shoot dry weights at 0 and 25% mixtures of coal refuse. Shoot biomass was greater with Gs when coal refuse contents were greater than 50%. Root biomass production was enhanced with VAM infection at higher coal refuse contents. In general, root biomass was greater in Gs infected plants where coal refuse was added to the growth medium.

Table 3. Root production as influenced by various soil refuse mixtures and two mycorrhiza (Glomus etunicatum (Ge) and Glomus sp. (Gs)).

| Growth Media | VAM Symbiont Ge | Gs | None | LSD (0.05) |
|---|---|---|---|---|
| % Refuse | ----- dry wt.(g) ----- | | | |
| 0 | 0.92 | 0.59 | 0.45 | 0.91 |
| 25 | 1.09 | 1.50 | 0.61 | 1.34 |
| 50 | 0.75 | 2.38 | 0.47 | 0.48 |
| 75 | 0.60 | 0.74 | 0.55 | 0.48 |
| 100 | 0.46 | 0.62 | 0.44 | 0.39 |
| LSD (0.05) | 0.96 | 0.96 | 0.36 | |

With increasing content of coal refuse in the growth medium, VAM-infected plants responded differently in the partitioning of dry matter (Table 5). A decrease in root production by Gs-infected plants at 75

Table 4. Shoot production as influenced by various soil refuse mixtures and two mycorrhiza (Glomus etunicatum (Ge) and Glomus sp. (Gs)).

| Growth Media | VAM Symbiont Ge | Gs | None | LSD (0.05) |
|---|---|---|---|---|
| % Refuse | ----- dry wt.(g) ----- | | | |
| 0 | 1.49 | 3.38 | 0.85 | 2.92 |
| 25 | 1.93 | 4.96 | 1.06 | 2.89 |
| 50 | 3.52 | 3.39 | 0.85 | 1.91 |
| 75 | 3.97 | 2.68 | 0.84 | 2.64 |
| 100 | 3.01 | 1.87 | 0.50 | 1.84 |
| LSD (0.05) | 2.52 | 3.19 | 0.64 | |

and 100% refuse shifted the shoot:root

ratio sharply to favor shoot production. While the partitioning ratio for Ge-infected plants was unchanged with refuse additions, total biomass declined.

Table 5. Shoot:root ratio based on the dry weight of plants grown in various soil refuse mixtures and two mycorrhiza (Glomus etunicatum (Ge) and Glomus sp. (Gs)).

| Growth Media | VAM Symbiont Ge | Gs | None | LSD (0.05) |
|---|---|---|---|---|
| % Refuse | ----- S:R ratio ----- | | | |
| 0 | 2.55 | 5.15 | 3.91 | 5.18 |
| 25 | 2.09 | 4.96 | 2.03 | 1.62 |
| 50 | 1.50 | 4.51 | 1.64 | 2.01 |
| 75 | 5.69 | 4.59 | 1.51 | 1.53 |
| 100 | 4.41 | 2.89 | 1.89 | 2.57 |
| LSD (0.05) | 1.81 | 3.55 | 2.73 | |

The RWC for all treatment combinations (Table 6) did not follow a consistant trend. VAM-infected plants had moderately better RWC as the refuse content increased to 50, 75 and 100 %. While several of the non-VAM treatments exhibited RWC below 90 % none were oberserved to have lost turgidity.

Vesicular-arbuscular mycorrhiza increased white clover tissue P concentrations at all levels of refuse addition (Table 7). Increased concentrations of coal refuse in the media did not affect the P concentration in the plant tissue.

Table 6. Relative water content of plants grown in various soil refuse mixtures and inoculated with two mycorrhiza (Glomus etunicatum (Ge) and Glomus sp. (Gs)).

| Growth Media | VAM Symbiont Ge | Gs | None | LSD (0.05) |
|---|---|---|---|---|
| % Refuse | ------ % Water ------- | | | |
| 0 | 92.5 | 87.5 | 94.2 | 6.9 |
| 25 | 89.9 | 91.8 | 93.1 | 4.1 |
| 50 | 92.7 | 95.4 | 89.9 | 2.9 |
| 75 | 93.4 | 95.0 | 84.8 | 3.9 |
| 100 | 89.9 | 92.3 | 88.4 | 6.3 |
| LSD (0.05) | 4.4 | 4.0 | 5.9 | |

Table 7. Tissue P concentrations of plants grown in various soil refuse mixtures and inoculated with two mycorrhiza (Glomus etunicatum (Ge) and Glomus sp. (Gs)).

| Growth Media | VAM Symbiont Ge | Gs | None |
|---|---|---|---|
| % Refuse | --------- % ---------- | | |
| 0 | 0.18 | 0.21 | 0.12 |
| 25 | 0.25 | 0.17 | 0.09 |
| 50 | 0.25 | 0.17 | 0.10 |
| 75 | 0.22 | 0.16 | 0.08 |
| 100 | 0.19 | 0.17 | 0.09 |
| Mean | 0.21 | 0.16 | 0.09 |
| LSD (0.05) | 0.04 | | |

## Discussion

Vesicular-arbuscular mycorrhiza enhanced white clover growth in coal refuse. Mycorrhizal plants are often observed to have dramatic increases in dry matter production (Smith and Gianinazzi-Pearson 1988). In this experiment interesting trends in dry-matter production and allocation developed as the content of coal refuse increased in the growth medium. As content increased to 50%, the amount and proportion of dry matter devoted to roots increased sharply in Gs infected plants, while little or no increase was observed for the Ge-infected plants. At the same time, shoot biomass increased in the Ge-infected plants from 0 to 25% coal refuse and declined with additional refuse inputs. Conversely, as refuse concentrations increased Gs-infected plants produced more shoot biomass. Infection with the Gs isolate appeared to enable the host plant to allocate more resources to shoot production. Increased shoot production with reduced root production indicates an increase in root efficiency.

Dry matter allocation is best illustrated as root:shoot ratios. Across all coal refuse combinations Ge infected plants maintained a stable S:R ratio. An overall reduction in root and shoot dry matter production was responsible for this stability. Root:shoot ratios for Gs-infected plants shifted sharply to favor shoot production at high concentrations of coal refuse, but were similar to non-VAM plants at lower concentrations (0, 25 and 50%). At some point beyond 50% refuse content Gs-infected plant roots became more

successful in maintaining sufficient metabolic processes to support increased shoot production.

Both VAM producing species were able to support higher P and plant water status at all soil:refuse combinations. Increases in P uptake have been linked to several parameters involved in the maintenance of good plant water relations (Smith and Gianinazzi-Pearson 1988). Plants infected with the Gs isolate had a generally higher RWC the higher refuse concentrations with a reduction in dry matter allocation to roots. While the leaf RWC was adequate to maintain turgor in the non-VAM plants, growth may have been limited by reductions in RWC as the concentration increased.

While the interactions between the Gs-isolate and higher coal refuse concentration are quite striking in these experiments, changes in the physical and chemical characteristics of the growth media may account for some of the observed results. As the concentration of refuse increased, both the pH and the level of soluble salts increased. These changes are a probable result of the addition of $CaCO_3$ (lime) prior to this study. Changes in the texture and bulk density of the growth media also occurred as the more coarse textured coal refuse was added to the finer textured soil.

## Summary

VAM infection enhanced white clover growth in coal refuse. Two VAM isolates produced different dry matter allocation patterns. An indigenous VAM isolate maintained a higher P and water status. Improvements in root uptake efficiency are likely responsible for better growth of white clover in coal refuse. Selection of a VAM isolate from a stressful site may have merit for use in improving plant establishment on coal refuse piles.

## Literature Cited

Biermann, Brenda and R.G. Lindermann. 1981. Quantifying vesicular-arbuscular mycorrhizae: A proposed method toward standardization. New Phytol. 87:63-67.

Daft, M. S. and E. Hacskaylo. 1976. Vesicular-arbuscular mycorrhiza in the antracite and bituminous coal wastes of Pennsylvania. J. Appl. Ecol. 13:523-531.

Daft. M. S. and T. H. Nicholson. 1974. Arbuscular mycorrhizas in plants colonizing coal wastes in Scotland. New Phytol. 73:1129-1135.

Donohue, S. J. and J. B. Fredericks. 1984. Laboratory Procedures: soil and plant testing. VA Extension Pub. no. 452-881.

Hetrick, B. A. D., J. A. Hetrick and J. Bloom. 1984. Interaction of mycorrhizal infection, phosphorus level, and soil moisture stress in growth of field corn. Can. J. Bot. 62:639-645.

Khan, A. G. 1978. Vesicular-arbuscular mycorrhizas in plants colonizing black wastes from bituminous coal mining in the Illawana Region of New South Wales. New Phytol. 81:25-32.

Kormanik, P. P., W. C. Bryan and R. Schultz. 1980. Procedures and equipment for staining large numbers of plant root samples for endomycorrhizal assay. Can. J. of Micro. 26:536-538.

Lambert, D. H., H. Cole and D. E. Baker. 1980. Adaptation of vesicular-arbuscular mycorrhizae to edaphic factors. 85:513-520.

Matin, M. A., J. H. Brown and H. Ferguson. 1989. Leaf water potential, relative water content and diffusive resistance as screening techniques for drought resistance in barley. Agron. J. 81:100-108.

Nelson, G. E. and G. R. Safir. 1982. Increased drought resistance of mycorrhizal onion plants caused by improved phosphorus nutrition. Planta 154:407-413.

SAS Institute Inc. 1988. SAS Circle, Box 8000, Cary, N. C. 27512

Schramm, J. R. 1966. Plant colonization on black wastes from anthracite mining in Pennsylvania. Trans. Am. Phil. Soc. 47:331-345.

Smith, S. E. and V. Gianinazzi-Pearson. 1988. Physiological interactions between symbionts in vesicular-arbuscular mycorrhizal plants. Ann. Rev. Plant Physiol. and Mol. Biol. 39:221-244.

Stahl, P. D. and W. K. Smith. 1984. Effects of different geographic isolates of Glomus sp. on the water relations of Andropyron smithii. Mycologia 76:261-267.

Tennant, D. 1975. A test of a modified line intersect method of estimating root length. J of Ecol. 63:995-1001

This page intentionally left blank.

Retardation of Pyrite Oxidation by Massive Fly Ash Application to Mine Spoils. D. K. Bhumbla, R. N. Singh, and R. F. Keefer, West Virginia University.

Field experiments were conducted to investigate the effect of massive application of fly ash with respect to retardation of pyrite oxidation and revegetation of pyritic mine spoil. Treatments consisted of 600 and 1200 Mg $ha^{-1}$ for each of three fly ash sources-- Albright, Fort Martin, and Harrison-- and a control of ground agricultural limestone placed in a randomized complete block design with four replications. The fly ashes varied in chemical composition and acid neutralization capacity (high for Fort Martin and Harrison fly ashes, but low for Albright fly ash). All plots received complete fertilizer and hay (to supply organic matter) which were disced in prior to seeding with a grass-legume mixture. A hay mulch was blown onto the surface of the whole area. Growth and chemical composition of vegetation were monitored. Crop growth and composition were related to composition and chemical changes of the growth media due to pyrite oxidation. The best crop establishment and growth during the first growing season were on plots receiving Albright fly ash in spite of no alkaline recharge to the subsurface spoil. However, application of this ash was not sufficient to significantly retard pyrite oxidation at the lower depths and resulted in decline in vegetative growth during the second year. Soil analyses with depth showed that alkaline recharge from Fort Martin and Harrison fly ashes was responsible for increases in subsurface spoil pH and retardation of pyrite oxidation.

ADDITIONAL KEY WORDS: Reclamation, Revegetation, Industrial Wastes, Industrial By-Products, Soluble Salts.

This page intentionally left blank.

# Abandoned Mine Land - General

This page intentionally left blank.

# SIMILITUDE MODEL EXPERIMENTS TO DETECT MINE CAVITIES[1]

F. Ziaie, S. S. Peng, and P. M. Lin[2]

Abstract. A resistivity technique for cavity location has been developed. This technique is a combination of the Bristow arrangement and line electrode method. In this technique three line electrodes are used so that the sinkhole electrode is placed far from the other two electrodes. When one of the two electrodes and the sinkhole electrode are activated, several resistivity profiles parallel and perpendicular to the line electrode are measured for different electrode activated. Subsurface cavities cause resistivity anomalies if they are crossed by the resistivity profiles. The anomalies are interpreted and used to locate the sources of the anomalies (cavities). A tank model and a similitude model are developed to verify the effectiveness of this method for cavity detection in the saline medium and in the actual materials. The results of the experiment indicate that the location and the dimensions of the cavities can be estimated successfully.

Additional Key Words: electrical resistivity, monopole, mine cavity, tank model, similitude model.

## Introduction

Abandoned mine cavities, especially those that are shallow, are an important source of surface instability. Therefore, such cavities should be detected and stabilized to protect the surface establishments. The earth resistivity technique is relatively inexpensive and has shown a high degree of success in locating subsurface cavities. One of the methods used for locating cavities is the tri-potential method of earth resistivity prospecting developed by Carpenter and Habberjam (1955). Later, Habberjam (1969) conducted an extensive study by analyzing theoretical consideration of this method for locating the cavities by using a brine tank modeling experiment. The monopole resistivity technique originally discussed by Logn (1954), and applied to cavity location by Bristow (1966), was eventually modified by Bates (1973). This technique was employed by the U.S. Bureau of Mines to locate abandoned mines and delineate the boundaries of old workings (Burdick et al. 1986). The line electrode resistivity technique which was modified using the Bristow resistivity arrangement was introduced by Ziaie et al. (1989). This technique was used in a brine tank model experiment in order to test its effectiveness in cavity location.

## Discussion of the new technique.

The proposed line electrode resistivity technique is analogous to the Bristow monopole configuration. In this method line electrodes are used instead of point electrodes. The point potential electrodes are used within or outside of the two active line electrodes to measure the potential gradient variation generated by the line electrodes. This potential variation is related to the subsurface structure. The major difference between this technique and the Bristow array is that the distribution of the potential in the subsurface delineates revolution of the half cylindrical surfaces rather than sperical surfaces. The axis of these surfaces are correlated with the closest line electrode when the two active electrodes are sufficiently far from each other (Fig. 1). If the subsurface structure does not vary laterally, then the intersections between the cylindrical surfaces and the earth surfaces are parallel equipotential lines. This equipotential gradient is montiored and is used to calculate the resistivity of the subsurface material. In the case of locally lateral variation of the subsurface material (such as the existence of mine cavities) the equipotential line will be distorted and the anomaly due to lateral variation of the subsurface can be interpreted to locate the source of the anomaly. The major advantage of this technique is that (1) the resultant anomaly for different profiles parallel and perpendicular to the line

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]F. Ziaie, S. S. Peng, and P. M. Lin are Graduate, Charles T. Holland Professor and Chairman, and Research Associate, respectively, at Department of Mining Engineering, College of Mineral and Energy Resources, West Virginia University, Morgantown, WV 26506.

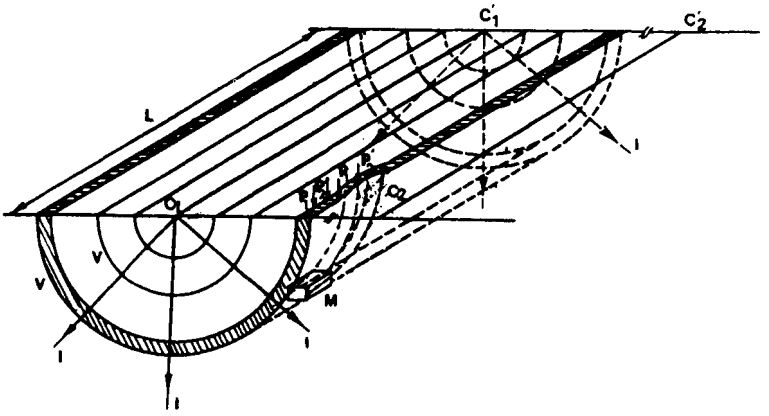



Fig. 1 Distortion of equipotential lines due to crossing of the cylindrical surfaces through a cavity. $C_1C'_1$ and $C_2C'_2$ are line electrodes and $P_1P'_1$, $P_2P'_2$, $P_3P'_3$... are potential electrodes, M is a mine cavity.

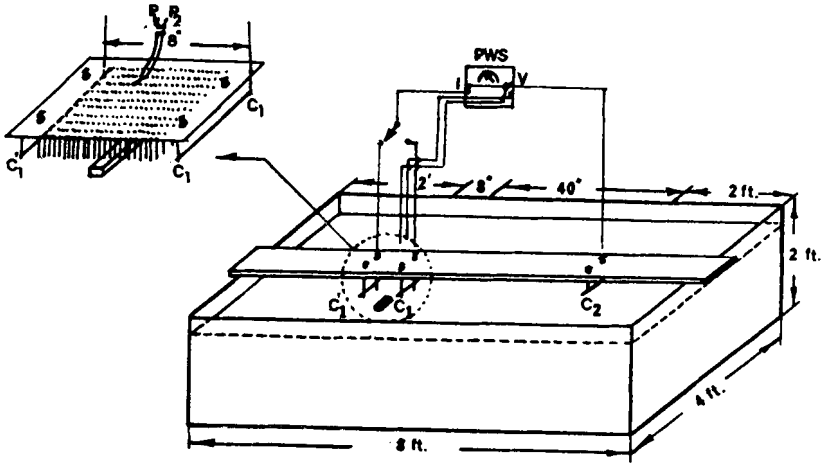



Fig. 2 Tank model.

electrode can be monitored simultaneously without the transmission of the line electrode, (2) the detected anomalies can be used to evaluate the cross sectional dimensions of the cavity for the profiles parallel and perpendicular to the line electrodes, and (3) the resolution is appear to be higher than Bristow technique. Therefore, the extension of the cavity in those profiles perpendicular to the line electrode can be measured. The mathematical relation of the potential distribution and subsurface apparent resistivity is available elsewhere (Telford et al. 1976) i.e.

$$\rho_a = \frac{T\pi(L^2-x^2)}{4\ dL} \left(\frac{\Delta v}{I}\right) \qquad (1)$$

where L is the half spacing of the two line electrodes (ft.) and x is the distance of midpoint of the two line electrodes from the midpoint of potential electrodes (ft.), d is the half spacing of two potential electrodes (ft.), I is the electrical current delivered to the earth (amps), V is the measured voltage (volts), T is the length of the line electrode (ft.), and $\rho_a$ is the apparent resistivity (ohm-ft).

Plotting of the resistivity anomalies and their interpretation. In this resistivity survey, the measured resistivity is associated with the stations located between the two potential electrodes. Thus for $C_1C_2$ electrodes activation the resistivity of all the stations are plotted versus their distances from the nearby activated line electrode. Since the same profile is used to measure the resistivity related to the other two activated line electrodes ($C'_1C_2$), the resistivity values of the same stations are plotted against their distances form the new activated current line electrodes repeatedly. The two profiles are superimposed on each other to obtain a combined profile for interpretation purpose. The potential electrodes spacing used for all the experiments was 0.25 inches and the minimum distance of the potential electrode from the current line electrode was 1.25 inches. The total length of each profile was 8 inches.

According to the resistivity response analysis of this type of experiments, 2/3 of the maximum peak anomaly response value associated to the cavity is used for the interpretation purpose. Therefore, a horizontal line from 2/3 of the maximum amplitude is drawn to intersect the anomaly curve at two adjacent points. The trajectories of the intersected points on the $C_1C'_1$ line are used to draw the appropriate arcs. The arcs are centered at the related activated line electrode. Their radii are the distances of the mentioned trajectory points from the nearby activated current line electrode. The intersections of the two pairs of arcs involved in any activated current line electrodes delineate the boundary of the detected cavity. The circumscribed square or circle within the intersected arcs shows the cavity location and dimension (refer to Figs. 4 and 5).

Room and pillar simulation in the tank model experiment. A brine tank, 8 feet long, 4 feet wide and 2 feet high, was designed (Fig. 2). Three current line 7 inch long electrodes were set up as shown in Figure 2. The survey area, 5 inches wide and 8 inches long, was located between the two current line electrodes, $C'_1$ and $C_1$. The potential electrodes with a spacing of 0.25 inches were moved within the survey area perpendicular to the line electrode. Four simulated square rooms 1 inch wide and 0.75 inches high with spacing of 1 inch were used and placed 1 inch deep in the survey area (Fig. 3). Along the length of the survey area, 22 profiles labeled A, B, C...V were spaced 0.25 inches apart for measuring the resistivity profiles perpendicular to the line electrodes. Along the width of the survey area 10 profiles labeled 1, 2, 3...10 with spacing of 0.50 inches were also chosen in order to enable measurement of another set of resistivity profiles (Fig. 3). The current line electrodes $C'_1$ and $C_2$ were activated and data were acquired in sequential order for profiles 1-1 to 10-10. Similarly, $C_1$ and $C_2$ line electrodes were activated and profiles 1-1 to 10-10 again were obtained.

Figure 4 shows the results of the calculation and plotting of the resistivity of the cross-section profile 3-3 which traversed the rooms. The broken line indicated the location of the detected rooms and pillars. The solid line shows the actual location of the rooms and pillars. Figure 5 shows the cross-section profile 7-7 and the result of the interpretation. The actual locations of the rooms are in good agreement with the location of those detected. Other cross-sections parallel to the line electrode (Figs. 6 and 7) were used to estimate the extension of the rooms in a direction parallel to the line electrodes. As shown in Figure 7 the detected extensions of the rooms are quite close to the actual length of the rooms. The solid line shows the detected length of the cross-section of the rooms and the broken line shows the actual length of the cross-section of the rooms. Therefore the height, width, length, and depth of the overburden for the detected rooms are fairly close to their actual location and dimension. Figure 8 shows the resistivity block diagram related to the room and pillar for $C_1C_2$ activation. This figure shows the resistivity anomaly response for the survey area. The four peak anomalies are related to the four rooms, and the anomaly amplitude diminishes when it is far from the rooms.

Similitude model study of cavity detection. The dimensions of the model are 8 feet long by 4 feet wide (Fig. 9). The very bottom layer or the sandstone layer consists of sand, lime and cement, in proportions of 20:4:1. The second layer consists of rock dust, lime and cement and the proportions of the material used are 10:3:1. The next layer contains very fine-grained coal, lime and cement. The subsequent layer on top of the coal layer is a limestone layer which consists of only lime. On top of this limestone layer is another layer which consists of sand and cement. The proportions of the material used are 5:1. Figure 9 shows the structure and sequences of the five layers used for the similitude model. For all the layers the most important factor is the resistivity property of the materials in each layer of the structure. Other factors such as the Poisson's ratio or elastic modulus were not important in these experiments. Actually, this simulation was designed to demonstrate the resistivity contrast between different layers and to examine how different layers affect the accuracy of the experiments results. Analogous to the tank model three line electrodes were used. The electrode setup was the same as the first experiment. This experiment was conducted to verify the characteristic curve of resistivity for

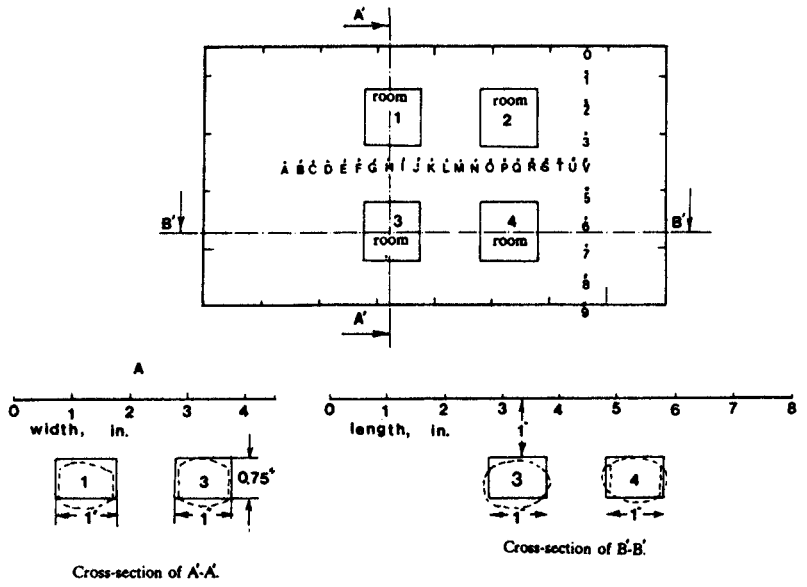



Fig. 3 Location of simulated rooms in the tank model.

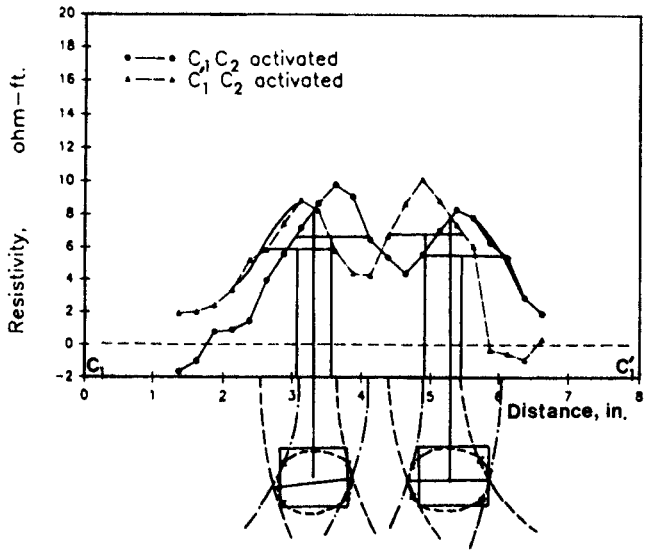



Fig. 4 Interpretation of the resistivity profile for cross-section 3-3.

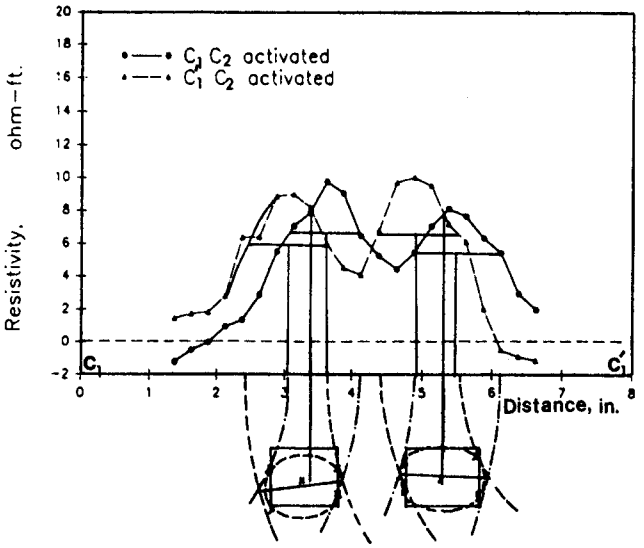



Fig. 5 Interpretation of the resistivity profile for cross-section 7-7

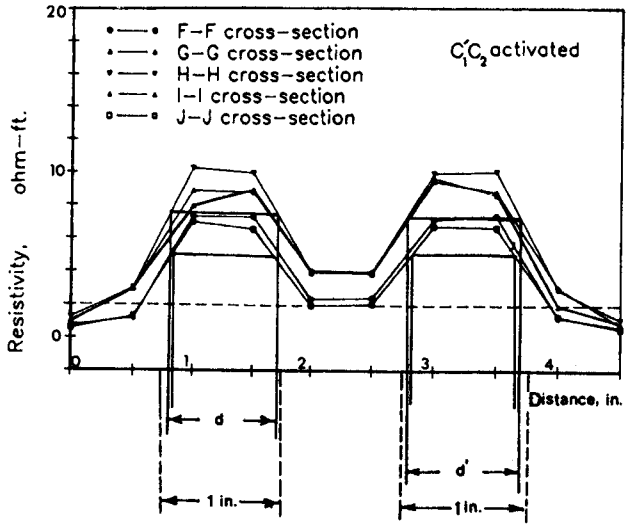



Fig. 6 Interpretation of the resistivity profiles for cross-sections parallel to the line electrodes.

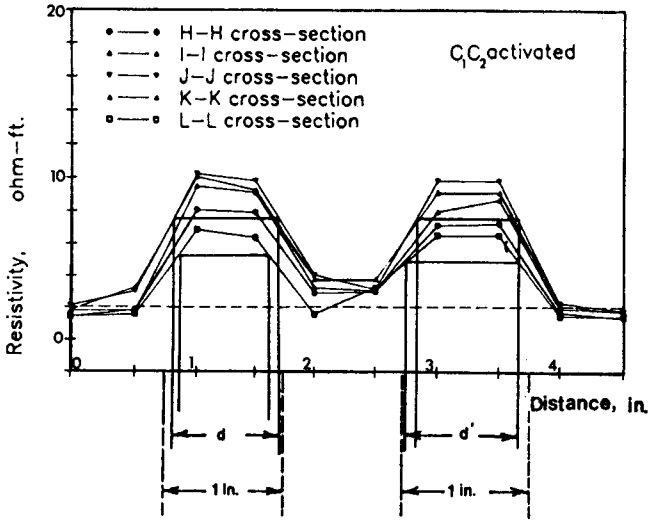



Fig. 7 Interpretation of the resistivity profiles for cross-sections parallel to the line electrodes.

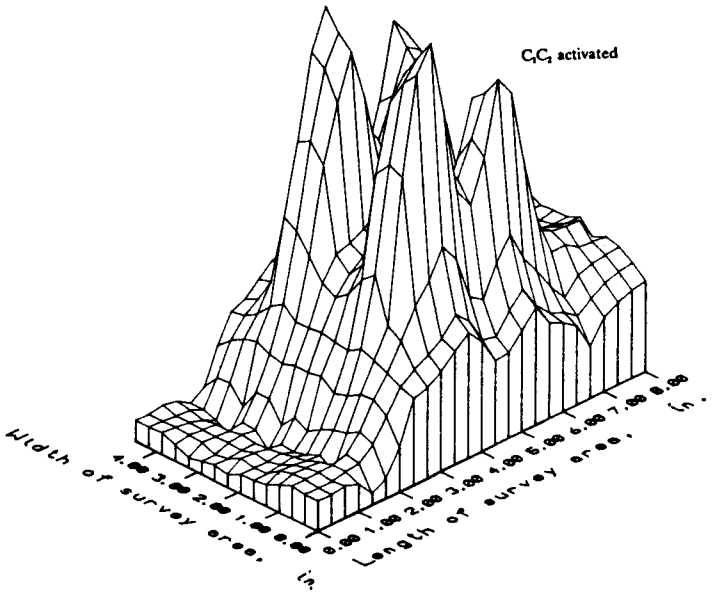



Fig. 8 Block diagram of measured resistivity for 1" depth room and pillar simulation.

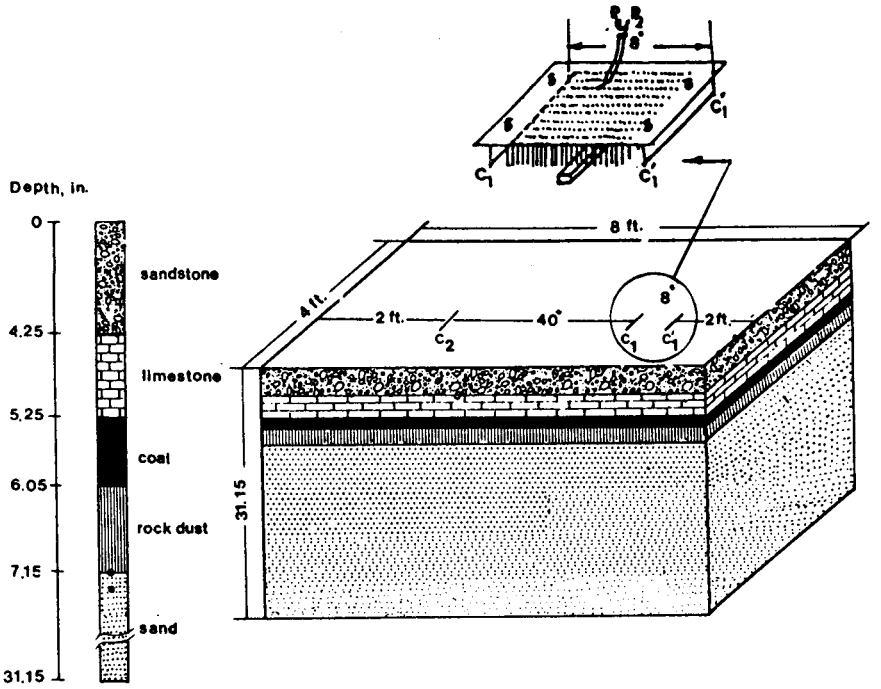



Fig. 9 The similitude model.

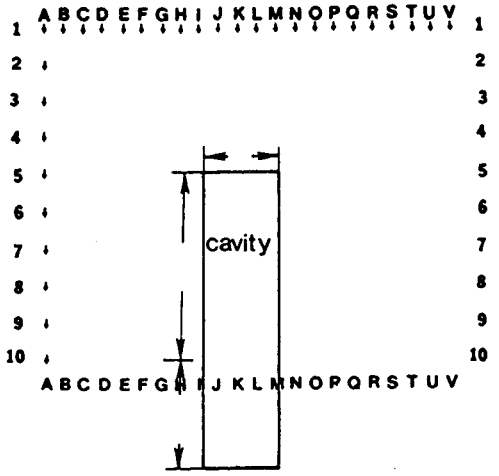



Fig. 10 Location of the created cavity within the survey area in the similitude model. The size, location, and depth of the cavity vary for any experiment.

the five layer structure without cavity creation in the similitude model. Figure 10 shows the view of the survey site. Figure 11 shows the resistivity of cross-section 8-8 stations located between midpoint of the potential electrodes versus their distance from $C_1C_2$ activated line electrodes. This resistivity curve is consistent with the characteristic curve of multiple layers. As shown, the apparent resistivity in this profile begins with the sandstone layer, increases in the limestone layer and then decreases in the coal layer as depth of the penetration of the injected current in the medium increases. For the rock dust layer, the apparent resistivity may be somewhere between the coal and sandstone layer because there is no apparent changed for this layer. Finally, the curve terminates with an increasingly apparent resistivity for the bottom sand layer. This type of curve is generated because the relative resistivity of the sandstone layer is less than the apparent resistivity of the limestone layer, but is higher than the apparent resistivity of the coal layer. The relative resistivity of the coal layer is also less than that of sand or rock dust layer. Therfore, the resistivity profile can provide an idea regarding the structure of the subsurface layers.

The next experiment was conducted to detect a circular cavity with a diameter of 0.70 inches that was located at a depth of 1.25 inches (Fig. 12 shows the cavity creation in the model). In the similitude model the resistivity profiles for the cross-section 8-8 was prepared (Fig. 13). The depth of the cavity was 1.30 inches in the sandstone layer. The two anomaly curves for different activation of line electrodes $C_1C_2$ and $C'_1C_2$ were plotted versus the distance of the midpoint of the potential electrode. For this type of the experiments started from one 3/4th of the maximum amplitude was used for drawing the horizontal line for interpretation purpose rather than 2/3 ratio used in the tank model experiments. The anomaly was interpreted and the predicted cavity is shown in broken line and the actual cavity in solid line. The location of the actual cavity and the detected one is close. The height of the cavity was estimated very closely to the actual one, but the width of the detected cavity was slightly overestimated. The calibration of the response anomaly related to the cavity dimension was practiced initially in order to be applied for anomaly interpretation of different resistivity profiles. Figure 14 shows the result of an anomaly response and the corresponding interpretation and locating of another cavity with a diameter of 0.20 inches which was located at a depth of 1.40 inches. In this case the detected cavity was found to be slightly deeper than its actual location. Also, the width of the cavity was overestimated slightly. The detectability ratio in this case was 7.0.

Finally, a cavity was created at a depth of 2.25 inches with a diameter of 0.20 inches. The profile of the cross section 8-8 is shown in Figure 15. The detected cavity is shown in broken line and the actual cavity in solid line. It is clear that the size and location of the cavity is in good agreement with the size and location of the actual cavity. The ratio of detectability was 11.5 in this experiment.

## Conclusion

The new technique of cavity location appears to be applicable for cavity detection in experimental study either in the brine tank model or similitude model. The dimension and the depth of the cavity can be estimated by this technique within certain limits. This technique also provides some ideas related to subsurface structure as well.

## Acknowledgment

This project was supported by the U.S. Bureau of Mines under Contract No. J0178030. The opinions expressed here are strictly the authors.

## References

Bates, E. R. 1973. Detection of subsurface cavities. U.S. Army Waterway Experiment Station Miscellaneous Paper. 5-73-40: 63 p.

Bristow, C. 1966. A new graphical resistivity technique for detecting air filled cavities. Study in Speleology 1(4): 204-277.

Burdick, R. G., L. E. Snyder, and W. F. Kimbrough. 1986. A new method for locating abandoned mines. U. S. Bureau of Mines RI-9050: 27.

Carpenter, E. W. and G. M. Habberjam. 1955. A tri-potential method for resistivity prospecting Geophysics. 21:455-470.

Habberjam, G. M. 1969. The location of spherical cavities using a tri-potential resistivity technique Geophysics 34(3):780-784.

Logn, O. 1954. Mapping vertical discontinuities by earth resistivities Geophysics 19:739-760.

Peng, S. S. and S. M. Hsiung. 1988. Development of guidelines for stabilizing and monitoring the effectiveness of abating subsurface prone area over AML. Annual report submitted to U. S. Bureau of Mines (contract no. J0178030):63.

Telford, W. M., L. P. Geldart, and R. E. Sheriff. 1976. Applied Geophysics, Cambridge University Press, 860 p.

Ziaie, F., S. S. Peng, and S. M. Hsiung. 1989. Experimental study of line electrode method to detect underground cavities. p. 369-376. Rock Mechanics as a Guide for Efficient Utilization of Natural Resources. Edited by A. W. Khair, A. A. Balkema publishers.

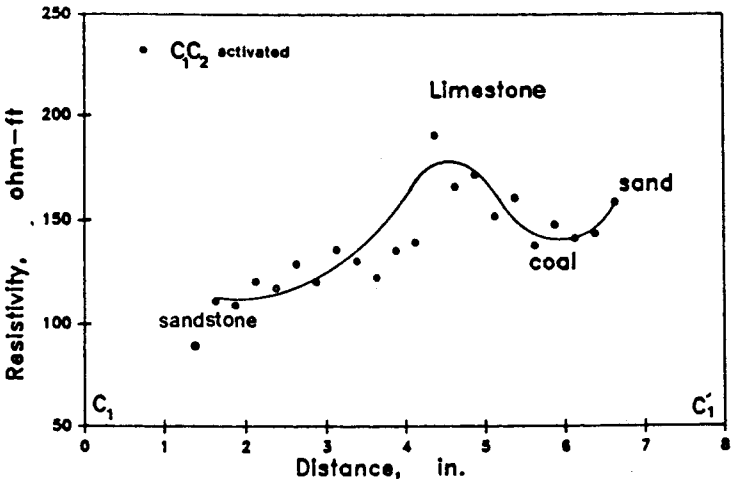



Fig. 11 Resistivity curve of the similitude model.

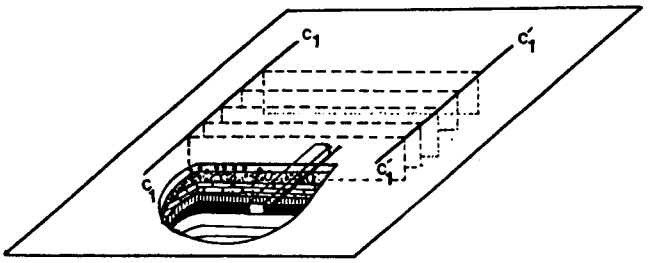



Fig. 12 Cavity creation in the similitude model.

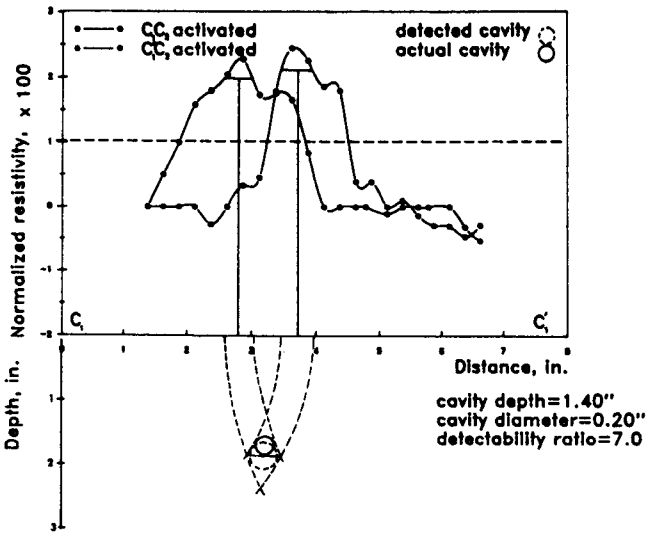



Fig. 13 Resistivity profiles for cross-section 8-8 when the cavity is located at 1.40" depth.

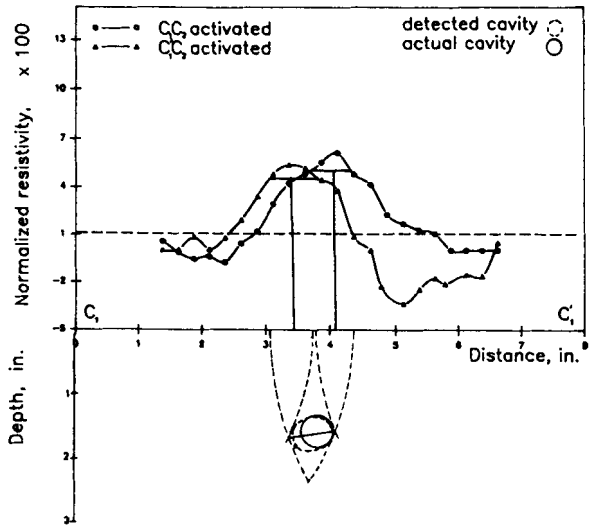



Fig. 14 Resistivity profiles for cross-section 9-9 when the cavity is located at 1.25" depth.

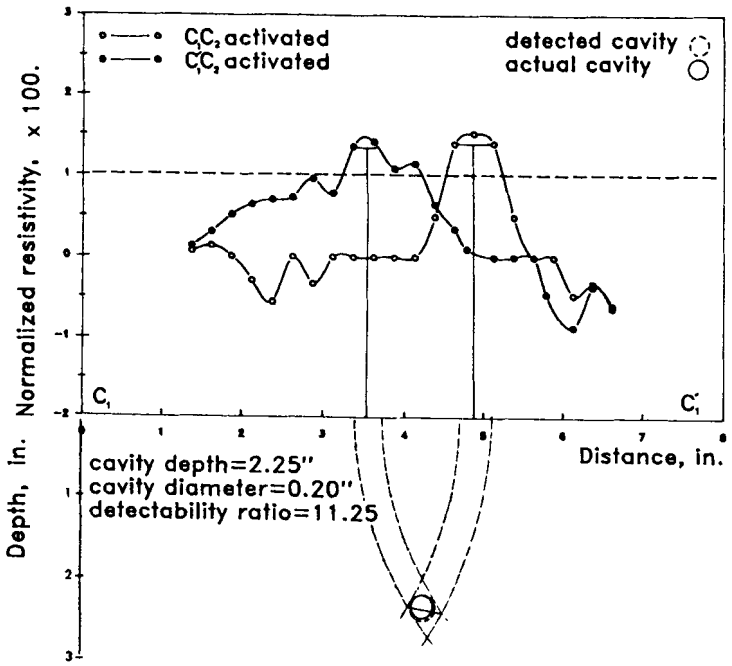



Fig. 15 Resistivity profiles for cross-section 8-8 when the cavity is located at 2.25" depth.

# HIGHWALL STABILITY ANALYSIS ON A PC[1]

P. M. Lin, D. Dutta, S. S. Peng[2]

**Abstract.** Conventional methods of slope stability analysis were found to be inadequate for analyzing highwalls consisting of different strata. A new method was developed to calculate the factor of safety for dry highwalls and the model was used to develop computer software for a personal computer. Presently, the program analyzes plane and circluar failures. Unlike the conventional random search technique, the critical slip surface is determined by a pattern optimation technique.

Additional Key Words: Highwall stability, Surface mining

## Introduction

Analysis of abandoned highwalls is important because of the dangers a failure of the highwall poses to life and property. The conventional approach is to use available limiting equilibrium analysis of Bishop (1955), Janbu (1954), or Morgenstern and Price (1965) developed for slope stability problems. The above methods require that the shape and location of the potential slip surface be specified; or the location is determined by different optimization methods depending on the program algorithms. In the case of highwalls consisting of a number of strata, the above methods are not only inefficient but also, sometimes they do not yield the actual slip surface. In this paper, in order to overcome this problem, the factor of safety is derived by considering available shear strength and mobilized shear stress along an arbitrary failure surface. However, for practical applications, the solution algorithm still needs to assume the shape of failure surface. Plane and circular failures have been fully analyzed in this paper using the new concept. The critical failure surface, a sliding surface which has the minimum factor of safety, is then obtained adusting the surface in order to minimize the factor of safety. Minimization of the factor of safety is achieved by an optimization technique which is a mathematical process of maximizing or minimizing a function. The new development to the problem of highwall stability analysis has been implemented on a PC.

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]P. M. Lin, D. Dutta and S. S. Peng are Research Associate, Graduate Student and Charles T. Holland Professor and Chairman, respectively, at Department of Mining and Engineering, College of Mineral and Energy Resouces, West Virginia University, Morgantown, WV 26505

## Theoretical basis

Referring to Figure 1, $y_1 = f(x)$ and $y_2 = g(x)$ are the equations of the slip surface and slope surface respectively. The toe of the slope is taken as the origin of the X-Y coordinate system. For an infinitely small slice of thickness dx, $d\tau_a$ is the available shear strength and $d\tau_m$ is the shear stress mobilized along the failure surface $y_1$. The force equilibrium equation for the system can be written as:

$$d\tau_m - d\tau_a = 0 \qquad (1)$$

From Figure 1, the following relationships for $d\tau_m$ and $d\tau_a$ can be established:

$$d\tau_m = \gamma(y_2 - y_1)\sin\alpha \, dx \qquad (2)$$

$$d\tau_a = c \, d\ell + \tan\phi \, \{\gamma(y_2 - y_1) \cos\alpha \, dx\} \qquad (3)$$

where, c is cohesion, $\phi$ is angle of internal friction ($\mu = \tan\phi$) and $\gamma$ is unit weight of the stratum; $\ell$ is the arc length along $y_1$. Substituting Equations (2) and (3) into Equation (1), we get:

$$\gamma(y_2 - y_1)\sin\alpha \, dx - c \, d\ell - \mu(\gamma(y_2 - y_1)\cos\alpha \, dx\} = 0 \qquad (4)$$

In addition, the following relationships can be derived from differential calculus:

$$\cos\alpha = \frac{dx}{d\ell} \qquad (5)$$

$$\sin\alpha = y_1' \frac{dx}{d\ell} \qquad (6)$$

$$d\ell = \sqrt{1 + y_1'^2} \, dx \qquad (7)$$

## Factor of safety

In order to quantify the margin of safety relative to a state of imminent failure, the strength parameters c and $\mu$ are to be reduced by a factor F known as factor of safety. Hence,

$$c = \frac{c_m}{F} \text{ and } \mu = \frac{\mu_m}{F} \tag{8}$$

where, $c_m$ and $\mu_m$ are mobilized cohesion and coefficient of friction respectively (we will use the symbol c and $\mu$ to denote $c_m$ and $\mu_m$ throughout the paper). Using Equations (5), (6), (7) and (8), Equation (4) can be written as:

$$\frac{\gamma(y_2 - y_1)y_i'}{\sqrt{1 + y_i'^2}}\,dx - \frac{c}{F}\sqrt{1 + y_i'^2}\,dx - \frac{\mu}{F}\frac{\gamma(y_2 - y_1)}{\sqrt{1 + y_i'^2}}\,dx = 0 \tag{9}$$

Equilibrium of the mass bound by OEG (Figure 1) is:

$$\int_{x_o}^{x_n} \{F\gamma(y_2 - y_1)y_i' - c(1 + y'^2) - \mu\gamma(y_2 - y_1)\}dx = 0 \tag{10}$$

Factor of safety F can be obtained from Equation (10) as:

$$F = \frac{\int_{x_o}^{x_n} \{c(1 \times y_i'^2) + \gamma\mu(y_2 - y_1)\}\,dx}{\int_{x_o}^{x_n} \gamma(y_2 - y_1)y_i'\,dx} \tag{11}$$

**Factor of safety for layered highwall.** The above formulation of the safety factor is for the single layered highwall. In practice, however, highwalls consist of different strata (Figure 2). A factor of safety for an n layered highwall can be formulated as:

$$F = \frac{\sum_{i=1}^{n} \int_{x_{i-1}}^{x_i} \{c_i(1+y_i'^2) + \gamma\mu_i(y_2-y_1)\}\,dx}{\int_{x_o}^{x_n} \gamma(y_2 - y_1)y_i'\,dx} \tag{12}$$

where $\gamma$ is taken to be the weighted average unit weight of all the strata in the highwall. This is done to avoid complexity in evaluating the above integrals. Note: $c_i$ and $\mu_i$ are for the layers in which the failure plane is passing.

## Plane shear failure

In the case of plane shear failure of a rectilinear highwall of height H and slope angle $\theta$ (Figure 3), the following relationships for $y_2$ and $y_1$ can be derived:

$$y_2 = x\tan\theta \quad \text{for} \quad x \leq \frac{H}{\tan\theta} \tag{13}$$

$$y_2 = H \quad \text{for} \quad x \geq \frac{H}{\tan\theta} \tag{14}$$

$$y_1 = x\tan\beta \tag{15}$$

The result of integrating each term of Equation (12) by using Equations (13), (14) and (15) for $y_2$ and $y_1$, is:

$$A = \sum_{i=1}^{n} \int_{x_{i-1}}^{x_i} \{c_i(1+y_i'^2)dx = \sec^2\beta \sum c_i (x_i - x_{i-1}) \tag{16}$$

$$B = \sum_{i=1}^{n} \int_{x_{i-1}}^{x_i} \gamma\mu_i(y_2 - y_1)dx = \sum_{i=1}^{n} \int_{x_{i-1}}^{x_i} \gamma\mu_i y_2 dx - \sum_{i=1}^{n} \int_{x_{i-1}}^{x_i} \gamma\mu_i y_1 dx \tag{17}$$

or

$$B = \gamma\Sigma\,\mu_i\left[\frac{\tan\theta}{H}(x_i^2 - x_{i-1}^2) - \frac{\tan\beta}{2}(x_i^2 - x_{i-1}^2)\right] \quad x \leq \frac{H}{\tan\theta} \tag{18}$$

$$B = \gamma\Sigma\,\mu_i\left[H(x_i - x_{i-1}) - \frac{\tan\beta}{H}(x_i^2 - x_{i-1}^2)\right] \quad x \geq \frac{H}{\tan\theta} \tag{19}$$

$$C = \int_{x_o}^{x_n} \gamma(y_2 - y_1)y_i'dx = \gamma\int_{o}^{H/\tan\theta} y_2 y_i'dx + \gamma\int_{H/\tan\theta}^{x_n} y_2 y_1 dx - \int_{x}^{x_n} y_1 y_i'dx \tag{20}$$

or

$$C = \gamma\left[\frac{\tan\theta\tan\beta}{2}\left(\frac{H}{\tan\theta}\right)^2 + H\tan\beta\left(x_n - \frac{H}{\tan\theta}\right) - \tan^2\beta\,\frac{x_n^2}{2}\right] \tag{21}$$

In order to establish the critical sliding surface, the factor of safety F = (A + B)/C is to be minimized for the unknown variable $x_n$. This is done by an optimization method described later.

## Circular failure

For a circular failure surface of radius r having the center at the coordinates $x_c$ $y_c$ (Figure 4), the sliding surface is:

$$y_1 = y_c - \sqrt{r^2 - (x - x_c)^2} \tag{22}$$

The result of integrating each term of Equation (12) by using Equations (13), (14) and (22) for $y_2$ and $y_1$, is:

$$A = \sum \frac{c_i r}{2} \left\{ \log\left[\frac{r + (x_i - x_c)}{r - (x_i - x_c)}\right] - \log\left[\frac{r + (x_{i-1} - x_c)}{r - (x_{i-1} - x_c)}\right]\right\} \quad (23)$$

$$B = B_1 - B_2 \quad (24)$$

for $x \leq \frac{H}{\tan\theta}$

$$B_1 = \gamma \sum \mu_i \left[\tan\theta \frac{x_i^2}{2} - y_c x_i + \frac{x_i - x_c}{2}\sqrt{r^2 - (x_i - x_c)^2} + \frac{r^2}{2}\sin^{-1}\frac{x_i - x_c}{r}\right] \quad (25)$$

$$B_2 = \gamma \sum \mu_i \left[\tan\theta \frac{x_{i-1}^2}{2} - y_c x_{i-1} + \frac{x_{i-1} - x_c}{2}\sqrt{r^2 - (x_{i-1} - x_c)^2} + \frac{r^2}{2}\sin^{-1}\frac{x_{i-1} - x_c}{r}\right] \quad (26)$$

and for $x \geq \frac{H}{\tan\theta}$

$$B_1 = \gamma \sum \mu_i \left[H x_i - y_c x_i + \frac{x_i - x_c}{2}\sqrt{r^2 - (x_i - x_c)^2} + \frac{r^2}{2}\sin^{-1}\frac{x_i - x_c}{r}\right] \quad (27)$$

$$B_2 = \gamma \sum \mu_i \left[H x_{i-1} - y_c x_{i-1} + \frac{x_{i-1} - x_c}{2}\sqrt{r^2 - (x_{i-1} - x_c)^2} + \frac{r^2}{2}\sin^{-1}\frac{x_{i-1} - x_c}{r}\right] \quad (28)$$

$$C = C_1 + C_2 + C_3 - C_4 \quad (29)$$

$$C_1 = \gamma \left\{ \int_0^{H/\tan\theta} \tan\theta \left[\left(\frac{r^2}{2} + x_c^2\right)\sin^{-1}\frac{x - x_c}{r} - \frac{1}{2}(x + 3x_c)\sqrt{r^2 - (x - x_c)^2}\right] \right\} \quad (30)$$

$$C_2 = \gamma \left\{ \int_0^{H/\tan\theta} -x_c \tan\theta \left[x_c \sin^{-1}\frac{x - x_c}{r} - \sqrt{r^2 - (x - x_c)^2}\right] \right\} \quad (31)$$

$$C_3 = \gamma \left\{ \int_{H/\tan\theta}^{x_n} -H\left[\sqrt{r^2 - (x - x_c)^2}\right] \right\} \quad (32)$$

$$C_4 = \gamma \left\{ \int_0^{x_n} x_c x - \frac{1}{2}x^2 - y_c\sqrt{r^2 - (x - x_c)^2} \right\} \quad (33)$$

From the above equations, the factor of safety can be determined as F = (A + B)/C. The four unknowns, $x_c$, $y_c$, r and $x_n$, in the objective function, F, are reduced to two unknowns by the following transformations (Figure 4):

$$r = \frac{H^2 + x_n^2}{2(x_n\cos\omega - H\sin\omega)} \quad (34)$$

$$x_c = x_n - r\cos\omega \quad (35)$$

$$y_c = H + r\sin\omega \quad (36)$$

where, $\omega$ is the angle between the radius EF of the arc and slope line EG.

By using the optimization method, $x_n$ and $\omega$ are determined for the minimum value of F.

## Optimization technique

For optimizing the function F, a pattern search method has been used in the program (Hooke and Jeeves 1967). This ia a straightforward method and intuitively appealing. Unlike the classical gradient method of optimization, the pattern search technique does not require finding the derivative of the function. Because of this, there is no assumption of continuity or the existence of first and second derivatives of the function in a specified interval.

Unlike the random search technique, the pattern search method employs sequential examination of the trial solutions. Each trial solution is compared with the "best" obtained up to that point and the next trail solution is determined by a strategy. The method evaluates the objective function at each base point. The procedure of going from a given base to the next base is called a move. A move is a success if the objective function decreases; otherwise, it is a failure. The pattern seach routine involves two types of moves. The first is an exploratory move designed to acquire knowledge about the behavior of the function. In the second move, the pattern move, the actual minimization of the function is performed by utilizing the result obtained from the exploratory move. Figures 5 and 6 show the detailed flow charts of the pattern and exploratory moves respectively. Table 1 explains the variables used in the flow charts.

## Program structure

The program is written in TURBO BASIC and consists of input, factor of safety, optimization, graph and print subroutines. The input subroutine requests necessary data, such as, H, $\theta$, $\gamma$, $c_i$, $\mu_i$ and $h_i$ (thickness of each stratum from the bottom of the highwall). The input data are checked for errors and an option is provided to store them on the disk. A separate subroutine calculates F for plane and circular failure with initial values of $x_n$ = (H/tan$\theta$) + 1 for plane failure and $x_n$ = (H/tan$\theta$), $\omega$ = 0 for circular failure. The optimization subroutine optimizes F by repeatedly calling the factor of safety routine. The minimum F is the current F which is less than the previous F for a minimum step size ($\delta$) 0.01. A graph subroutine displays the slope with

the failure surface on the screen. The print subroutine prints the output on the screen as well as on the line printer. Table 2 shows a demonstration run of the program. For plane failure, the minimum factor of safety is found to be 0.87 and the intersection of the slope surface and sliding surface ($x_n$) is located at a distance of 61.7 feet from the origin. For circular failure, the radius of the slip circle is 87.8 feet having its center at -42, 77. The minimum factor of safety for the circular failure is 0.72 and $x_n$ is at a distance of 45.5 feet from the origin.

For computing the factor of safety, the following steps are performed.

For plane failure:

1. Enter initial value of $x_n = (H/\tan\theta) + 1$.
2. Calculate $\tan\beta = H/x_n$.
3. Calculate $x_i$ ($i = 1$ to $n - 1$) from the equation:

$$x_i = \frac{\Sigma h_i}{\tan\beta}$$

4. Calculate A, B, and C from Equations (16), (18) or (19), and (21).
5. Calculate the factor of safety $F = (A + B)/C$.

For circular failure:

1. Enter initial values of $x_n = H/\tan\theta$ and $\omega = 0$.
2. Calculate r, $x_c$, and $y_c$ using Equations (34), (35), and (36).
3. Calculate $x_i$($i = 1$ to $n - 1$) from equation:

$$x_i = x_c + \sqrt{r^2 - (\Sigma h_i - y_c)^2} \qquad (38)$$

4. Calculate A, B, and C from Equations (23), (24), and (29).
5. Calculate the factor of safety $F = (A + B)/C$.

The factor of safety subroutine consists of steps 2 through 5. The optimization subroutine changes values of $x_n$ and $\omega$ and calls the factor of safety subroutine to calculate F. The current F is compared with the previous F for undertaking further moves. The operation will stop when the current F is less than the previous F. Figures 5 and 6 explain the operation of the optimization subroutine.

## Conclusion

Analysis of highwall stability by using the conventional force equilibrium method has been successfully accomplished on a PC. The random search techniques to locate the critical slip surface have been replaced by an optimization method. Computational procedures and applications have proved that it is possible to run a comprehensive analysis of highwall stability on a PC involving circular or plane failures.

The approach described here to calculate factor of safety is very powerful. Though only simple cases like plane and circular failures have been fully analyzed herein, other modes of failures, such as, plane failure with tension cracks (no water in the tension crack), non-circular slip surfaces or even wedge failures can be analyzed by using the same procedure. It needs further modification and development of the current method if noncircular slip surfaces or wedge failures are to be incorporated.

The method described in this paper is one of the limit equilibrium methods and that only force equilibrium is being satisfied. The method does not account for moment equilibrium. So, the method does not satisfy complete equilibrium condition. The method has not been yet tested in the field.

## References


Bishop, A. W., 1955. The use of the slip circle in the stability analysis of earth slopes. Geotechnique 5(1): 7-17.

Hooke, R. and Jeeves, T. A., 1967. Direct search solution of numerical and statistical problems. Journal of Association of Computing Machinery 5: 212-229.

Janbu N., 1954. Application of composite slip surface for stability analysis. Proceedings of the European Conference on Stability of Earth Slopes. (Stockholm), p. 43-50.

Morgenstern, N. R. and Price, V. E., 1965. The analysis of the stability of the general slip surfaces. Geotechnique 5(1): 79-93.

Table 1. Explanations of the variables used in flow charts.

The variables $\Psi$, $\vartheta$ and $\varphi$ are points in n dimensional space.

| | |
|---|---|
| $\vartheta$ | The previous base point. |
| $\Psi$ | The current base point. |
| $\varphi$ | The base point resulting from the current move. |
| $F(\Psi)$ | The functional value at the base point. |
| $F(\varphi)$ | The functional value for this move. |
| F | The functional value befor this move. |
| $\Delta$ | The current step size. |
| $\delta$ | Minimum step size. |
| $\rho$ | Reduction for step size. |
| $\varphi$ | One of the coordinate values. |
| $E:F,\varphi$ | A program E(exploratory routine) will change F and $\varphi$. |

Table 2. Demonstration run of the program.

| | |
|---|---|
| **Geometrical Information of the Highwall** | |
| Height of the highwall (ft.): | 70 |
| Slope angle of the highwall: | 70 |
| **Strata Information** | |
| Strata 1. | |
| Thickness (ft.) | 25 |
| Cohesion ($lb/ft^2$) | 800 |
| Friction angle | 25 |
| Unit weight ($lb/ft^3$) | 160 |
| Strata 2. | |
| Thickness (ft.) | 15 |
| Cohesion ($lb/ft^2$) | 700 |
| Friction angle | 20 |
| Unit weight ($lb/ft^3$) | 170 |
| Strata 3. | |
| Thickness (ft.) | 30 |
| Cohesion ($lb/ft^2$) | 900 |
| Friction angle | 25 |
| Unit weight ($lb/ft^3$) | 180 |
| **Plane Failure** | |
| x(n) (ft.) | 61.7 |
| Factor of Safety | 0.87 |
| **Circular Failure** | |
| Radius of slip circle (ft.) | 87.8 |
| $x_c$ | -42 |
| $y_c$ | 77 |
| x(n) (ft.) | 45.5 |
| Factor of safety | 0.72 |

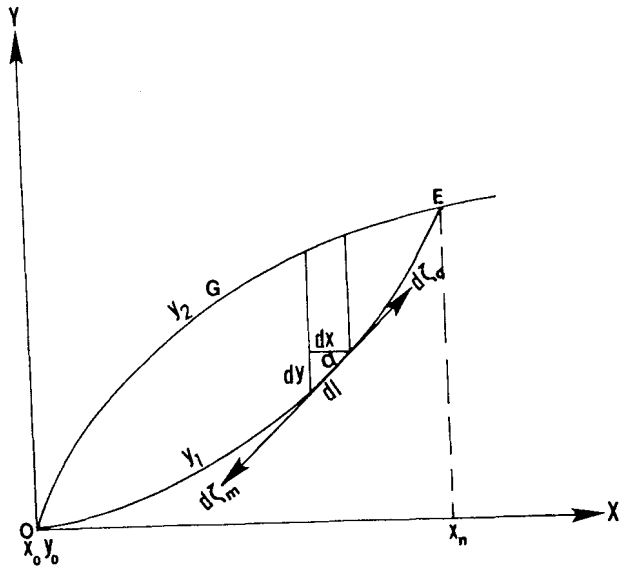



Figure 1. Basic conventions and definitions.

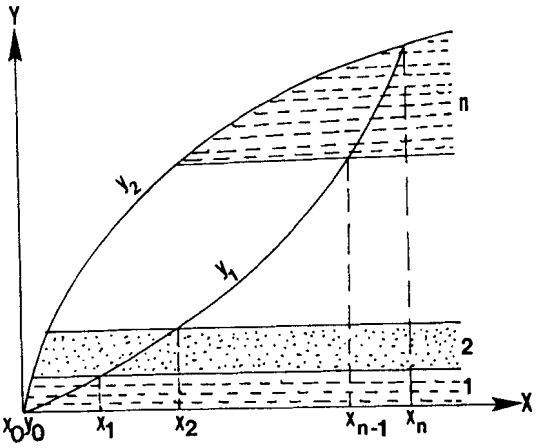



Figure 2. Conventions for layered highwall.

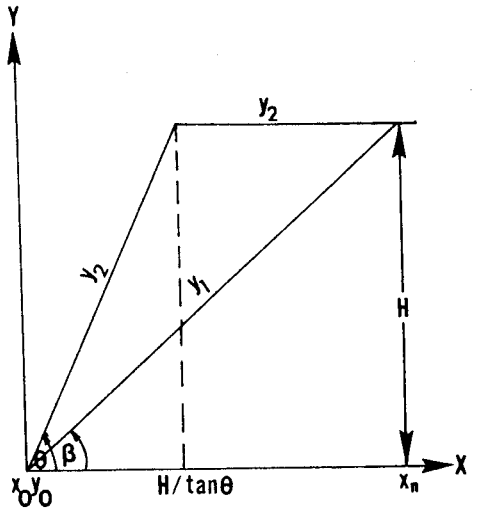



Figure 3. Plane failure.

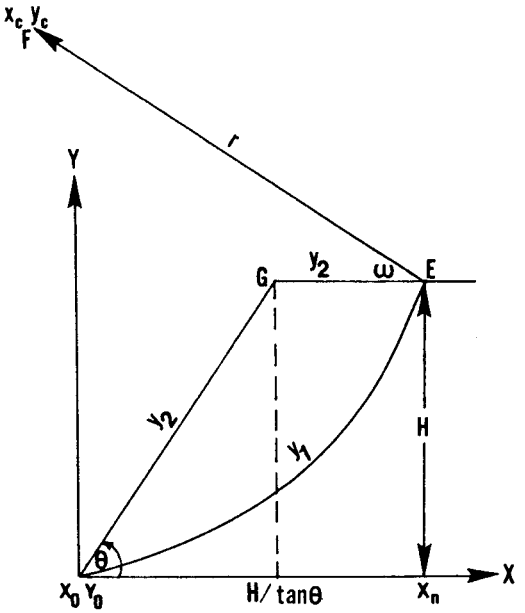



Figure 4. Circular failure.

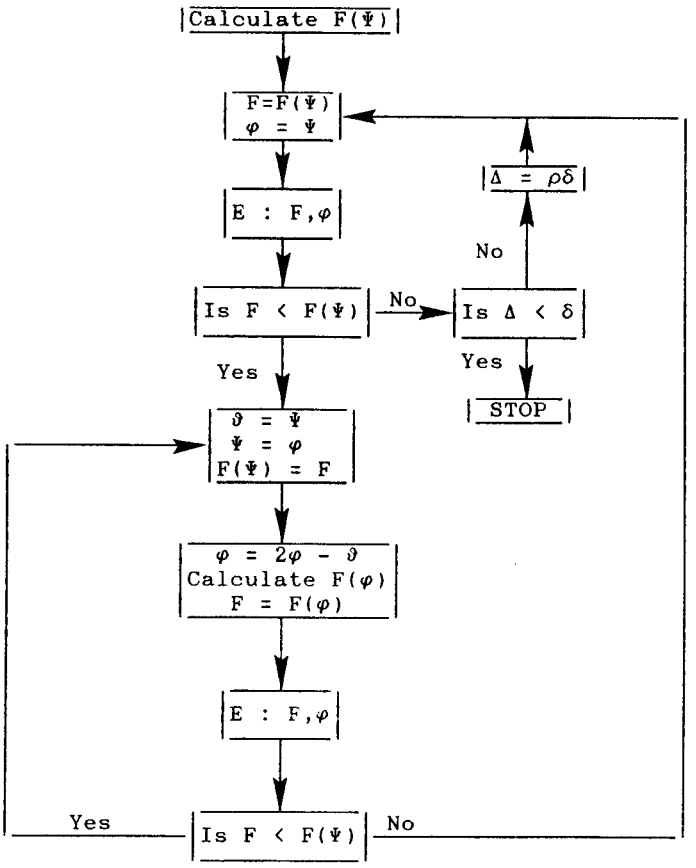
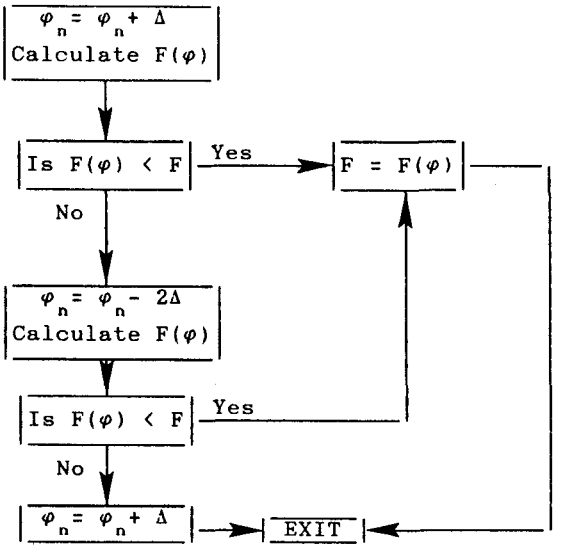



Figure 5. Flow chart for pattern move.




(This routine is carried out for each variable separately)

Figure 6. Flow chart for exploratory move.

This page intentionally left blank.

# USING A MAGNETOMETER FOR INVESTIGATING UNDERGROUND COAL MINE FIRES, BURNING COAL REFUSE BANKS, AND FOR LOCATING AMD SOURCE AREAS ON SURFACE MINES[1]

by

Joseph Schueck, P.E.[2]

**Abstract.** A variety of problems ranging from post mining acidic discharges to underground mine fires may develop on completed coal mining sites. Unless the site conditions can be accurately defined, solutions to these problems have little chance of success. Studies indicate acid mine drainage (AMD) often forms in localized areas within a backfilled strip mine. Underground mine fires and burning waste banks do not burn in a laterally continuous pattern. Accurate definition of site conditions using drilling is almost impossible because of this lateral discontinuity. However, geophysical surveying can provide much of this information.

Chemical or thermal oxidation of pyrite forms magnetic minerals. Determining the locations of the transformed pyrite equates to a knowledge of the AMD production areas or areas which have burned and cooled. Magnetic surveying can determine the spatial distribution of the resultant magnetic minerals. Electromagnetic terrain conductivity can delineate pollution plumes on AMD production sites.

An underground mine fire, a burning refuse bank, and a reclaimed strip mine producing AMD were mapped using magnetometry. Terrain conductivity was also used on the AMD site. Areas of the underground mine and refuse bank which had burned and cooled were delineated. AMD source areas and the resultant pollutional groundwater plume were defined on the third site.

Additional Key Words: magnetometry, electromagnetic terrain conductivity, acid mine drainage, underground mine fires, burning refuse banks, magnetic susceptibility

## Introduction

Acid mine drainage production and burning underground coal mines or refuse piles are two of the most significant environmental problems associated with coal mining. There are over 2500 miles of streams polluted by acid mine drainage (PaDER 1971) and more than 500 fires burning in abandoned coal waste banks and coal deposits across the United States (Chaiken, 1989). Subsidence and noxious gases related to underground mine fires caused hundreds of people to move from their homes and communities. Burning refuse banks cause air pollution. Living conditions near these piles are often intolerable due to high levels of noxious gases. Millions of dollars have been spent investigating and trying to abate these problems, usually with very limited success. An inability to define site conditions accurately leads to the limited success.

Underground mine fires often start along the crop line because of trash burning. Here the fire is visible and simple to define. However, once the fire spreads to the underground workings the location of the combustion zone is unpredictable. Venting smoke and fumes rising from subsidence cracks confirm that combustion is taking place but may appear hundreds of feet from the actual combustion zone. The US Bureau of Mines developed a deep mine fire diagnostics program to locate combustion zones (Justin and Kim 1988). However, this program requires extensive drilling and gas sampling. Knowing which areas have burned and cooled as well as areas which are heating or cooling would certainly aid in planning drill hole locations for this program.

Locating combustion zones in burning refuse banks is also quite difficult. Similar to a deep mine, venting gases may surface tens of feet from the combustion zone. Limited oxygen within the bank slows the rate of combustion and, to a large extent, controls the direction of the fire spread. Unlike underground mine fires which are stratigraphicaly controlled, a third dimension is added to refuse banks as many are over 100 feet in height. Refuse banks require definition of the combustion zones vertically as well as horizontally before solutions can be determined.

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]Joseph Schueck is a Hydrogeologist, Pennsylvania Department of Environmental Resources, Bureau of Mining and Reclamation, Harrisburg, PA 17120

Infiltrating precipitation or laterally moving groundwater entering a backfilled strip mine may come into contact with pyrite, forming acid mine drainage. The normal tendency is for this drainage to flow down-dip along the pit floor. It then emerges near the crop line as a toe of spoil discharge. Buried coal stockpiles, pit cleanings, and buried refuse piles often contain high concentrations of pyrite. Studies show these to be prime sources of AMD production even though they may be no larger than a few feet in any dimension. There is no apparent evidence on the reclaimed surface to indicate the locations of these toxic piles. Drilling programs seldom accurately delineate the AMD source areas. Drilling may locate a pile of toxics, but the investigator cannot determine the limits of the pile with confidence without extensive additional drilling. For this reason, past solutions to control AMD production at its source have been largely unsuccessful.

The ability of localized piles of pyritic materials to accelerate AMD production was observed on an active mine site in Fayette County. A small pile on pit cleanings intercepted water flowing across the pit floor. The changes in water quality were dramatic considering the short contact time. The acidity increased by 1600 mg/l, iron by 500 mg/l and sulfate by 1950 mg/l.

Geophysical mapping of these areas can provide valuable site information not obtainable through drilling. Each geophysical technique measures a physical property of the earth within the equipment's range of detection. Success with geophysics comes about when the physical property of the target which the technique measures differs sufficiently from the host material to permit detection. Combustion metamorphism causes significant changes in physical properties of coal and related pyritic materials. Chemical oxidation also causes major changes in physical properties of pyritic materials on AMD producing sites. In both cases, the physical properties change enough to allow detection by geophysical methods.

Thermal oxidation of pyrite as in a mine fire (Cisowski 1987) or chemical oxidation of pyrite as in a site producing AMD (Henderson 1984) results in the formation of magnetic minerals. Under proper conditions, these magnetic minerals can be detected and their distribution mapped through magnetic surveying. Acid mine drainage is a good electrical conductor because of ions in solution. On the other hand, mine spoil is a poor electrical conductor. Electromagnetic terrain conductivity can successfully delineate plumes and pools of AMD because of this contrast.

## Magnetic Surveying - Theoretical Considerations

The earth's magnetic field consists of lines of magnetic flux surrounding the earth similar to that of a large bar magnet. The total field intensity at any given point depends upon the flux line density which is twice as great at the poles as it is at the equator. A proton processing magnetometer measures the total field intensity. The magnetometer records magnetic intensities in units of gammas. In Pennsylvania, the total field intensity is about 55,500 gammas.

A magnetic object creates a local disturbance, or anomaly, in the earth's magnetic field. The magnitude of the anomaly as recorded by the magnetometer depends on several factors. These include: 1) the mass of the magnetic material present, 2) the magnetic material's ability to enhance the field locally (magnetic susceptibility), 3) the burial depth or source-to-sensor distance, 4) the shape and size of the source, and 5) the amount of magnetic material in the source compared to the adjacent rocks.

The ability of a material to enhance the local field is called magnetic susceptibility, k. For most materials k is much less than 1 and is usually $10^{-6}$ cgs or smaller. The susceptibility or k of magnetite is approximately 0.3 cgs. The magnetic susceptibility of a rock containing magnetite or any other magnetic mineral is simply related to the amount of magnetite is contains. For example, rock containing 1% magnetite will have a volume susceptibility of $3 \times 10^{-3}$ cgs. The volume magnetic susceptibility for most sedimentary rocks is $10^{-6}$ to $10^{-5}$ cgs. (Breiner 1973).

Similar to a bar magnet, magnetic objects are polar, that is north and south seeking poles are present. When magnetic targets are mapped and the results contoured, the effects of the poles are reflected in the shape of the anomaly. Values of magnetic intensity greater than background levels (positive anomalies) will be recorded directly above the target. Values lower than background (negative anomalies) will normally be recorded adjacent to the positive anomaly (Breiner 1973). For clarity, only the positive portions of the anomalies are shown in the figures used in this paper.

At the three sites included in this study the targets were magnetic minerals resulting from chemical or thermal alteration of pyrite. The percent pyrite and the process by which the pyrite is altered ultimately determines the volume magnetic susceptibility, hence its ability to be detected. In the case of the underground mine fire and the burning refuse bank the volume magnetic susceptibility of the ash, also known as klinkers or red dog, was determined to be approximately $2 \times 10^{-3}$ cgs. The volume magnetic susceptibility of the chemically altered pyrite at the AMD site was determined to be approximately $4 \times 10^{-5}$ cgs. Since thermally altered pyrite exhibits a volume magnetic susceptibility two orders of magnitude greater than that of chemically altered pyrite, the magnetometer can more readily detect thermally altered pyrite.

The burning material in an underground mine fire consists of the coal pillars, binder material, and sometimes roof rock. Pyrite is common to all these materials. Temperatures in the burning zone commonly are in the range of 800 to 1000 degrees C, well above the Curie temperature (Chaiken et al 1989). Upon heating above the Curie temperature, magnetic minerals lose their natural magnetism. Small regions, called domains, within the crystal structure reorient themselves. Upon cooling they align themselves more or less in the direction of the ambient magnetic field and thus parallel to each other. This remanent magnetism may be several orders of magnitude greater than the natural magnetism (Breiner,1973). The minerals which are formed and their respective magnetic strengths depend on the temperatures reached and the amount of oxygen available (Cisowski 1987).

The chemical reactions which produce acid mine drainage also provide a mechanism for the pyritic source areas to be detected and mapped using a magnetometer. When proper pH-Eh conditions exist, a magnetic mineral phase of iron oxide forms. This mineral is thought to be maghemite, $Fe_2O_3$, a dimorphous form of hematite (Henderson 1984). When this mineral forms in a localized area, such as in buried piles of refuse or coal stockpiles, the volume magnetic susceptibility is sufficient for detection by a magnetometer.

Tipple refuse detection may be enhanced by one or two additional magnetic sources not found in pit cleanings or coal stockpiles. Two sites were mapped in 1987 where tipple refuse was buried. Analysis of the refuse at both sites revealed traces of magnetite crystals and ash particles. Magnetite crystals used in the coal separation process were lost to the refuse. It is thought that the ash particles resulted from spontaneous combustion. Both the magnetite and the ash exhibit high magnetic susceptibilities (Schueck 1988).

## Magnetic Surveying - Field Methods

A proton processing magnetometer with a sensitivity of 0.1 gamma was used for this study. It consists of a sensor mounted on top of an 8 foot aluminum staff. The sensor is connected to a power source and recording device carried in a harness worn on the chest.

Taking advantage of the source-to-sensor distance fall-off factor, a second sensor can be mounted on the staff two or four feet lower than the top sensor. The magnetic gradient is measured in this mode and the magnetometer functions as a gradiometer. Field intensity readings taken by both sensors helps to resolve and better define the source.

A recording base station was used at the AMD site. This permits rapid removal of diurnal fluctuations in the magnetic field from the field readings, thus improving accuracy.

Grids were established in the field at the three sites. Readings were then taken at each grid intersection. The required grid spacing depends on the expected target size, burial depth, and volume magnetic susceptibility. The grid spacing varied from site to site but averaged 25 feet. This spacing was adequate to locate the magnetic sources. Several of the smaller anomalies required mapping on a five by ten foot grid in the gradiometer mode for better definition.

A commercially available magnetic forward modeling computer program was used to interpret the recorded anomalies. Parameters such as shape, burial depth, and magnetic susceptibility are input into the program. The output is a theoretical field curve which is compared to the actual field readings. Changes in the input data are made until a close agreement is reached between the theoretical and recorded values.

## Electromagnetic Terrain Conductivity - Theoretical Considerations and Field Operation

Electromagnetic terrain conductivity (EM) has been used successfully for several years to map contaminant plumes (Ladwig 1982). EM operates as follows. A sinusoidally varying magnetic field from a transmitter electromagnetically induces currents in the ground. This is done in such a manner that their amplitude is linearly proportional to the terrain conductivity. The magnitude of these currents is determined by measuring the magnetic field which they in turn generate (McNeill 1980). The receiver displays these measurements in units of mmhos/m.

The depth of investigation for EM can be varied by using different coil orientations and spacings. Thus the near surface conditions as well as conditions at depth can be determined. The instrument used during this study was a single man unit with capabilities of determining terrain conductivities to depths of about ten and 20 feet.

The ground is generally a poor electrical conductor. By contrast acid mine drainage is an excellent conductor because it is an electrolytic solution. Thus, plumes and pools of mine drainage are excellent targets for EM and are easily detected. The electrical conductivity of an entire mine site can be determined rather quickly by obtaining readings along a grid system. The recorded values are then plotted and contoured. Correlation of the results can be achieved with a limited amount of drilling.

EM may provide information about a mine site which monitoring wells may not. Following a precipitation event, water commonly moves through the mine spoil as a wetting front. Monitoring wells located within the plume may not indicate the presence of water unless it pools in the area of the well. However, as the wetting front moves through the spoil, the rocks remain wet. Water associated with AMD source areas and flow paths will contain more dissolved ions than water in areas not producing mine drainage. These areas can then be defined with EM even though their presence is not indicated by monitoring wells.

## Case Studies

### An Underground Coal Mine Fire

An underground coal mine in southwestern Pennsylvania has been burning for decades. Three fires of separate origin are slowly spreading through this mine. Designated DMF1, this 120 acre deep mine was developed in the Pittsburgh coal seam. The coal averaged a thickness of nine feet. The carbonaceous shale roof rock is also combustible. The maximum cover over the mine is 120 feet. Retreat mining was extensive, according to the mine maps. Thus, roof collapse would have been prevalent and the previously established headings would no longer be well defined. This restricts air circulation, slowing the fire spread. Both the state and federal governments have spent considerable sums of money investigating and flushing various portions of the mine with fly ash to protect homes in the area. Active subsidence beneath a highway crossing the mine fire is a continual maintenance problem.

About 70 of the 120 acres were mapped during June, 1988, using magnetic surveying. Twenty five acres were remapped in June, 1989. The purpose of the mapping was to determine the magnetometer's ability to define three features associated with the fire. These include 1) the distribution of the

of the areas which had burned and cooled, 2) the actual fire locations, and 3) areas within the mine which were heating up and those which were cooling down.

Total magnetic field readings were recorded on a 25 by 50 foot grid. After correction for diurnal variation, the readings were plotted and contoured. Based on visual observation, areas of the mine along the crop line which had burned and cooled resulted in positive magnetic anomalies. Total field intensities of more than 700 gammas above background levels were recorded for these areas. The contour plots show a continuance of the ash into the mine workings. Under a maximum cover of 120 feet the recorded field intensities were still 280 gammas above background levels. Magnetic modeling was used to define the limits of the burned areas within the workings. Figure 1 shows the positive magnetic anomalies superimposed over the deep mine workings within the study area. The areas interpreted to have burned and cooled to ash are shaded. When determining the limits, it was assumed that a portion of the carbonaceous shale roof rock also burned.

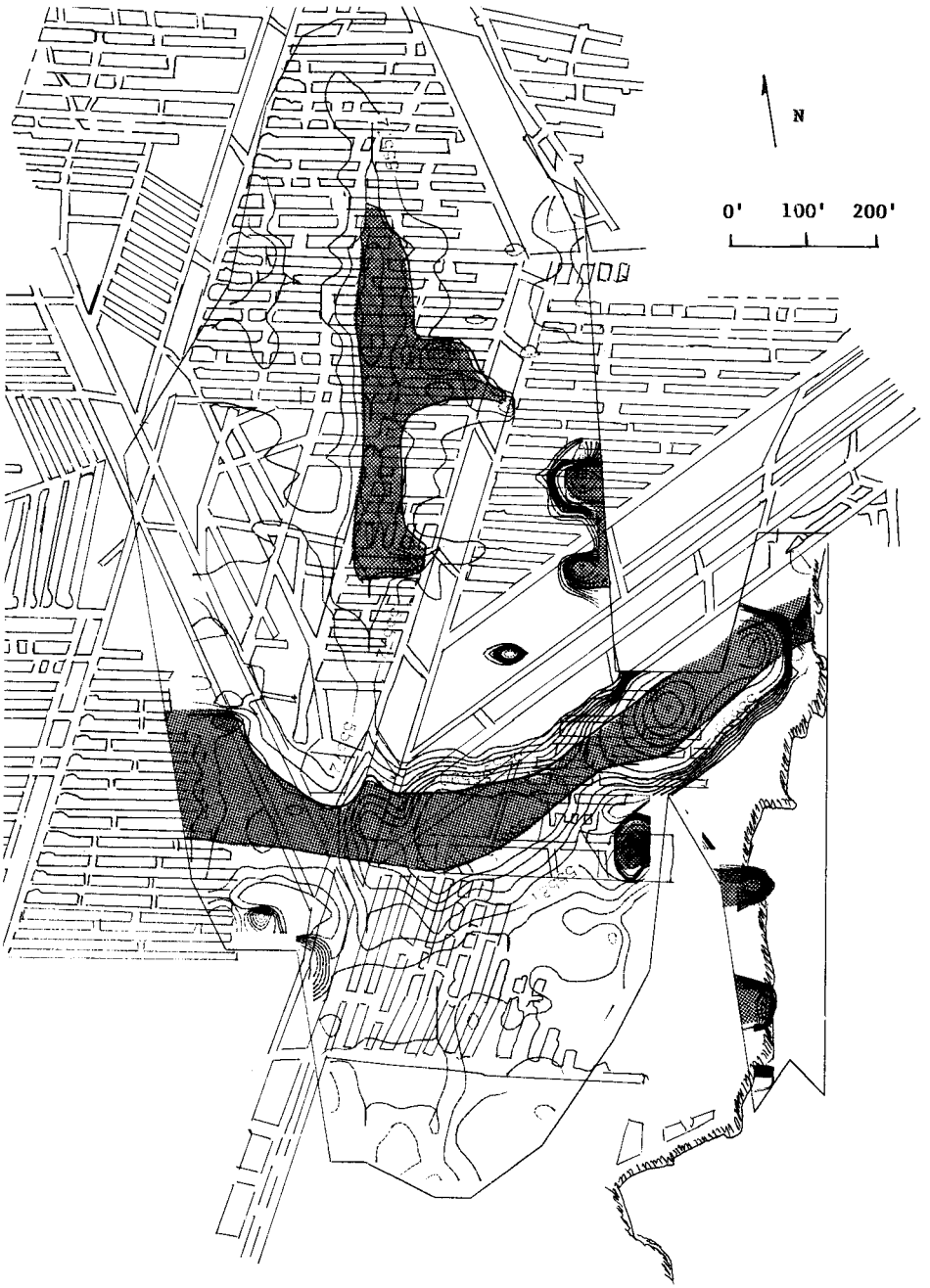



FIGURE 1. DMF1 POSITIVE MAGNETIC ANOMALIES SUPERIMPOSED OVER DEEP MINE WORKINGS. SHADED AREAS SHOW AREAS INTERPRETED TO HAVE BURNED AND COOLED.

The magnetic surveying did not provide a clear definition of any fire zones. Heating to above the Curie temperature results in a loss of natural magnetism. Since sedimentary rocks contain at least traces of magnetic minerals, a local reduction in the total magnetic field in the vicinity of the mine fire would result. The magnetometer used for this study had a sensitivity of only + or - 1 gamma and no recording base station was available. It is possible that a more sensitive magnetometer along with a base station might have been successful in areas of low cover. However, other geophysical methods, especially electrical resistivity, are likely to be more successful in locating fire zones.

On the other hand, the progress of a mine fire might be successfully monitored with magnetics by repeatedly mapping the same area over time. Increases in magnetic intensity would indicate areas which are cooling. Decreases in magnetic intensity would indicate an increase in burning activity. However, any decreases in magnetic intensity are likely to be seen only in areas which were previously partially burned and cooled. These areas would have developed enough magnetic susceptibility that a later decrease would be detectable.

Twenty five acres of the area mapped in 1988 were resurveyed again in June, 1989. The differences between the two sets of readings were plotted and contoured, (Figure 2). Figure 2 outlines the area surveyed in 1988. The areas which have burned and cooled are shaded. Cross hatching shows where the magnetic intensity has increased (cooling zones) and where the magnetic intensity has decreased (increased burning activity). Noted changes in magnetic intensity were slight. This is consistent with the slow movement of many deep mine fires, considering that mapping efforts were only one year apart. However, when using this technique, care must be taken to reoccupy the same grid locations.

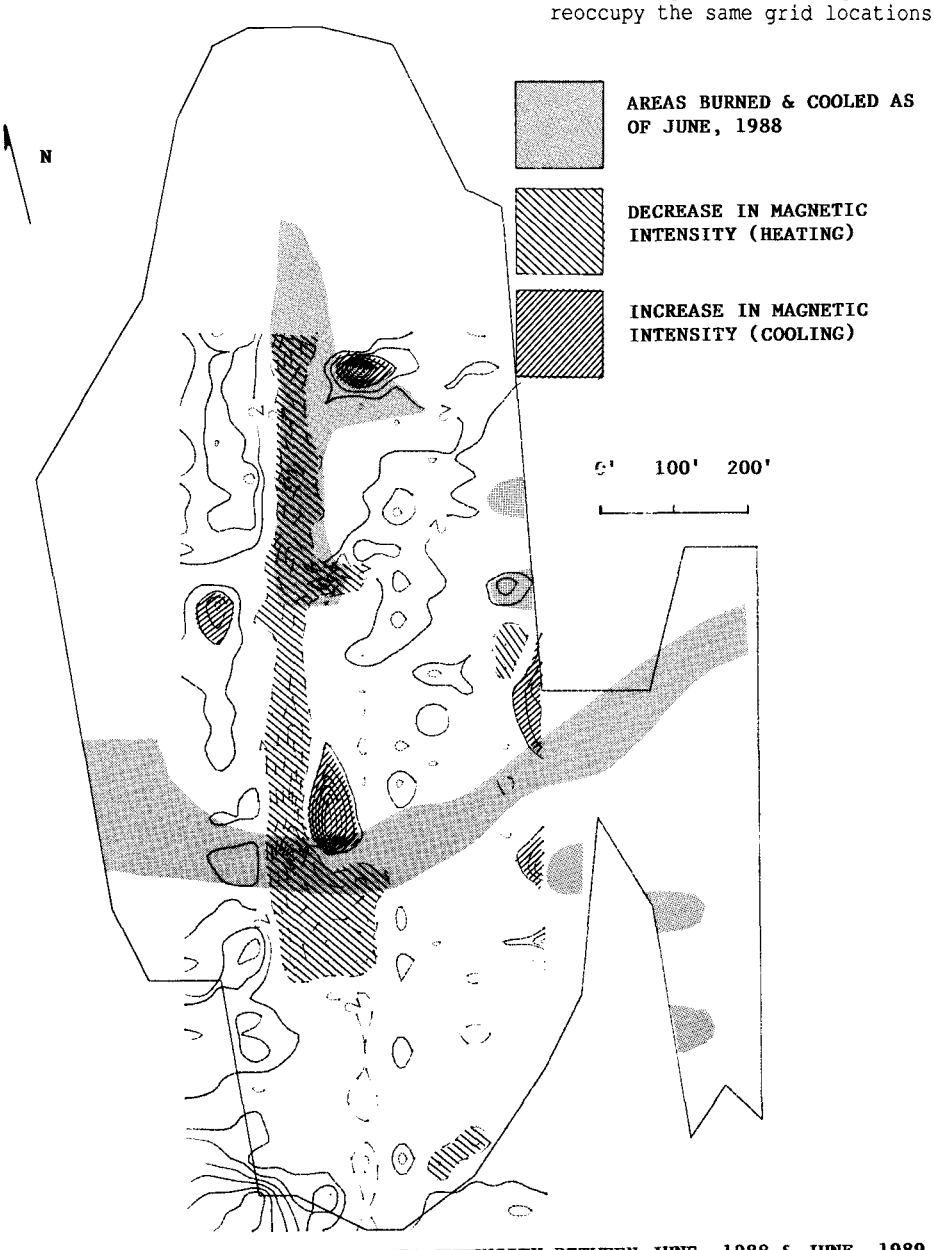



FIGURE 2. DMF1 CHANGES IN MAGNETIC INTENSITY BETWEEN JUNE, 1988 & JUNE, 1989

## A Burning Coal Refuse Bank

Combustion metamorphism of pyrite provides for magnetic mapping of burning refuse piles in the same way as is does for mapping deep mine fires. The primary difference is that combustion in an underground mine fire is limited to the deep mine and overlying or underlying strata. Combustion can occur anywhere within a refuse pile and determining the combustion depths becomes more difficult. Piles higher than 100 feet high are common.

There are also differences in the characteristics of fire propagation. Oxygen is more limited in a refuse pile than in a deep mine, thus burning occurs at a slower rate. Combustion zone temperatures may also be lower. This would ultimately limit the maximum magnetic susceptibilities realized. In a refuse bank the tendency is for the fire zone to be contiguous with the burned zone. Fire in an underground mine can break out anywhere conditions are right.

A five acre refuse bank, designated SWRB1, was mapped in Fayette County using magnetics during May, 1988. This pile occupies the south side and the head of a hollow and is about 160 feet high. Patches of ash, termed red dog, are visible on the outslopes. Venting vapors and noxious fumes were visible in one area along the side of the hollow during the mapping.

The purpose of the mapping was to determine the ability of the magnetometer to locate zones which had burned and cooled. The premise was, that any active combustion zones would be adjacent to these areas. The refuse bank was crossed by an overhead high voltage line which interfered with readings in the center of the study area. Thus, the pile was surveyed in two segments. Readings were obtained along a ten by ten foot grid on this site. Neither a recording base station nor an additional sensor were available for this mapping.

The positive anomalies resulting from the magnetic surveying of the western portion of the bank are shown in Figure 3. Maximum field intensities recorded were 260 gammas above background. The positive magnetic anomaly in the lower left portion is the one of primary interest. This is the area which was venting during the investigation.

Theoretically, the portion of the combustion zone which had burned and cooled would be beneath the positive anomaly although its size would be slightly smaller. Any active combustion should be along the perimeter of this anomaly. A drilling project which occurred nine months before the mapping shows this to be the case.

The site was drilled in June, 1987, under the direction of the Pennsylvania Department of Environmental Resources Bureau of Abandoned Mine Reclamation. Drill hole locations are shown in Figure 3 with an *. Temperatures were recorded in degrees Fahrenheit for each hole. Maximum temperatures recorded for each hole are plotted next to the drill hole location. Note that the highest recorded temperatures are along the perimeter of the magnetic anomaly, especially to the east and west. Temperatures toward the center and beyond the perimeter of the anomaly are substantially lower.

Additional supporting evidence is the relative amounts of ash, or red dog, encountered. The drill holes located within the positive anomaly penetrated intervals where red dog was the predominant material. Drill holes along and beyond the perimeter only encountered traces of red dog. Thus the positive magnetic anomaly is located above the area of the most complete combustion.

There has been no attempt to determine the depths of the burned zones through magnetic modeling. A second sensor should be added and the

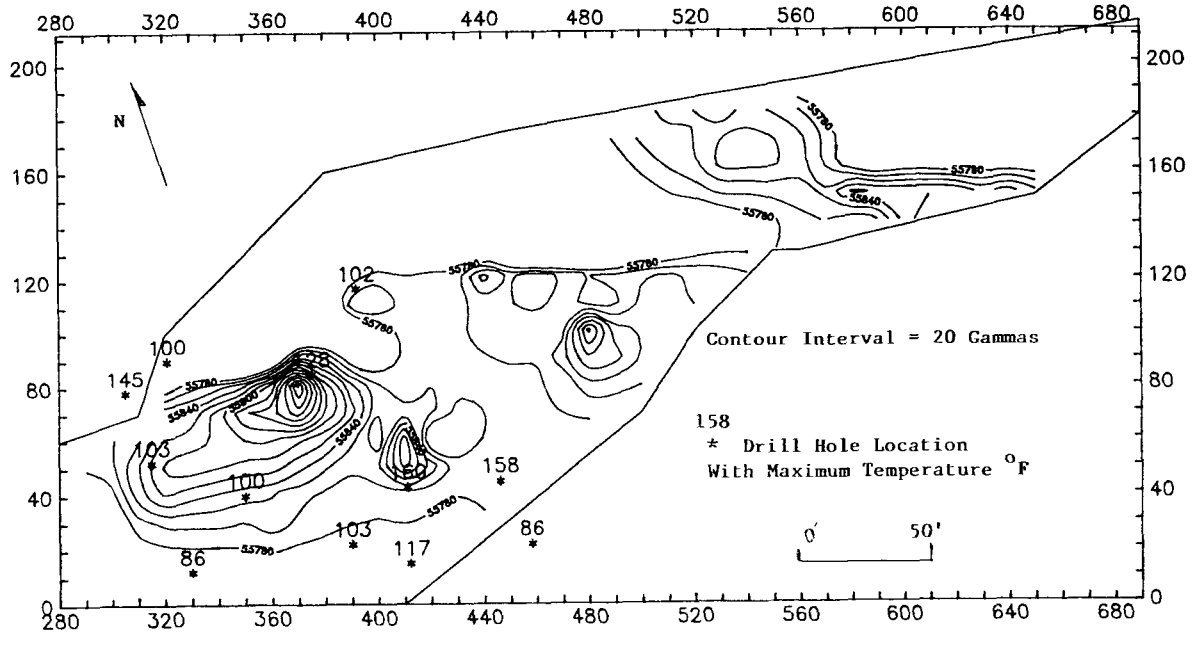



FIGURE 3. POSITIVE MAGNETIC ANOMALIES RECORDED AT SWRB1.

magnetometer used as a gradiometer for confidence in modeling at this type of site. A recording base station would also be helpful.

## Acid Mine Drainage Production Site

A 37 acre reclaimed strip mine in north central Pennsylvania was mapped in April, 1989, using a combination of magnetic surveying and electromagnetic terrain conductivity. The main portion of the site lies to the north and consists of about 29 acres of reclaimed spoil. A southern and an eastern lobe area are also present, each about four acres in size. Acid mine drainage from this site has destroyed five miles of trout stream. The purpose of the mapping was to locate AMD production source areas and to define the resultant pollutional plume. The ultimate goal for this site is to design and implement an AMD abatement plan.

The mining at this site, designated as NCPA1, was completed in 1977. The site exhibited rather uniform overburden thickness and composition, according to the drill logs and persons familiar with the site. The Lower Kittanning coal seam was present in two splits separated by 10' of clay. Only the upper split was mined which left behind a thick underclay as a pavement. Based on drill logs and discharge locations, the pavement dips SSE. The coal was overlain by 17' of black shale capped by a sandstone unit. This is a mountain top site and the only source of groundwater is infiltrating precipitation.

Two areas of perennial seepage occur. These are located to the south of the southern and eastern lobes. The average quality of the discharges is as follows: pH 2.5, acidity 3260 mg/l, iron 330 mg/l, aluminum 350 mg/l, and sulfates 3750 mg/l.

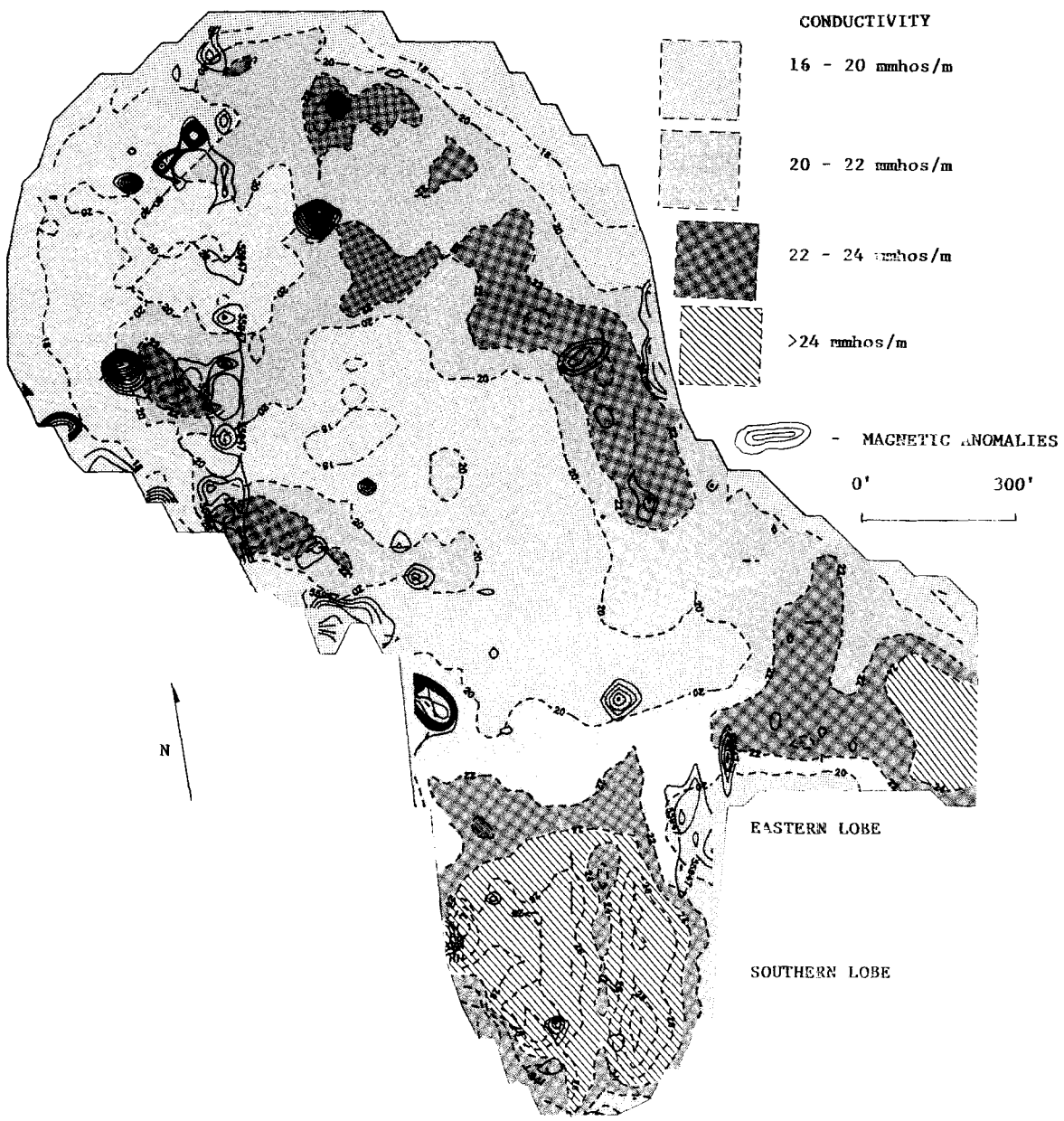



FIGURE 4. POSITIVE MAGNETIC ANOMALIES AT NCPA1 SUPERIMPOSED ON EM CONTOUR PLOT

A 25 by 25 foot grid was set up across the site. Readings were taken at each grid point during both the magnetic and EM surveys. Personnel from the US Bureau of Mines helped with the mapping.

After correction for diurnal variation, the magnetic data was plotted and contoured. Twenty magnetic anomalies were located during the magnetometer survey over the 37 acre site. The magnetic anomalies were superimposed on the terrain conductivity mapping and are presented in Figure 4. Most of the magnetic anomalies were small, indicating piles of pyritic material less than 50 feet across.

The maximum recorded field intensities of the anomalies fell into two distinct ranges, suggesting two different material types. A third of the anomalies were quite obvious even during the field mapping. Their maximum amplitudes ranged from 60 to 130 gammas above background and steep gradients indicated shallow burial. Most of these were located in the northwestern portion of the mine site which had been stripped before 1966. The rest of the anomalies were subtle with maximum intensities of only 5 to 20 gammas above background. Flat gradients indicated deeper burial. Most of these were located in the eastern and southern portions of the mine site which had been mined in the late 1970's (Figure 4).

A test pit excavated by a backhoe into one of the anomalies of low magnetic intensity (20 gammas above background) found pit cleanings to be the source of the altered pyritic material. The other anomalies (60 to 180 gammas above background) most likely result from tipple refuse. One person familiar with the mining during the 1970's indicated several truckloads of refuse from a nearby tipple were buried in the northwestern portion of the mine. His recollection of the dumping locations coincided with the mapped locations of the anomalies. Also, the source tipple for the refuse used magnetite in the separation process. Magnetite lost to the refuse would account for the higher magnetic intensities of these anomalies. A drilling program is planned for April, 1990. The drilling will be used to define the composition of all the anomalies and to install monitoring wells in the northern portion of the site.

Several of these anomalies were mapped on a 5 by 10 or a 10 by 10 foot grid with the magnetometer being used as a gradiometer. Using the magnetometer in the gradiometer mode and the closer grid spacing provides much greater detail about the target material. Using this information and the modeling program, reliable estimates as to the depth of burial and the lateral extent of the piles can be made.

The terrain conductivity data was obtained at the same grid locations as the magnetometry data. The conductivity values were contoured so the two surveys could be compared. This site was originally mapped by Ken Ladwig of the US Bureau of Mines in 1982, as part of a joint effort with Woodward-Clyde Consultants(Ladwig 1982). The results of the 1989 mapping compared favorably with Ladwig's results.

The terrain conductivity readings were fairly uniform across the northern portion of the site with a gradual increase toward the ESE, or down-dip direction. High conductivity readings were recorded in both the eastern and southern lobes.

The site was drilled shortly after Ladwig's mapping effort. The high conductivity zones in the two lobes are attributable to impounded water above the pavement. Intact monitoring wells in the two lobes indicate that this condition continues to exist. Virtually no pooled water above the pavement was found in the northern portion of the mine (Ladwig 1982).

Note in Figure 4 that, in almost every case, a magnetic anomaly (pyritic material) is adjacent to a conductivity high. The relationship between the terrain conductivity and the magnetic anomalies is a strong indication that the buried piles of pyritic material are prime sources of AMD formation. AMD is generated when infiltrating precipitation comes into contact with the oxidation products of pyrite. AMD, an excellent electrolyte, exiting the pile is the reason for the conductivity high in the immediate vicinity of the pile. After the water exits the pile, it spreads and is diluted. The recorded conductivities decrease. Overall, conductivity values increase downgradient as the water exiting several piles gradually merge, ultimately exiting as polluted toe-of-spoil discharges from the two lobes.

## Summary and Conclusions

A proton processing magnetometer was used in an attempt to delineate fires and burnt areas within an underground coal mine and a coal refuse bank. The very small negative magnetic anomalies associated with fires makes detection of fire areas difficult. However, areas that have burnt and cooled create significant positive magnetic anomalies due to increased magnetic susceptibility. Magnetic surveying can readily detect and delineate these areas even at significant depths. The magnetometer also offers promise as a means of determining the progression of a mine fire. This is done by repeatedly mapping an area and determining the changes in magnetic intensity through time.

The magnetometer along with terrain conductivity has been used on a surface mine that is producing severe AMD pollution. Several positive magnetic anomalies were recorded. These were interpreted to be areas of concentrated pyrite oxidation products (altered to magnetic minerals). These anomalies were interpreted to be buried piles of tipple refuse or pit cleanings. Areas of high terrain conductivity were mapped adjacent to these piles. High EM conductivity on this site results from elevated dissolved solids in the groundwater due to AMD production. Thus, EM and magnetometry can be used together to identify sources of AMD production. This allows for potential remedial efforts to ameliorate the pollution problem.

## Acknowledgments

I would like to extend sincere thanks to my co-workers Dale R. Thompson, Michael Klimkos, and Dr. Charles Miller, for their assistance with the field work. I would also like to thank Terry Ackman, Mining Engineer, and Craig Hustwit, Civil Engineer for their active participation in the

field mapping, use of their equipment, and an open exchange of ideas and opinions. Both are employed by the U.S. Bureau of Mines, Pittsburgh Research Center.

## References Cited


Breiner, S. 1973. Applications Manual for Portable Magnetometer. Geometrics. Sunnydale, CA, 57p.

Chaiken, Robert F., Louis E. Dalverny, Ann G Kim, 1989. Calamity Hollow Mine Fire Project 2.Operation of the Burnout Control System, U.S. Bureau of Mines, Pittsburgh, Pa. Preface.

Cisowski, S.M., M. Fuller, 1987. The Generation of Magnetic Anomalies by Combustion Metamorphism of Sedimentary Rock, and Its Significance to Hydrocarbon Exploration, Geological Society of America Bulletin 99:21-29.

Henderson, Roger. 1984. Direct Indication of Hydrocarbons from Airborne Magnetics, Exploration Geophysics 15:213-219.

Irani, Meherwan C., Robert F. Chaiken, Louis E. Dalverney, Gregory M. Molinda, Karen E Soroka, 1983. Calamity Hollow Mine Fire Project 1.Development and Construction of the Burnout Control Ventilation System, p.2. U.S. Bureau of Mines, Pittsburgh, Pa.

Justin, Thomas R., Ann G. Kim, 1988. Mine Fire Diagnostics to Locate and Monitor Abandoned Mine Fires p. 348-355 in: Proceedings of 1988 Mine Drainage and Surface Mine Reclamation Conference. (Pittsburgh, PA)

Ladwig, K.J., 1982. Delineation of Zones of Acid Mine Drainage Using Surface Geophysics p. 279-287 in: Proceedings of 1982 Symposium of Surface Mining, Hydrology, Sedimentology, and Reclamation. (Lexington, KY)

McNeill, J.D. 1980. Electromagnetic Terrain Conductivity Measurements at Low Induction Numbers. Technical Note 6, Geonics Limited, Mississauga, Ontario, Canada.

Pennsylvania Department of Environmental Resources, 1971. Programs and Planning for the Management of the Water Resources of Pennsylvania p.275. PA D.E.R., Harrisburg, Pa.

Schueck, J.H. 1988. Mapping Buried Tipple Refuse - Is The Magnetometer Better Than Terrain Conductivity? p.117-130 in: Proceedings of 1988 Mine Drainage and Surface Mine Reclamation Conference. (Pittsburgh, PA)

This page intentionally left blank.

Locating and Abating Sources of Acid Mine Drainage at Reclaimed Strip Mines. Terry E. Ackman, Mining Engineer, U.S. Bureau of Mines, Pittsburgh Research Center, Pittsburgh, PA.

Surface and ground water pollution is a common problem associated with post-surface mining operations and can lead to many costly problems such as: (1) a long-term treatment commitment; (2) sludge disposal: (3) re-excavation and relocation of acid-producing materials; (4) forfeiture of bond monies; and (5) bankruptcy. A reclaimed strip mine in Upshur County, WV is being studied with the objective of developing an effective, economical and permanent method that abates post-mining water pollution problems. Historical water quality, terrain conductivity and flow data from this site, dating back to 1983, supplement more recent hydrologic baseline data. Magnetometry, which has recently been proven capable of locating some types of acid-producing materials by means of magnetic anomalies, was used to target grouting efforts. A thin cement and fly ash slurry (density of about 87 lbs/cu.ft) was injected under pressure into bore holes as means of infusing the entire target zone. The magnetic properties associated with the fly ash also served as a tracer for identifying slurry propagation through the use of post-grouting magnetometry surveys. As the result, the grouted zones of acid-producing materials were transformed into a relatively impermeable and inert mass of sub-surface material. The long-term water quality monitoring of the existing 12 wells and surface seep will be used to evaluate the effectiveness of this water pollution abatement technique.

ADDITIONAL KEY WORDS: Magnetometry, Terrain Conductivity, Fly ash, Cement

This page intentionally left blank.

DISTRIBUTION AND CHARACTERISTICS OF OUTCROP FIRES IN HORIZONTAL STRATA IN EASTERN AND WESTERN COAL PROVINCES. Bernard R. Maynard, Office of Surface Mining Reclamation and Enforcement (OSM), Eastern Field Operations, Technical Assistance Division, Pittsburgh, PA 15220.

Field evaluations of 93 underground mine fires in the Eastern and Western Coal Provinces on the National Abandoned Mine Land Inventory (NAMLI) show that 74 are bituminous coal fires. Forty (54%) of the 74 bituminous fires are near-surface burning seams or outcrops (as opposed to mine fires burning at depth). Eastern Coal Province outcrop fires number 23 (58%), while the Western Province contains 17 (42%) outcrop fires. Similar fire characteristics are found in both coal provinces, including: 1) propagation of fires relative to valley shape, orientation and the direction of prevailing winds; 2) venting fissures extend to the surface, forming parallel to the contour and developing in response to stress release along existing fracture systems; 3) venting intensity appears to increase along the highest fissures as ventilation develops relative to strata conditions and the existing fracture system; 4) non-contiguous zones of combustion occur along the length of the fire; and, 5) vegetational delineation above fire zones appear related to the tolerance of selected species within climatic regimes. Diagnostic procedures have not been developed to identify non-contiguous combustion segments and the depth into the seam that combustion occurs. Applied research during abatement projects could characterize and quantify outcrop fires. Innovative, cost-effective techniques to improve existing control methods are needed. One recently adapted technology utilizes foam concentrates such as those used in fighting forest fires.

This page intentionally left blank.

# ABANDONED MINE LAND RECLAMATION - UNIQUE PROJECTS IN WEST VIRGINIA[1]

by

Neil A. Robinson[2]

Abstract. The Abandoned Mine Lands Program (AML) in West Virginia has completed reclamation on 215 projects at a total construction cost of $90 million. Typical projects encountered daily by the AML staff including mine portal sealing, coal refuse pile regrading and extinguishment (if burning), landslide correction, subsidence abatement, and mine drainage control. Some of these projects require a different reclamation approach than is commonly used on typical AML projects. This paper provides information concerning the abatement and correction of several AML projects that required different approaches. One site consisted of a burning coal refuse pile covered with asbestos which had to be handled and disposed of in a specific manner. The use of a coarse sand aggregate was utilized to extinguish a deep mine seam fire on a project in Kistler, West Virginia. Two projects required special water handling and treatment facilities. Transforming a hazardous burning coal refuse pile into an extention of a state park campground facility was the reclamation objective of one AML site. Another project involved correcting a subsidence problem with steel rod reinforcement.

Additional key words: Abandoned Mine Land Reclamation, mine portal, coal refuse pile, mine fire, subsidence and acid mine drainage.

## Introduction

The purpose of this paper is to provide information on West Virginia AML projects which required unique or unconventional reclamation Techniques. Visual observation of the original problem area does not always provide the information necessary to determine what reclamation is required to eliminate the associated hazards. Detailed geotechnical investigations occasionally reveal a completely different situation that what is actually occurring. Once the actual situation is understood, the reclamation design must often be approached from a different angle. New reclamation techniques also have to be considered when developing a plan of action.

## Materials and Methods

### Iroquois Burning Coal Refuse Pile

A burning coal refuse pile, covering approximately eight acres in southern West Virginia, was the site of an experimental extinguishment project conducted by the West Virginia Air Pollution Control Commission between 1966 and 1968. Various mixtures of petroleum asphalt, water, sand, flyash and asbestos cement were applied to the surface of the burning pile in an attempt to eliminate the flow of oxygen to the fire. Temperature reductions throughout the pile were noted after the sealer was applied, but continuous settling of the pile cracked the sealer opening the pile to oxygen.

In the summer of 1988, West Virginia AML personnel investigated the burning pile to evaluate extinguishment and reclamation alternatives. The burning area was delineated during a subsurface investigation and samples of the asbestos material

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, held in Charleston, West Virginia, April 23-26, 1990.

[2]N.A. Robinson is a Project Manager for the WVDOE, AML&R, Charleston, WV 25311.

were taken to determine if special handling would be necessary. The material was found to contain 5% chrysotile (a form of asbestos) which indicated that the material would require special handling and disposal.

Despite numerous conversations with the Environmental Protection Agency (EPA), it could not be determined which EPA program regulates the removal of asbestos from a coal refuse pile. Since EPA did not impose any additional regulations, only OSHA requirements needed to be followed for the removal and disposal of the asbestos materials.

Kistler Mine Fire

During a field investigation of what was thought to be a burning coal refuse pile, located in the town of Kistler, W.Va. it was noticed that smoke was venting along the coal outcrop beyond the small burning pile. The refuse pile fire had ignited the abandoned coal mine at the outcrop and subsequently seven acres of deep mine workings were burning. As the fire burned from the outcrop back into the mine, the overburden rock subsided leaving large fractures in the ground surface. Heat, sparks or flames moving up through these fractures ignited several forest fires threatening the safety of the Kistler residents.

Four reclamation alternatives were examined for mine fire control and extinguishment. The first option considered utilized a combination of a flush barrier and total excavation of the mine fire. The flush barrier would limit the advancement of the fire during the total excavation process. This option would guarantee total extinguishment of the fire but would leave a 450 foot highwall above the town and would be cost prohibitive.

The second option discussed was referred to as bench seal and stream diversion. This method consisted of sealing the coal outcrop and diverting a nearby stream into the mine workings to extinguish the fire. There were many inherent problems associated with this method. To extinguish the fire, water would have to be impounded in the mine in order to come in contact with the roof where most of the fire is typically located. Impounding of water would create the hazard of an uncontrolled blowout which could flood the residents of Kistler.

An attempt to eliminate oxygen from fueling the fire was the basis for the third option-bench seal and flush barrier. A flush barrier would be constructed as in option one in conjunction with sealing off the coal outcrop from oxygen inflow. This option was rejected due to the fact that it was not possible to locate and seal off all sources of air flow into the mine. The fire would continue to burn until all coal reserves are depleted.

The method selected for reclamation of the mine fire was total blind flushing. Blind flushing is a method of filling the void with an inert material through injection boreholes. The locations of the injection holes were determined by overlying the deep mine map onto a 100 scale topographic map of the project area. Material used to fill the mine void consisted of a coarse sand and water slurry. The objective was to fill the entire mine void with the slurry material, thus extinguishing the fire.

Higgenbotham Subsidence

The Higgenbotham residence in northern West Virginia was experiencing severe structural distress. Concrete block basement walls contained several vertical and horizontal cracks from hairline up to 1/2 inch in width. The family room floor had sunk at one end and the interior doors would not close.

An abandoned deep mine causing the structural damage was located approximately 385 feet below the house. An exploratory borehole was drilled next to the Higgenbotham residence to the mine level. Results of this drilling provided evidence of subsidence due to the highly fractured geologic material that the borehole encountered.

Due to the depth of the mine workings, stabilization by injection grouting would not have been cost effective. There would have been little control over grout flow. Additionally, the effectiveness of the stabilization process could not be monitored as it could be at shallow depths with a borehole camera. It also would have been expensive to have to repeatedly drill a 385 foot borehole when the mine void was not encountered.

The method used to stabilize the Higgenbotham home was referred to as fractured rock reinforcing. With this method an attempt is made to tie the fractured rock strata together so that it will respond as a single unit during a subsidence event. This design does not attempt to eliminate settlement but it does eliminate abrupt differential settlement that results in structural damage due to mine subsidence.

Six rows of 4 inch diameter holes were drilled on either side of the house at spacings of 12 feet between rows and 3 feet between holes. The holes were angle drilled beneath the house ranging from 4.5 degrees to 51 degrees from vertical forming an interlocking pattern (see Figure 1). The bottom elevations of all holes were the same. Hole depth varied from 50 to 90 feet depending on the ground surface elevation from which it was drilled. Number 10 deformed reinforcement

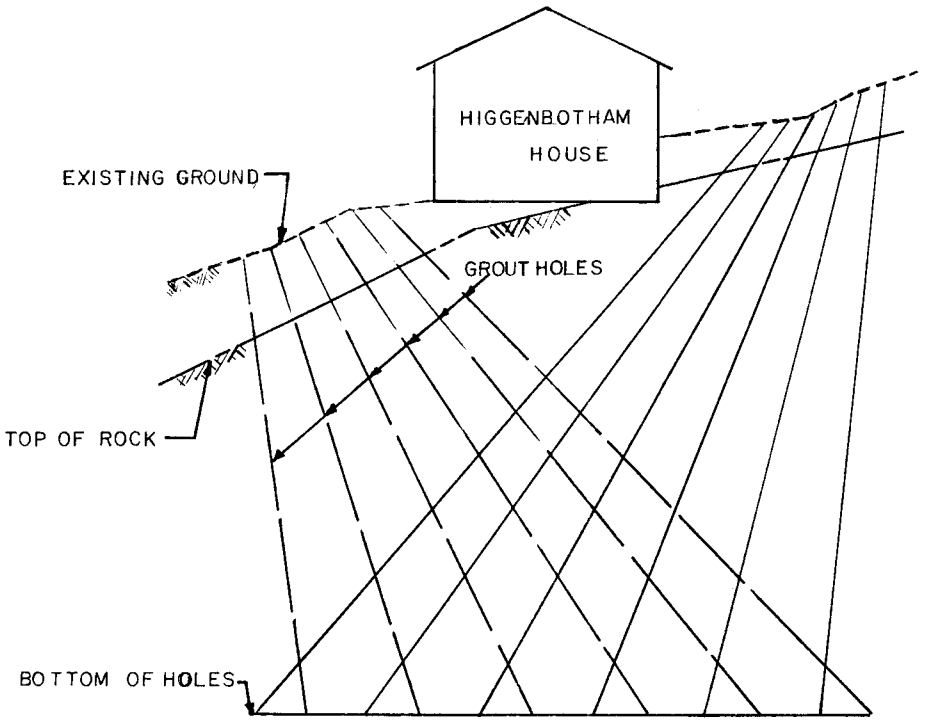



Figure 1. Higgenbotham grout borehole locations.

bars (rebars) with centralizer were placed in the hole. A grout mixture was then pumped to fill the holes around the rebar. Extensometers were installed prior to the start of construction to monitor ground movements, before, during and after the work conducted at this site.

Meriden Drainage

Mine drainage (583 GPM) from what was assumed to be a collapsed mine entry was causing saturated and unstable conditions on a mountainside above a county road along the Tygart River. Preliminary research of existing mine maps indicated that the mine drainage was associated with the Middle Kittanning coal seam.

An exploratory drilling program was initiated to determine the amount of water impounded in the mine and the magnitude of hydraulic head buildup. After several attempts to intercept the mine void failed, questions began to arise as to the origin of the drainage. The decision was then made to advance another borehole an additional 45 feet below the Middle Kittanning to the Lower Kittanning coal seam. The rock strata between the seams was found to be highly fractured. A collapsed void was encountered at the Lower Kittanning elevation. When the mine void was intercepted by the drill bit, the water level in the borehole rose 93.5 feet indicating the presence of an extreme hydraulic head buildup. The Lower Kittanning coal seam dipped below the Tygart River, which was recharging the subsided mine. The river recharge in combination with mine water caused the water level to rise up through the fractured rock into the Middle Kittanning coal seam and seeping out onto the mountainside.

Initial reclamation plans were to wet seal the draining mine entry and regrade the sliding material. Results from the drilling program indicated that the initial plan would have to be significantly revised. Reclamation of this site consisted of installing a deep trench drain down to the base of the Middle Kittanning coal seam to collect all water being forced up from below. Large diameter relief wells were to be drilled to the Lower Kittanning mine to relieve the hydraulic head.

Typical quality of water from the Kittanning coal seams is some of the most acidic of all coal seams in this region. Water quality samples taken along the county road revealed a pH of 6.3, sulfates at 760 mg/l, iron at 23 mg/l and alkalinity at 35 mg/l. Further investigation into the reasons for relatively good water quality than that which is normally found indicated that the water is flowing through fractured and subsided alkaline shales between the Lower and Middle

Kittanning coal seams. Due to this discovery, the large diameter wells were eliminated so that the water would come in contact with the alkaline material, thus improving water quality.

Webster Refuse

The Webster Refuse site consisted of approximately 1800 feet of highwall, five acres of coal refuse, and three collapsed deep mine openings discharging roughly 53 gpm of acid mine drainage into Webster Run. The quality of the drainage was pH 3.8, iron 1.8 mg/1, aluminum 6.2 mg/1, manganese 2.0 mg/1 sulfate 800 mg/1 and alkalinity 0 mg/1.

Any increase in the water quality of Webster Run would have a significant effect on the receiving stream. The improvement of the Webster Refuse water quality would greatly improve the productivity of the receiving stream.

Along with using typical reclamation techniques the reclamation design of this project emphasized an experimental attempt to improve water quality. The entire highwall, including all of the exposed coal seam and pits, were backfilled with existing spoil material to eliminate any direct contact of surface drainage with the coal seam and acid producing spoil. All acid producing coal refuse was covered with one foot of topsoil obtained from a borrow area. A sediment control pond was constructed as the soil borrow was removed.

Wet mine seals were installed in the draining mine entries. During construction the drainage was treated in-line with soda ash. Upon completion of regrading and establishment of permanent vegetation, the sediment pond was converted into an alkaline leach bed. The principle spillway of the sediment pond was perforated all around so that the drainage into the pond would immediately flow out rather than having to reach the top of the spillway pipe. Limestone aggregate (AASHTO No. 2) was used to fill the leach bed. The pipeline from the wet seals to the leach bed was connected to a 12" PVC pipe header and then into three 6" perforated PVC pipes lying on top of the limestone. The three 6" pipes functioned to disperse the acid mine drainage across the top of the limestone. As the drainage filtered down through the limestone it picked up the alkalinity needed to increase water quality. Once the drainage reached the bottom of the leach bed; it flowed through the perforated principal spillway and then into Webster Run.

Chief Logan State Park

This abandoned mine land project was located within the boundaries of Chief Logan State Park in southern West Virginia. The site consisted of several coal refuse disposal areas encompassing approximately 30 acres. A portion of this refuse was burning and contained dangerous underground voids. Smoke from the burning areas often fill the narrow valleys of the park. Acid mine drainage from the old underground workings and sediment from the refuse piles had greatly deteriorated the quality of the receiving stream. Collapsed mine entries with pooled water created a potential for hydraulic head buildup and subsequent blowout. In short, the entire area represented a hazardous and environmentally degraded condition within the confines of a highly used and popular recreation facility.

The project area could have been successfully reclaimed to improve the environment and eliminate the hazards with a minimal planning and construction effort. The unique potential of the site and its location, however, led abandoned mine land personnel to initiate a joint project with the West Virginia Department of Commerce, Parks Division. Funding for the project was secured through the National Abandoned Mine Lands Fund, state appropriations, and Soil and Water Conservation Funds. The construction activities on the site not only resulted in complete reclamation but was further developed to include the establishment of a 25-site campground and associated recreational facilities.

## Results and Discussion

Iroquois Burning Coal Refuse Pile

Reclamation proceded initially with the removal of the asbestos - containing material. The asbestos material was removed by hand, thus the workers were required to wear disposal coveralls, respirators, rubber boots and gloves. Air sampling instruments were placed on the workers to monitor their exposure to asbestos fibers during the removal process. If the asbestos levels in these samples would have exceeded 0.2 fibers/cc, it would have been necessary to set up a decontamination facility where each worker could be processed before leaving the site. Asbestos levels measured on the workers never exceeded 0.04 fibers/cc, thus the decontamination facility was not needed.

Since the asbestos material could be seen on the surface of the refuse pile as a thin, broken, bituminous layer, the most efficient method of removing the material was by raking and shoveling. The material was continually kept moist to reduce the amount of asbestos released into the air. A dump trailer lined with a 6 mil plastic bed liner was kept on-site to haul the asbestos material to a landfill approved to receive such materials.

Once all asbestos material was removed from the site, the Contractor began to excavate and extinguish the

burning coal refuse. Extinguishment was achieved by excavating the burning material, spreading it out and applying water. Quenched material was allowed to dry and then incorporated back into the fill. A subsurface investigation, conducted during project design delineated the limits of burning. When the Contractor reached these limits, burning temperatures were found existing beyond these limits. To ensure that all burning was completely extinguished, excavation continued beyond the initial limits.

Excavation quantities were initially estimated at 73,886 cubic yards. Due to the extent of the burning the entire pile had to be excavated which increased the quantity to 278,000 cubic yards. The significant underestimate in the quantity of burning material occurred during the subsurface investigation. The boreholes that were advanced into the pile to determine the extent of burning encountered large heat fused masses of reddog, known as "clinkers". These clinkers were misinterpreted as original ground, thus the boring was terminated. During construction it was found that there was a large amount of burning material below the clinkers. It has also been found that coal refuse fires can increase or decrease in size and intensity in the time interval between exploratory drilling and construction. This makes it extremely difficult to try to estimate the volume of burning for competitive bidding purposes. Reclamation of the Iroquois Burning Coal Refuse Pile was completed within a ten month time period at a total cost of $1,179,734.09.

Kistler Mine Fire

Construction proceeded on the Kistler Mine Fire for approximately one year. At project completion a total of 145 boreholes had been drilled at a total linear footage of 23,292. The boreholes were broken down into injection holes and assurance holes. A total of 38,385 tons of coarse sand aggregate was injected into the mine through the injection holes. Assurance holes were installed to determine the relative success of the injection material filling the mine with coarse sand slurry or to monitor mine level temperatures. Temperature readings taken upon completion of the project revealed that there were three small areas that were still burning. These hot spots caused little concern because they were each surrounded by inert injection slurry which would eliminate the fire from spreading. Subsequent monitoring of assurance holes since completion of reclamation has indicated a slow decline in temperature. Total cost of reclamation for the Kistler Mine Fire project was $1,585,124.41.

Higgenbotham Subsidence

Rock reinforcing was completed in August 1989 after approximately 90 days of construction at a total cost of $99,500.00. Construction was slowed due to rainy weather and encountering water in some boreholes at the 40 foot depth. Water in the boreholes was remedied by grouting the boreholes from the bottom and displacing the water.

According to reports, no additional structural distress to the Higgenbotham residence have been observed. Therefore, it is assumed that the reinforcement technique was successful.

Meriden Drainage

After 240 days of work, reclamation was completed in July 1989 at a construction cost of $701,673.46. At the present time all drainage is being collected in the trench drains and conveyed to the river in riprap channels. No seepage can be detected in the final regrade and the slope has been stabilized.

Webster Refuse

An initial concern with the leach bed technology was that within a relatively short period of time the limestone would coat with iron precipitate rendering it ineffective for water treatment. However, the iron levels in this drainage were only 1.8 mg/1. Thus the chances of complete limestone coating would require a considerable time period. As the drainage flowed across the aggregate and the limestone decomposed a minimal amount of iron coating hopefully would be broken up and fresh reactive surfaces of limestone would be exposed.

The contract specifications for this project stated that if after one year from the date of completion of reclamation water quality tests indicated that the leach bed was no longer functioning to significantly increase water quality it would be removed, the site regraded and drainage carried to the receiving stream by riprap channels. Water quality data was obtained before, during and at one month intervals after the completion of reclamation. A portion of this data is contained in Table 1. Alkalinity and pH values have increased, while iron has decreased.

The major quesiton that remains is how long will the leach bed continue to produce good water quality? This project was selected as the northern West Virginia 1989 Abandoned Mine Land Reclamation Award winner.

Chief Logan State Park

Reclamation of this site included the excavation and extinguishment of the burning coal refuse material. A detailed regrading plan was specifically designed to support the camping facility. A minimum of 12 inches of soil material was used to cover the entire refuse area with 3 feet of soil under the camping areas

Table 1. Water quality data from the Webster Refuse Project after the water was treated by the limestone leach bed.

| Sampling Date | pH | Fe | Al | Mn | SO4 | Alkalinity |
|---|---|---|---|---|---|---|
| | | ----------Mg/1---------- | | | | |
| 04-26-88 | 3.9 | 8.8 | 11.0 | 2.3 | 1000 | 0 |
| 06-15-88 | 3.6 | 31.5 | 16.5 | 4.2 | 1400 | 0 |
| 07-21-88[a] | 7.0 | 0.2 | 0.4 | 2.4 | 600 | 45 |
| 12-01-88 | 7.3 | 0.0 | 0.1 | 0.0 | 1000 | 59 |
| 04-27-89 | 6.7 | 0.1 | 0.1 | 0.1 | 1000 | 131 |
| 09-05-89 | 6.6 | 0.0 | 0.3 | 0.0 | 1260 | 138 |

[a] Date on which acid mine drainage was routed through the limestone leach bed.

and facility buildings. The camping sites were constructed with paved spurs and level grass pads with water, sanitary, and electrical hookups. Earthen mounds were constructed between campsites to provide the maximum amount of privacy.

Stream channel reconstruction was accomplished to provide environmental improvements throughout the site. The stream channel before reclamation flowed through and eroded the acidic refuse material, contributing large amounts of sediment and acid loads to the stream. Improvements to the channel included lining the regraded channel with an impervious clay material and using filter fabric and rock riprap to stabilize the stream. The impervious channel lining functioned to keep the stream from coming in contact with the acidic coal refuse material.

A borrow area was excavated for both soil and rock which was used as soil cover and channel lining, respectively. Upon completion of borrowing activities, the borrow pit was backfilled and regraded into the configuration of an amphitheater. The amphitheater has been used as a meeting place for interpretive programs.

The drainage from mine entries, located in a side hollow, was causing saturated conditions which posed a threat to people using the park. Many of the entries were also open and accessible to the general public. Sealing these entries was accomplished by using aggregate bulkhead seals with PVC pipe, allowing the entries to drain, thus eliminating hydraulic head buildup. Drainage was diverted around the saturated areas so that these areas could dry and be regraded.

Many of the existing structures left from past mining activities were rehabilitated to provide new uses. At the campground entrance an old railroad trestle was restored and is serving as a roadway bridge crossing the stream. Overlooking the new campground was a coal load out chute which facilitated the movement of mined coal from the mountainside deep mine entries to the valley floor. This chute was upgraded to serve as an overlook platform located off one of the nature trails. A series of haulroads leading from the valley floor up to the mine entries and along the mountainside serve as hiking/nature trails.

The project also included the construction of various facility buildings for the basic functioning of the campground. These buildings included a campground check-in building, wastewater treatment facility, booster pump station and bathhouse.

The project illustrates that future land-use planning is a complimentary addition to reclamation. The emphasis that was placed on future land-use for the Chief Logan State Park project represents the highest goal in the abandoned mine land program: the transformation of useless, hazardous land into useful or productive areas.

# A CASE STUDY
# OF
# A LARGE OPEN PIT URANIUM AML PROJECT
# GAS HILLS, WYOMING[1]


## BY
## Michael Koopman and Ronald Rash[2]



**Abstract:**

The Abandoned Mine Lands Program (AML), authorized under the Surface Mining Control and Reclamation Act of 1977 provides funding for the abatement of health and safety hazards on lands disturbed by mining prior to enactment of the Act. A good example of the implementation of the AML Program in Wyoming is the A-8 Pit. The reclamation site is located in the East Gas Hills Uranium Mining district of Wyoming. Reclamation activities include selective handling of 3.5 million cubic yards of backfill, controlling pit dewatering and water treatment, installing second order drainage channel and riprap control structures, and salvaging sufficient coversoils and topsoils for site revegetation.



**Additional Key Words:**

Abandoned Mine Land Program, Surface Mining Control and Reclamation Act, Wyoming.


## Introduction

The Surface Mining Control and Reclamation Act of 1977 established national standards for conducting coal mining and reclamation activities. In addition, Section IV of the Act created the Abandoned Mine Land (AML) Program which provides a funding mechanism and priority system for the reclamation of disturbed areas before the Act was enacted. The AML Program first encourages the reclamation of areas that pose a serious health and safety hazard to people, and areas with only environmental problems are reclaimed next. Funding for the AML Program is derived by a tax on active coal producers, to be collected from 1978 through 1992. Pending legislation will extend the program and adjust the current levied tax rates for mined coal. The tax is levied at a rate of 35 cents per ton on surface mined coal, 15 cents per ton on underground mined coal, and 10 cents per ton for lignite. At least 50% of the monies collected within each state are returned to that state for reclamation of abandoned mine sites.

Each year, States with a federally approved AML Programs select reclamation projects from their established inventories of eligible sites and submit grant requests to the Federal Government to conduct the necessary work. These requests include detailed descriptions of engineering costs for each project plus the anticipated benefits of reclamation.

In some States, particularly in the West, problems associated with non-coal mines are more severe than those caused by coal mines. Federal approval for the expenditure of AML monies on these lands is possible if the state certifies the presence of threats to public health and safety. Environmental concerns at non-coal mines may be addressed through the program only after all abandoned coal mines in the state have been reclaimed.

Because Wyoming has reclaimed all of its AML coal sites, reclamation of non-coal sites, including open pit uranium mines, is possible. Prior to 1977, spoil and associated ore were usually randomly mixed and stockpiled. These stockpiles generally have side slopes near the angle of repose. Geotechnical data of a few spoil piles in the Gas Hills region indicates silt/clay contents at about 35% and median particle sizes (sediment) of ($D_{50}$) 0.28 mm. Plastic Indexes (PI) are very high at 13-35. These soils are potentially highly erosive. Vegetation is sparse to non-existent. Wind and water borne sediments are transported offsite and impact streams and drainageways. Spoil materials may be rich in oxidized (sulfides) and/or radioactive elements. Reclamation of these sites usually requires grading and backfilling to bury unsuitable materials, and a re-establishment of drainage patterns. Application of specialized and innovative revegetation practices is necessary due to the lack of moisture, poor soils, and extreme weather conditions.

---

[1] Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, April 24-26, 1990, Charleston, West Virginia.

[2] Michael Koopman is an AML Engineering Specialist, employed by the Wyoming Department of Environmental Quality, Cheyenne, Wyoming 82002; Ronald Rash is the Project Manager for the AML 16E-1 project, employed by Forsgren Associates as Area Manager in Casper, Wyoming 82601.

Another problem commonly seen in the Gas Hills region are highwalls resulting from open pit mining. These highwalls, comprised of course grained sandstones, conglomerate and claystone, often several hundred feet high are obvious hazards because of their height, steepness, and instability. There are often water impoundments left in the bottoms of these unreclaimed pits with the toes of highwalls submerged. Backfill and highwall reduction methods are complicated by the presence of pit water.

The A-8 open pit uranium mine as shown in Figure 1, was abandoned in the early 1970's, without any reclamation measures. Reclamation activities now are in progress to control hazards and further environmental degradation. Approximately 3.5 million cubic yards of backfill will be placed in the pits. Surface drainage channels will be constructed and cover soils placed to control Radium 226 emissions and to assist with revegetation.

## Site Description

Reclamation efforts are complicated by the presence of an estimated 210 million gallons of pit water, varying in depths up to 85 feet. This pit water, the analysis of which is shown on Table 1., is acidic and radioactive and will require treatment when discharged. Additional complications result from several geologic faults that occur through the pit, as noted on Figure 1. Adequate quantities of spoils for backfill activities are stockpiled adjacent to the pit. However, the last materials removed from the pit, (lower 25 feet), are unsuitable for all backfill uses and will require selective handling. These unsuitable spoil materials have higher concentrations of arsenic, iron, aluminum, uranium (natural), and its daughter product Radium 226. Typical laboratory analysis of the spoils pile are represented on Table 2.

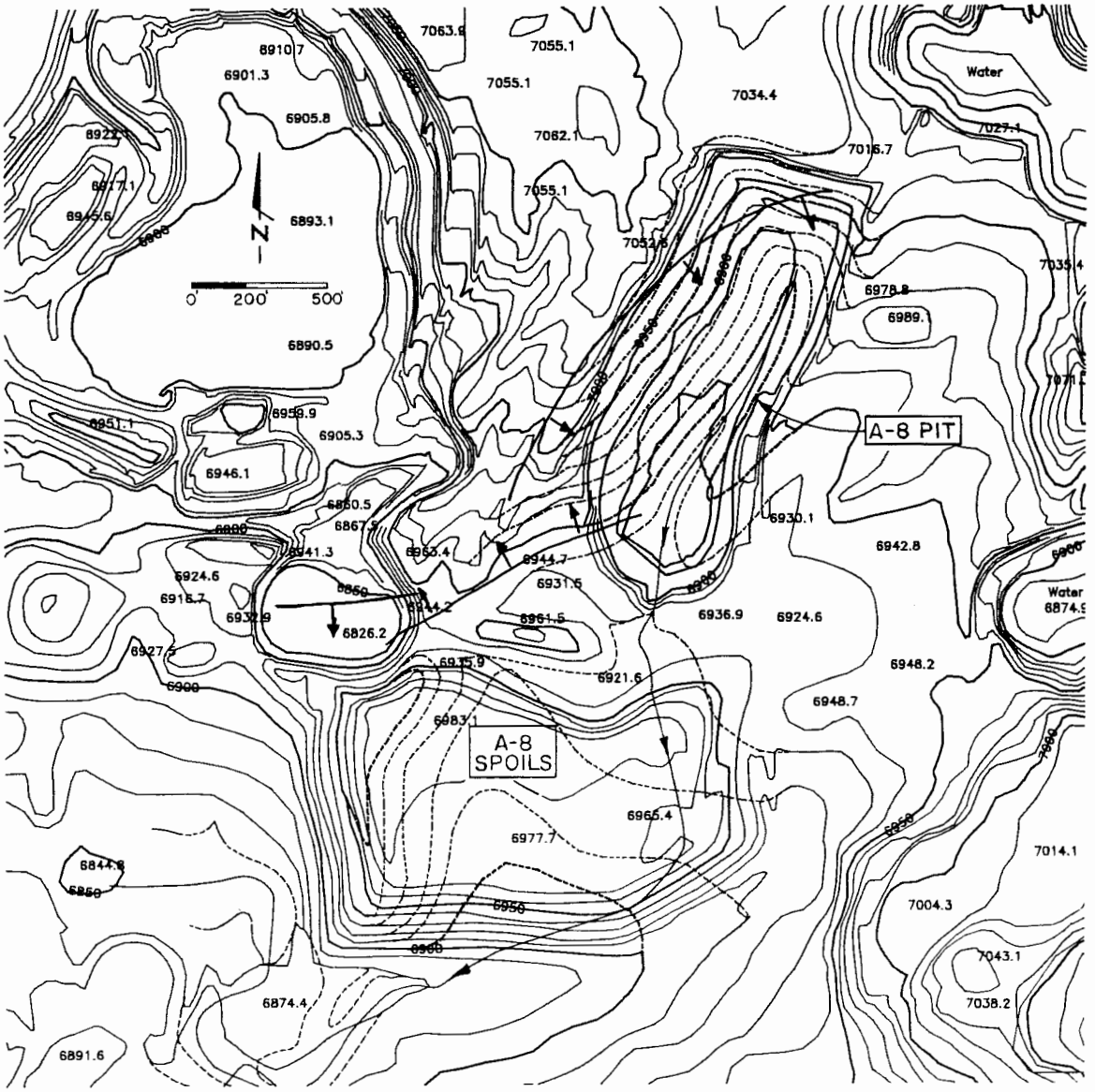



Figure 1. Site map of the A-8 Project as located in the East Gas Hills region of Central Wyoming.

Table 1. Existing water quality for pit water in the A-8 open pit uranium mine.

| Testing Parameters | mg/1 | Testing Parameters | mg/1 |
|---|---|---|---|
| Total Suspended Solids | 6.0 | Boron (B) | 0.08 |
| Total Dissolved Solids | 3017 | Cadmium (Cd) | <0.01 |
| Conductivity @ 25c,umho/cm | 2608 | Chromium (Cr) | <0.01 |
| Sodium (Na) | 58 | Copper (Cu) | <0.01 |
| Potassium (K) | 23.6 | Fluoride (F) | 0.28 |
| Calcium (Ca) | 568 | Iron (Fe) | <0.03 |
| Magnesium (Mg) | 99 | Lead (Pb) | 0.08 |
| Chloride (Cl) | 23 | Manganese (Mn) | 3.8 |
| Sulfate (SO4) | 1940 | Mercury (Hg) | <0.001 |
| Bicarbonate (HCO3) | 3.1 | Molybdenum (Mo) | <0.05 |
| Carbonate (CO3) | 0 | Nickel (Ni) | <0.03 |
| pH, Units | 5.08 | Nitrate (NO3 as N) | 0.19 |
| Aluminum (Al) | <0.1 | Selenium (Se) | 0.012 |
| Ammonia (NH3 as N) | 0.42 | Uranium (U) | 0.083 |
| Arsenic (As) | 0.001 | Vanadium (V) | <0.1 |
| Barium (Ba) | <0.05 | Zinc (Zn) | 0.19 |

| | pCi/l | LLD uCi/ml |
|---|---|---|
| Radium 226 | 6.86 ± 0.45 | 2x10E-10 |
| Gross Alpha | 2.49 ± 47 | 1x10E-9 |
| Gross Beta | 105 ± 21 | 1x10E-9 |

Table 2. Typical soils laboratory analysis for the A-8 Spoils pile to be used as backfill.

| Sample I.D: | A-8-4 | A-8-4 | A-8-4 |
|---|---|---|---|
| Interval: | 5-15 | 35-45 | 50-65 |
| Sample Date: | 10-89 | 10-89 | 10-89 |
| Sample Number: | 11-08-89 | 11-08-89 | 11-08-89 |
| Parameter List: | Long | Long | Short |
| pH, s.u. | 7.3 | 7.3 | 7.4 |
| Moisture % | 12.30 | 8.00 | 8.33 |
| Saturation % | 44.6 | 30.0 | |
| Cond., mmhos/cm | 2.94 | 2.81 | 2.80 |
| Sand % | 54 | 68 | 60 |
| Silt % | 30 | 18 | 20 |
| Clay % | 16 | 14 | 20 |
| Texture | SL | SL | SCL |
| Organic Matter % | 0.4 | 0.4 | |
| Organic Carbon % | 0.2 | 0.2 | |
| Nitrate as N, ppm | 1.9 | 1.6 | |
| Lime % | 1.4 | 2.0 | 1.2 |
| Neut. Pot., ppt (1) | 14 | 20 | 12 |
| Acid Pot., meq/100g | 86 | 33 | 26 |
| Acid Base Pot., (1) | -28 | 4 | -1 |

| Sample I.D: | A-8-4 | A-8-4 | A-8-4 |
|---|---|---|---|
| Interval: | 5-15 | 35-45 | 50-65 |
| Sample Date: | 10-89 | 10-89 | 10-89 |
| Sample Number: | 11-08-89 | 11-08-89 | 11-08-89 |
| Parameter List: | Long | Long | Short |
| Calcium, meq/1 | 30.8 | 27.2 | |
| Magnesium, meq/1 | 7.65 | 4.02 | |
| Sodium, meq/1 | 1.43 | 1.39 | |
| SAR | 0.33 | 0.35 | |
| ** | | | |
| Arsenic, ppm | 29 | 27 | 18 |
| Aluminum, ppm | 37000 | 27000 | 4400 |
| Boron, ppm | 270 | 330 | <1.0 |
| Copper, ppm | 11 | 9.4 | 10 |
| Iron, ppm | 21120 | 15880 | 11660 |
| Lead, ppm | 23 | 13 | 12 |
| Manganese, ppm | 57 | 44 | 49 |
| Molybdenum, ppm | 3.5 | 6.1 | <1.0 |
| Selenium, ppm | 4.2 | 5.9 | 3.8 |
| Zinc, ppm | 38 | 28 | 18 |
| Uranium, pCi/gm | 95.8 | 32.0 | 18.5 |
| Ra-226, pCi/gm | 31.5 | 21.2 | 6.8 |
| Ra-226 Prec. +/- | 0.6 | 0.5 | 0.3 |

(1) T. CaCO3/100 T.;
"-" indicates lime deficiency.

** Total Metals reported following SW-3050 digestion unless otherwise noted.

## History

Mining activities for the A-8 Uranium Pit occurred between 1969 and 1973 and then again in 1975 when some limited underground mining occurred. Originally, the pit was designed to be 2000 feet by 450 feet and 200 feet deep. The pit was mined with scrapers, backhoes and trucks. When the excavation reached an elevation of 6,855, or a depth of 175 feet at the North end of the pit, the groundwater began flooding the pit at up to 600 gpm. When mining activities concluded at the South end of the pit the ore bearing roll front was 85 feet below the water table. The pit quickly filled with water after the mining activities ceased.

Most of the spoils removed from the pit were stockpiled down gradient from the pit in the A-8 Spoils pile. Some spoils were used to backfill the A-9 Pit, which is continuous with the A-8 Pit area. Because the A-8 Spoils pile was reaffected after 1977, it is not eligible for reclamation under the AML criteria. Predesign activities, which were completed during the Fall of 1989, are source documents for this report.

## Existing Conditions

The A-8 Pit is a long narrow pit measuring about 1700 feet in the North-South direction and 500 in the East-West direction. Exposed highwalls (above the pit water surface) range from a height of about 80 feet at the South end to about 170 feet at the North. Sonar mapping of the pit bottom indicates the water depth is only about 15 to 20 feet near the North end, reaching a maximum of about 85 feet near the South end. The North wall of the pit is comprised of backfilled mine spoils, while the remainder is cut into bedrock. The slope of the backfill in the North end is between 33 to 35 degrees. Most of the West wall has been flattened to about 1.5:1, due to a fault which intersected that face. A major slope failure occurred after mining activities ceased.

The rock units exposed in the highwalls are members of the Upper Wind River Formation which are Tertiary sedimentary deposits and consist of fine to coarse grained sandstone, claystone and discontinuous conglomerate beds. These materials were deposited during a combination of marine and braided stream environments which were interrupted by periods of erosion. The resulting profile consists of discontinuous layers, with intermixing of soils within major layers. Following deposition, regional uplifting occurred to the South and West resulting in a series of faults in the Wind River Formation. The faulting frequency is variable but is locally intense in the project vicinity. These faults usually strike from East - West to Northeast - Southwest and may dip either North or South, usually at inclinations of 55 degrees or more. Several faults were mapped in the highwalls within the boundaries of the project as noted on Figure 1. Other faults were observed, but were not accessible for determination of strike and dip. Visual observations indicate highwall failures attributable to the faults approximately paralleling to and dipping toward the pit walls (Pool, 1989).

One of the primary factors influencing the surface water hydrology and geomorphic conditions in the Gas Hills is the Beaver Divide which is a prominent plateau. The A-8 Pit is located relatively close to the Beaver Divide and this results in high relief and steeper basin slopes at the site. The primary surface water and geomorphic design consideration is to reconstruct the drainage basins and channels in the disturbed areas. Of particular interest, is the reconstruction of the West Fork of the Canyon Creek drainage. The headwaters of this drainage were diverted by the historic mining operations. Presently the runoff is conveyed to a stock pond located just to the East of the A-8 Pit. Most of the mining disturbances have intersected the Canyon Creek drainage and placement of spoil piles including the A-8 Spoils have blocked the historic drainage channel (Lidstone, 1989).

The site was evaluated for threatened and endangered plants because of the relatively high potential for the occurrence of such species, but none such species were observed. No special precautions are necessary for protecting threatened or endangered plant species on this site (Schreibeis, 1989).

AML Project 16E Site A-8 does not provide critical habitat for threatened or endangered wildlife species. There are no active raptor nests on any of the highwalls or spoil piles. Reclamation should not adversely affect any threatened or endangered species.

The existing pit water is unavailable to most wildlife due to restricted access and the steep rim of the highwalls. The water is acidic (pH 4.84-5.50) with a high radium level (9.2 pCi/l) No wetland vegetation is present in the pit (Kling, 1989).

Two previously recorded archaeological sites had been reported in the area, but no evidence of either could be found. As such, no cultural resources were located, and clearance is recommended for the project (Frontier Archaeology, 1989).

The natural background Radium 226 (Ra-226) soil concentration in the Gas Hills was previously determined to be 3.7 +/- 1.7 pCi/g (Radiological Risk Assessment for Various Abandoned Mine Lands Clean-Up Criteria, Hersloff, 1988). The proposed EPA standard for mine waste of 5 pCi/g above background would then be 8.4 pCi/g. The linear correlation of gamma exposure rate at 5 cm above the surface and Ra-226 concentration in soil at the A-8 Project area is defined by:

$$\text{Ra-226, pCi/g} = 0.43 \times \text{uR/hr} - 10.54$$

A soil Ra-226 concentration of 8.4 pCi/g corresponds to a total unshielded gamma exposure rate of 44 uR/hr, with a certainty of 85% (Hersloff,1989).

The Goose Egg (Triassic-Permian) formation has been deposited, on the surface, along the South limits of the project site. It is easily identified by the reddish color. The fine grained Goose Egg was transported down drainage channels to this deposition site from the South. These soils are an excellent plant growth medium source. Minimal surface disturbance would be required to obtain large quantities of these suitable coversoils. Finished reclaimed slopes should be no steeper than 5:1, and where winter time sun light exposures are most direct (South face), 6:1 slopes will help minimize erosion to slopes (Schreibeis, 1989). The A-8 Spoils stockpile is unacceptable as a plant growth media. Acid-Base Potential and Radium 226 values are greater than Wyoming Department of Environmental Quality and Environmental Protection Agency guidelines. Examples of Acid Base Potentials and Ra-226 values for the A-8 Spoils are shown on Table 2.

Backfilling the A-8 Pit requires an evaluation of the impact on the groundwater quality. The existing pit water, which is solely fed from groundwater, is acidic and generally of poor quality. Groundwater around the site is less acidic, with fewer TDS, radionucleides and metals, are shown on Tables 1 and 2. Batch tests and column leach tests were conducted on the combinations of A-8 Spoils, pitwater and groundwater to determine the impact of down gradient groundwater quality as a result of backfilling. The interaction was evaluated by using a

geochemical program PHREEQE, which simulates mixing of different waters. It predicts resultant water quality at equilibrium conditions the results of these evaluations are shown on Table 3. The overall quality is not impacted significantly, and the A-8 Pit water will become more neutral resulting in the reduction in metal concentrations (e.g., Al, Fe, Pb Mn, Ni and Zn). However, concentrations of U, Ra-226, Se and Mo will increase due to the reaction with backfilled spoils. This can be seen by referring to Tables 3 and 4 for comparisons of water quality of pit water and predicted down gradient groundwater (Olsen, 1989).

The A-8 Spoils pile is segregated. The upper 25 feet was removed last in the mining operation and is a poorer quality material. The two critical metals (Selenium and Molybdenum) are elevated in the upper sequence, as are radionuclides, Uranium and Radium. Tables 4 and 5 represent the laboratory analysis of the A-8 Spoils pile by depths.

Table 3 Comparison of Predicted to Existing Groundwater Quality Down Gradient From A-8 Pit.

| Parameter | Units | Existing | Predicted |
|---|---|---|---|
| General | | | |
| pH | S.U | 7.36 | 7.26 |
| TDS | mg/L | 1,428 | 1,728 |
| Major Cations | | | |
| Ca | mg/L | 290 | 373.2 |
| Mg | mg/L | 40.4 | 45.0 |
| K | mg/L | 12.0 | 32.8 |
| Na | mg/L | 61.0 | 64.4 |
| Metals | | | |
| Al | mg/L | <0.10 | 0.04 |
| B | mg/L | 0.11 | 0.5 |
| Cu | mg/l | <0.01 | <0.01 |
| Ni | mg/L | <0.05 | <0.01 |
| Se | mg/L | <0.001 | 0.20 |
| Zn | mg/L | 0.04 | 0.04 |
| U | mg/L | 0.422 | 2.6 |
| Ra-226 | piC/L | 11.9 | 14.5 |

Table 4. Summary of Total Metal Constituents in the A-8 Spoils Pile.

| | | | | TOTAL METALS | | | | |
|---|---|---|---|---|---|---|---|---|
| | pH | EC | Lime | As | Mo | Se | U | $Ra^{226}$ |
| | su | mmhos/cm | % | ppm | ppm | ppm | pC/gm | pC/gm |
| MEAN (A8) | 6.7 | 2.8 | 2.2 | 23.5 | 2.9 | 5.2 | 33.7 | 12.1 |
| SDEV (A8) | 1.3 | 1.3 | 1.9 | 11.1 | 2.7 | 4.8 | 49.5 | 16.5 |
| MIN (A8) | 1.5 | 0.7 | ND | 3.4 | 1.0 | 0.2 | 3.0 | 0.9 |
| MAX (A8) | 7.9 | 10.2 | 11.4 | 77.0 | 12.0 | 27.0 | 368.0 | 118.0 |

Table 5. Statistical Analysis of the A-8 Spoils Chemistry by Depth

| | pH | ABP | U | $Ra^{226}$ |
|---|---|---|---|---|
| | (s.u) | ($TCaCO_3$/1000T) | (pC/gm) | (pC/gm) |
| MEAN 0-25'[A] | 6.9 | -13.2 | 74.3 | 16.3 |
| MEAN 25-TD[B] | 7.3 | +15.0 | 26.9 | 11.2 |
| SDEV 0-25' | 1.0 | 8.6 | 102.5 | 18.0 |
| SDEV 25-TD | 0.2 | 14.8 | 15.7 | 8.1 |
| MIN 0-25' | 4.3 | -28.0 | 11.3 | 2.7 |
| MIN 25-TD | 6.8 | -10.0 | 4.0 | 2.2 |
| MAX 0-25' | 7.6 | +2.0 | 368.0 | 46.9 |
| MAX 25-TD | 7.5 | +40.0 | 84.9 | 33.9 |

[A] n = 10 Samples
[B] n = 30 Samples
TD = Total Depth

## Results and Discussion

The overall goal of the Wyoming Abandoned Mine Lands program is to reduce safety and health hazards and to mitigate environmental disturbances. This goal is to be accomplished with the most cost effective reclamation plan. Based on the technical evaluation of the A-8 Pit site some specific reclamation criterion are:

- Pit highwalls are unstable and consequently should be regraded or backfilled to 23-28 degree slopes.
- Backfill should be accomplished by maintaining pit water level near present levels, by pumping while backfilling spoils into pit water starting at shallow (North) end building a pad towards deep (South) end.
- Construct backfill into 12-inch thick (loose) lifts.
- Limit fill slopes to 5:1, except on South exposures then use 6:1.
- Use erosion control ditches on longer slope faces.
- Topsoil and/or riprap all 2nd order channels.
- Design channel capacities for 100-year 24-hour rainfall runoff events and stability to 10 year 6-hour events.
- Selectively place A-8 Spoils in pit and backfill as to minimize groundwater degradation and concentrations of Radium 226 near finished ground levels.
- Coversoils and topsoils are limited at the project site. Strip and stockpile separately all sources especially the Goose Egg formation at the south end of A-8 Spoils.
- Utilize native seed mixes including western wheat grass (Agropyron Smithii), thick spike wheat grass (Agropyron Cristatum), bluebunch wheat grass (Agropyron Spicatum), indian ricegrass (Oryzopsis Hymenoides), among others.
- Pitting should be used in conjunction with seeding.
- Reconstruct access roadway through site and fence all reclamation.
- Maximum Radium 226 soil concentration criterion is 20 pico Curies / per gram (pCi/g), including background, with average Radium 226 soil concentration of 10 pCi/g in the top five feet of backfill.

## Alternate Reclamation Plans

Three basic reclamation plans were developed for evaluation by technical staff and design engineers. The options ranged from: 1. complete backfill and recontouring to original condition; 2. partial backfill allowing for exterior drainage out of the pit; and 3. backfill most of pit leaving the groundwater (pond) exposed.

**Option 1** This was considered the most costly of the three options. An additional one million cubic yards of backfill would be required over Option 2. The only advantage this plan had over Option 2 was surface grading could allow for dispersion of run off rather than concentrating the flows to less stable channels.

**Option 2** A partial backfill and external drainage plan accomplished all technical objectives, except for leaving an accessible water source to the surface. However, because the groundwater is a poor quality source and radionucliedes are high it was not considered to have any environmental value.

**Option 3** Partial backfill with an impoundment at the lower (South) end of the pit would require the least backfill the three options. But highwalls would need to be reduced and the fill slopes below water level would not be compacted leaving material loose and dangerous to anything attempting to access the pit water. Quality of the pit water is poor, limiting its value. Surface runoff generated within the reclaimed pit would be retained in the pond.

Even though Option 3 would be cheaper to construct than Option 2, problems would develop. Building a stable pond and constructing stable fill slopes steeper than 5:1 on the entire site would be difficult. Also, surface runoff is more useful in a proposed retention pond down gradient from the A-8 site make Option 3 less attractive than Option 2.

## Final Reclamation Plan (Option 2)

The reclamation plan selected for implementation is represented in Figure 1. It reflects the general finished grading and surface drainage plan. The estimated construction cost is $4,180,000. Construction is scheduled to occur between June, 1990 and November, 1991.

Basic elements of the reclamation plan are as follows:

**Pit Dewatering** Pit water will be removed (pumped) from the pit continually during construction. The static level of the pit water will be maintained at a constant level to minimize concerns for highwall failure due to saturation of highwalls. Pumping water will be about 2 cfs (1,000 gpm). Permitting for discharge of pit water will address treatment and channel degradation issues.

**Existing Roadway** The existing roadway, North and South along the West edge of the A-8 Pit to the South of A-8 Spoils, will be retained and/or reconstructed.

**Fencing** Field fencing will be constructed around the entire disturbance to act as a protection against over-grazing during the vegetation establishment period. Cattle guards in lieu of a gate will be installed at each crossing.

**Existing Power Poles** All existing power and telephone poles will be removed from the site and stockpiled for Landowner's use.

**Backfilling of Project** The A-8 Spoils will be used to backfill the pit. Some selective handling will be required. All spoils placed below the projected groundwater level and within five feet of finished ground surface is to be taken from the lower half of the spoils pile (top 25 feet of Spoils pile is to be sandwiched into the backfill). The backfilling activities in the A-8 Pit commence at the North end of the pit where existing water is shallow. The backfill pad which would be constructed 3-5 feet above the existing water level of the pit would be continually worked towards the South end.

**A-8 Spoils** The A-8 Spoils will be removed in quantities needed to complete the backfill operations at the A-8 Pit,

after which the spoils would be left in a reclaimable condition. Reclamation will be accomplished by others (Spoils pile is not eligible for AML reclamation).

**Cover Soils** There are adequate quantities of cover soils available just South of the A-8 Spoils. Materials consist of the Goose Egg formation. There are also some cover soils available at the South end of the A-8 Pit which will also be utilized to cover the site for revegetation. Small piles of topsoil were left along the North and East side of the A-8 Pit and also at the East side of the A-8 Spoils. There are some topsoils that have been salvaged by past mining operations and stockpiled or winrowed around the A-8 Spoils.

**Drainage and Erosion Control Measures** Drainage and erosion control measures will be accomplished by construction of stable earthen and riprapped channels. Some erosion control ditches will be constructed along the long fill slopes in the A-8 Pit to mitigate rilling and gullying which normally occur on such reclaimed slopes.

## Conclusions

The purpose of the Abandoned Mine Lands program is to reduce safety and health hazards and to enhance the environment. Reclamation of the A-8 Pit in the East Gas Hills region of Central Wyoming accomplishes this goal, in the following ways:

- Existing unstable highwalls (200 feet) will be backfilled to eliminate safety hazards.
- Water quality of groundwater will be enhanced by backfilling the pit with select spoils to attenuate and buffer heavy metals and radionucliedes present in the pit water.
- Construct stable externally draining surface channels to direct surface runoff to existing wetland areas.
- Revegetate site (29 acres) with wildlife cover and food source.
- Restore environment as near as possible to a stable yet natural looking condition.

## Literature Cited


Pool, John M., 1989. Geotechnical Investigation Project 16E East Gas Hills, Wyoming. Design Investigation Report for Project 16E-1, A-8/C-17 and C-18 Site. Chen-Northern, Inc.

Lidstone, Christopher. 1989. Surface Water Hydrology and Geomorphology for the East Gas Hills AML 16-E Project. Design Investigation Report for Project 16E-1, A-8/C-17 & C-18 Site. Lidstone & Anderson.

Schreibeis, Ronald. 1989. Threatened and Endangered Plant Survey. Range Inventory & Analysis.

Kling, Craig. 1989. Wildlife Resources on AML Project 16E A-8 Site, Fremont County, Wyoming. Mariah Associates, Inc.

Welch, James. 1989. Class III Cultural Resource Inventory of Abandoned Mine Lands Project 16E. Frontier Archaeology.

Hersloff, Lyda. 1989. Recommended Radium-226 Clean-up Criterion for the AML 16E-1 Project. Radiant Energy Management.

Schreibeis, Ronald. 1989. Soils and Spoils. Range Inventory & Analysis.

Olsen, Roger. 1989. Evaluation of Post-Reclamation Groundwater Quality (A-8, Tee and B-2/B-3 Pits), AML 16-E. Camp Dresser & McKee, Inc.

This page intentionally left blank.

# FIRE DIAGNOSTIC SIMULATION FOR BURNING COAL WASTE BANKS[1]

by

J. Richard Jones and Robert F. Chaiken[2]

Abstract. Past research has shown that extinguising abandoned underground mine fires has not been successful because of the inability to locate fire zones accurately. The U.S. Bureau of Mines has developed a mine fire diagnostic technique which determines gas signatures and pressure flows in the mine atmosphere from a two-dimensional array of boreholes which intersect the strata of an abandoned mine. This technique has worked well in delineating both heated and non-heated subsurface zones at several abandoned mine fire sites. To apply this same technology to fire zones in burning coal waste piles requires a three-dimensional array of boreholes in the pile. Utilizing a scale model of a coal waste pile, the Bureau is currently studying the flow fields that would be induced by a three-dimensional array of boreholes. This report describes some initial results of these physical modeling studies.

Additional key words: Ventilation analysis, waste banks, AML Fires, fire diagnostics.

## Introduction

Uncontrolled fires in abandoned coal mines and coal refuse piles can cause devastating results to both life and property. Although it has been estimated that more than 500 coal waste bank or coal strata fires exist in the United States, it is reasonable to assume that additional fires have gone undetected and will likely remain so until surface anomalies become visible, i.e., subsidence, noxious fumes, or spoiled vegetation (Johnson and Miller 1979). It is known that elevated coal temperatures can be sustained for long periods with little oxygen supply and little or no noticeable surface disturbances. Therefore, the development of physical models of dynamic coal fire environments are necessary to assure more accurate diagnosis.

Previous surface induced ventilation tests of underground coal strata at Bureau of Mines field projects at Renton and Carbondale, Pa suggest that the presence of heated zones significantly changes the gas flow and stoichiometry of the region enough to create a signature for those zones (Kim, et. al. 1989; Dalverny and Chaiken, in prep.). When underground flow patterns were induced by a surface exhaust fan, it was found - through ventilation network analysis -that the effective underground resistance to gas flow was not uniform across suspected fire zones, and that calculated resistance values could indicate the presence or absence of a fire zone. In these projects, it was not possible to fully verify the results of the ventilation network analyses, since ground truth is unobtainable without detailed excavation. While proof of the concept that fires can be located by the analysis of borehole pressure measurements will eventually lie in actual field studies, calibration and evaluation of the derived network algorithm can be accomplished through laboratory modeling studies involving flow dynamics within a scale-model of coal waste material. An analysis of the induced flow patterns within a scale model waste bank material should enable prediction of the fluid dynamics that are involved with in situ abandoned mined-land fires which occur in unconsolidated waste banks.

## Methodology

### Materials

The coal waste material used for analysis was collected from the preparation plant of an active underground mine. The material was unweathered. The results of the proximate and ultimate analysis of the waste material are presented in Table 1. The particle size distribution and porosity of the waste material were determined according to standard testing methods (Lambe 1968). The porosity is 28.5% and the general size distribution of the material is illustrated in Figure 1. Using data collected from a constant head permeameter (6

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]J. Richard Jones is a Geologist and Robert F. Chaiken is a Supervisory Research Chemist, Pittsburgh Research Center, U.S. Bureau of Mines, Pittsburgh, PA 15236.

ft in length, 8 in in diameter) with Darcy's Flow Law Equation, the permeability of the waste material was found to average $10^{-2}$ Darcy's or 0.02in/sec. The physical characteristics of this material are consistent with those of other types of coarse-grained coal waste materials.

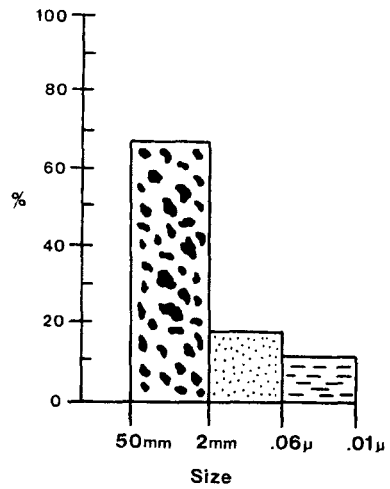



Figure 1. Histogram of sediments size-distribution of coal waste material.

## Waste Bank Scale Model

The waste bank scale model consists of a 10 x 5 x 4 foot (200 cubic feet) porous side steel-grated frame box containing coal waste (Fig. 2). Two vertical boreholes of 4 in and 2 in diameter were placed within the box 30 in from the top surface of the pile. These boreholes can be alternatively used to change the borehole suction diameter. The vacuum suction pressure through the waste material is maintained by an exhaust fan. A series of 120 stainless steel pressure probe rods were inserted into the waste material by drilling through the sides of the box at 1, 2, and 3 ft depths beneath the pile surface at 1 ft centers (Fig. 2). After suction was applied to the respective boreholes, probe vacuum changes were monitored with a series of magnehelic gauges.

Sixteen suction ventilation tests were completed in order to access the flow patterns of air under vacuum through the waste bank material for suction at both boreholes and by varying the rate of fan suction through the system. Reduction in the rate of fan suction was accomplished by a controlling baffle placed near the source of suction. The controlled fan suction pressures used within the pipe averaged 36, 26, and 17 inches of water or 100%, 75%, and 50% capacity of maximum suction. A 50% reduction in flow velocity did, however, affect the vacuum pressure diffusion pattern by about a factor of two when compared with the 100% and 75% flow rates, i.e., the probes recorded approximately 50% less vacuum at each monitoring port.

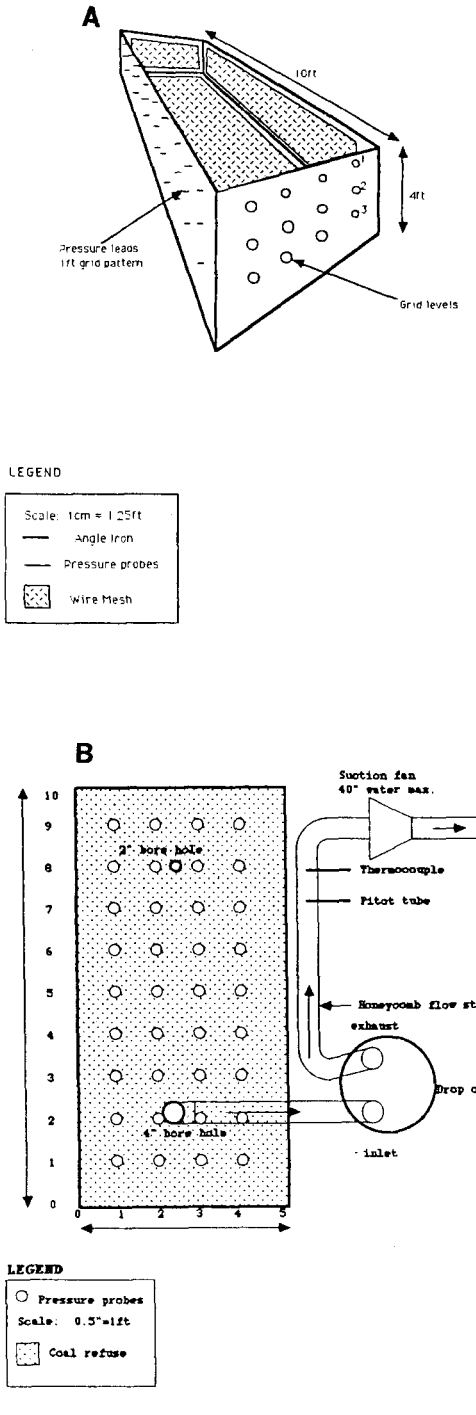



Figure 2. Perspective view of size-dimensions of waste bank scale model (A). Planar view of waste bank scale model (B).

To model the diffusion of air through the waste material under vacuum, suction was applied to both borehole diameters at 100% suction capacity of the fan. Eight separate tests were completed for each borehole to test the reproducibility of the flow patterns.

Table 1. Refuse Characteristics.

| Proximate Analysis | As-received | Dry | Dry Ash-free |
|---|---|---|---|
| Moisture | 2.78 | -- | --- |
| Ash | 67.37 | 69.29 | --- |
| Volatile Matter | 12.49 | 12.85 | 41.85 |
| Fixed Carbon | 17.36 | 17.86 | 58.15 |

| Ultimate Analysis | As-received | Dry | Dry Ash-free |
|---|---|---|---|
| Hydrogen | 1.84 | 1.57 | 5.11 |
| Carbon | 19.23 | 19.78 | 64.42 |
| Nitrogen | 0.27 | 0.28 | 0.91 |
| Sulfur | 5.29 | 5.44 | 17.72 |
| Oxygen | 6.00 | 3.64 | 11.84 |
| Ash | 67.37 | 69.29 | --- |
| Heating Valve (BTU/LB) | 3344 | 3440 | 11203 |

Two sets of mathematical equations were fit to the vacuum data: 1) a four- dimensional surface where vacuum pressure patterns were tested as a function of the three orientation co-ordinates (x,y,z) of the monitoring ports within the waste bank and 2) a three-dimensional surface for each depth level where vacuum pressure patterns were tested as a function of the x-and y- co-ordinates of the monitoring ports within the box.

## Analysis

An analysis of variance model was used to test the differences between the three fan suction rates. For the 4 inch borehole tests vacuum rates at the monitoring ports were not significantly different between 100% and 75% fan capacities. For suction at the 4 inch borehole, differences were found between the 50% fan capacity, and both 100% and 75% fan capacities. Similar results were also found between the monitoring ports at respective fan capacity tests for suctions at the 2 inch borehole. Differences in vacuum flows within the waste bank were not observed between the 4 inch and 2 inch diameter boreholes at either 100% or 75% fan capacity.

Since gas flowing toward a borehole under suction should follow a radial flow pattern, it was assumed that second-order polynomial equations would best model and predict vacuum flow patterns toward both the 2 inch and 4 inch diameter boreholes. Results of the four-dimensional analysis for the sixteen tests gave predictive surface $R^2$ values between 0.75 and 0.83 for the respective 2 inch and 4 inch diameter borehole suction tests. The result of the 3-dimensional analysis for the sixteen tests gave predictive surface $R^2$ values between 0.76 and 0.98 for the respective 2 inch and 4 inch diameter borehole tests. These high values for $R^2$ (explained variance) predict very well the experiment flow patterns.

A three-dimensional plot comparison between the test data and predicted surfaces for the 3 levels for a four inch borehole diameter test is presented in Figure 3. There is good correspondence between the test data and the predicted values at each monitoring level. Level 3, which occurs at a depth of 3 feet beneath the top of the coal waste material, provided the least predictive surface with a $R^2$ value of 0.76. The other two levels have considerably better fits with respective $R^2$ values of 0.87 and 0.98.

An analysis of the residuals associated with the respective predictive surfaces did not show any secondary surfaces or patterns that might suggest either an uneven size distribution of the waste material or irregular vacuum flow rates through the material toward the borehole. Analysis of the pressure flow characteristics between the monitoring probes flowing toward the bottom of the borehole show that flows were within the regime of Darcy conditions for each of the tests. Thus it appears appropriate to model the flows within the constraints of Darcy's Law.

## Conclusions and Technical Evaluations

Flows toward the boreholes generally follow a radial pattern and can be well predicted with a series of polynomial equations. The least predictive surfaces were found to occur within the lower (3 ft) levels of the waste bank box. It is believed that the effect of the impermeable steel bottom of the waste bank box is causing vaccum flows to follow a non radial pattern at this level, i.e. flows are not radial distributed throughout the coal waste material. The vacuum distance of borehole suction within the waste material radiates about 6 feet at level 3 and about 4 feet at levels 1 and 2 (Fig. 3). For monitoring ports beyond these distances for the respective tests suction pressures were undetectable.

These tests have revealed promising results which should enable modeling of vacuum flows within *in situ* waste banks:

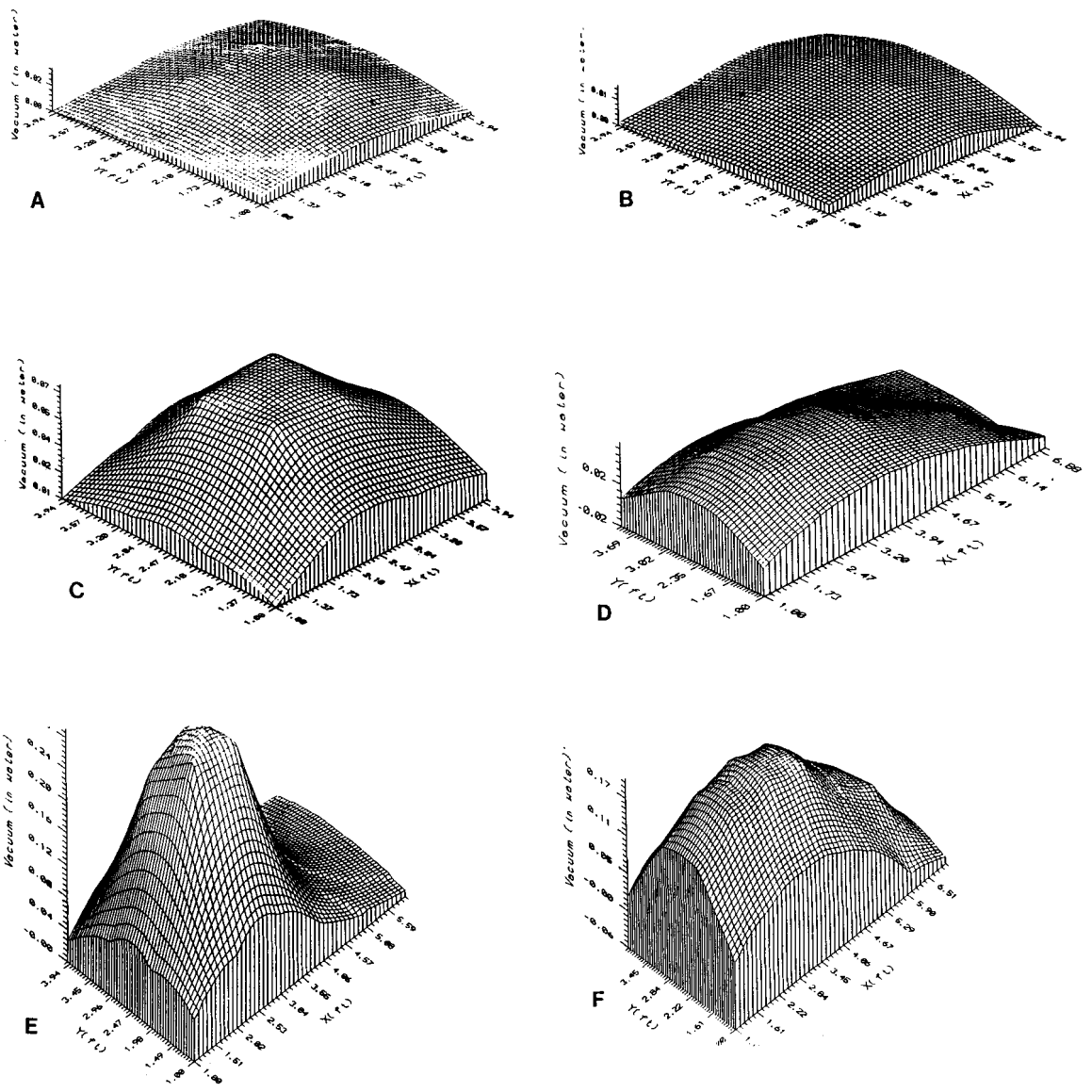



Figure 3. Comparison between three-dimensional experimental and predicted surfaces for vacuum flows by levels. Experimental Level 1 (A), Predicted Level 1 (B), Experimental Level 2 (C), Predicted Level 2 (D), Experimental Level 3 (E), Predicted Level (F).

1) Flow patterns through coal waste material generally follow a radial pattern.
2) These flows follow Darcy's Law and as such, their fluid dynamics can be modeled with Darcy's Flow Equations.
3) The flows can be predicted with a series of second-order polynomial equations with a high degree of statistical reliability.
4) For borehole suctions within the range of 36 to 25 inches of water at the fan source either a 2 inch or 4 inch borehole can be used without significantly altering the flow patterns within the waste material.

The results of these experiments combined with the existing fire diagnostic technology suggest that field tests at an <u>in situ</u> burning waste bank are warranted. Field testing will include the initial placement of boreholes within a pattern that will provide optimum coverage of the waste bank. The pattern will be dictated by the physical characteristics of the waste material (size, porosity, permeability, etc.) as well as the geometric dimensions of the waste bank. Two inch diameter boreholes appear to be adequate as long as flow pressures at the suction source can be maintained above 25 inches of water. Fan suctions will be applied with vacuum pressures monitored with the magnehelic gauges. Comparison will be made between the field-collected data and the prediction model. This will enable verification with the four-dimensional and three-dimensional predictive surfaces determined from the waste bank model experiments.

## Acknowledgements

Mr. Thomas Justin assited with the experimental design. Jeff Puzas (West Virginia University) and Scott Schell (The Pennsylvania State University) assisted with simulation tests while employed as summer students at the Bureau of Mines. Ms. Deanna Baumann is thanked for manuscript preparation.

## Literature Cited

Dalverny, L. E. and R. F. Chaiken, in prep. Mine Fire Diagnostics and Implementation of WIFE at Renton, PA. Bureau of Mines, Report of Investigation.

Johnson, W. and G. C. Miller, 1978. Abandoned Coal-Mined Lands-Nature, Extent, and Cost of Reclamation. Bureau of Mines Spec. Pub. 6-79, 29pp.

Kim, A. G., L. E. Dalverny, and T. R. Justin, 1989. Mine Fire Diagnostics, Recent Developments, p. 151-170. In: Proceedings Symposium on the Evolution of Abandoned Mine Land Technologies (Riverton, WY, June 14-16, 1989). Wyoming Department of Environmental Quality.

Lambe, T. W. 1967. Soil Testing for Engineers. John Wiley and Sons, New York, 165p.

This page intentionally left blank.

# MINE FIRE DIAGNOSTICS IN ABANDONED BITUMINOUS COAL MINES[1]

by

Louis E. Dalverny, Robert F. Chaiken, and Ann G. Kim[2]

Abstract. The Bureau of Mines is developing a Mine Fire Diagnostic methodology to locate and monitor fires in abandoned coal mines and waste banks. The method assumes that: (1) measurable changes in the emission of low molecular weight hydrocarbons from coal are directly temperature dependent, and (2) analysis of controlled underground air flow between borehole sampling points can determine the source of the hydrocarbons.

A hydrocarbon ratio, R1, a signature for heated coal, is related to the relative concentrations of methane and total hydrocarbons (methane through pentane). An exhaust fan attached to a borehole creates underground pressure gradients that control the movement of gases past a network of borehole sampling points. Changes in R1 value are correlated with the assumed direction of gas flow between borehole pairs. Multiple tests with different orientations allow spatial definition of both heated and cold subsurface zones.

The Mine Fire Diagnostic methodology has been used to define non-contiguous combustion zones, locate cold boundaries in remote combustion areas, determine effects of extinguishment activities on the fire, and accomplish long-term monitoring. Results obtained at two abandoned bituminous coal mine fires are discussed. This method's advantages are: (1) hydrocarbon emission changes are caused only by temperature changes; (2) emission changes can be detected for temperatures well below the ignition point of coal; (3) hydrocarbons in gas samples can be detected at concentrations of 1 ppm; (4) use of a ratio eliminates air dilution effects; (5) induced vacuums can be detected and mine gases withdrawn several hundred feet from the suction point; and, (6) system operation is relatively simple, requiring only readily available equipment.

## Introduction

Fires in abandoned coal mines and waste banks are a relatively common occurrence found in every coal producing state. Abandoned mine fires present serious health, safety, and environmental hazards caused by the emission of toxic fumes, subsidence, and the deterioration of air quality. Such fires usually depress property values for affected land and for adjacent areas. Surveys from 10 years ago (Johnson 1978, McNay 1971) listed 292 waste bank fires and 261 underground fires. Data from the 1988 National Abandoned Mine Lands Inventory indicate about 95 underground mine fires (Maynard 1989). In a study of problems associated with anthracite fires (Chaiken 1983), it was observed that successful fire control efforts are expensive and that many fires have required repeated efforts to finally control them. Although there are several fire extinguishment methods, in many cases the high cost (between $250,000 and $100 million per project) and low efficiency (less than 75 pct) are related to the inability (1) to accurately determine the location and extent of the combustion zones within an abandoned mine, and (2) to identify some point at which the fire can be reliably considered extinguished.

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, WV, April 23-26, 1990.

[2]Louis E. Dalverny, Robert F. Chaiken, and Ann G. Kim are Physicist, Supervisory Research Chemist, and Research Chemist, respectively, at the U.S. Bureau of Mines, Pittsburgh Research Center, Pittsburgh, PA.

The emission of smoke and fumes at surface fractures and vents is the conventional indicator of an abandoned mine fire. Because hot gases follow the path of least resistance, the surface evidence of fires may not be related by straight line paths to the source of combustion. Aerial infrared photography determines temperature variations within a few inches of the surface, but is inappropriate when heated areas lie one hundred or more feet beneath the surface. Interpretation of aerial infrared can also be complicated by the presence of heat absorbing surface features. Temperature and gas samples taken at boreholes are point source measurements; their accuracy is limited to a very small volume at the base of the borehole. Because wasted coal fires may be confined to one or more areas of smoldering combustion that may or may not be connected, and because surrounding rock and coal masses can serve as insulating media, it is possible to measure near normal underground temperatures within several feet of a combustion zone. In abandoned mines, submicron particulate detectors have given ambiguous results, probably because of the high moisture content of the mine atmosphere.

Indicators based on the evolution of products of combustion such as $CO_2$, CO, $H_2$, and oxygen deficiency have not given reliable results, probably because changes in these indicators may be produced by processes other than combustion. Also, there may be a reservoir of such gases in which changes in concentration caused by combustion are insignificant. In contrast, previous work by the Bureau (Kim 1974, Kim 1978) has shown that the desorption of low molecular weight hydrocarbon gases from coal is strongly temperature dependent. Changes in the concentration of desorbed hydrocarbons can be detected at temperatures of less than 100° C. The Bureau's Mine Fire Diagnostic method is based on the temperature dependent desorption of low molecular weight hydrocarbon gases from coal and controlled subsurface sampling and is not constrained by the other factors.

In order to locate a fire, it is necessary that (1) a fire have a measurable characteristic, (2) the characteristic be detectable through appropriate sampling methods, and (3) the sampling data are interpreted correctly. In the method discussed here, the measurable characteristics are borehole temperature and pressure and the concentrations of low molecular weight hydrocarbons in the mine atmosphere. Their initial values are determined under natural or baseline conditions. Then, these values are compared to those obtained when a suction fan is used to impose a pressure gradient that causes mine gases to flow radially from the surrounding underground areas toward the point at which the suction is applied. Differences in the concentration of hydrocarbon gases (methane and the C2 to C5 alkanes) at various points underground can be related to the movement of the desorbed gases in assumed directions from the source toward the suction point.

The Mine Fire Diagnostic method has several advantages: (1) changes in hydrocarbon emission are caused only by changes in temperature; (2) changes in hydrocarbon emission can be detected for temperatures below the ignition point of coal, increasing the effective sampling volume and sensitivity of the method; (3) hydrocarbons can be detected at concentrations as low as 1 ppm; (4) the use of an appropriate ratio can eliminate the effects of sample dilution; (5) induced pressure effects usually can be detected several hundred feet from the suction point; and, (6) operation of the system is fairly simple, requiring only readily available equipment and normal technical skills.

This paper includes short discussions of: the laboratory study to develop a data base relating hydrocarbon desorption from various coals to temperature under controlled conditions in the effort to establish measurable criteria for determining actual extinguishment; the resultant development of a ratio (a signature value) based on the concentrations of desorbed hydrocarbons that is characteristic of heated coal; the field method that creates and controls the dynamic sampling environment; and the interpretive method that defines heated and cold subsurface zones. Examples of the use of the Mine Fire Diagnostics method are also given. (Extensive descriptions are found in Dalverny 1990, Kim 1989, and Kim 1989)

## Coal Fire Characterization by Hydrocarbon Signatures

The internal structure of coal is like a sponge with many small interconnected pores (micropores) with an average diameter of 5 to 20 angstroms (Kim 1978). Coal also has a system of cracks or fractures (macropores) which intersect the micropore system. Both the fracture density and the width of the fractures can vary (Kissell 1975). Hydrocarbon gases formed during coalification are adsorbed on both the micropore and macropore surfaces. These gases are a mixture of methane, ethane, propane, iso- and normal butanes and pentanes (Kim 1973, 1973, 1988). The adsorbed gas in coal may also contain small amounts of carbon dioxide and hydrogen, products of coalification; oxygen and nitrogen, occluded during deposition or introduced by percolating ground water; and helium, a product of radioactive decay. Normally, coalbed gas is more than 90 % hydrocarbons. Elevated concentrations of carbon dioxide are frequently associated with areas of tectonic activity. Carbon monoxide and sulfur compounds are not normally found in gas removed from coalbeds. All of the low molecular weight hydrocarbons associated with coal would be in the gaseous state at normal ground temperatures. Under all conditions, some gas escapes from the coal and is emitted into the atmosphere.

The rate of gas emission from coal to the atmosphere is dependent upon temperature, the pressure differential between the coal and the atmosphere, the rank of the coal, the degree of fracturing, the permeability of the coal, and the permeability of adjacent strata. The desorption and emission of hydrocarbons other than methane significantly affect the overall emission rate. Changes in the concentration of desorbed methane and the C2 to C5 alkanes can be detected at temperatures below 100° C.

At normal ground temperatures, the gas drained from coalbeds is over 90 % methane, probably because more methane is formed during coalification and because it requires less energy to desorb the lighter methane molecule. At 150° C, the

concentration of methane is about 35 %. In active mines, changes in the emission rate may be related to changes in surface area due to coal breakage or to changes in barometric pressure. In abandoned mines, temperature is the major factor which affects the emission of hydrocarbons from the coal and carbonaceous shale; hence, the composition of hydrocarbon gases in the mine atmosphere would relate to the temperature of the coal in the mine.

In order to quantitatively relate hydrocarbon desorption to changes in the temperature of coal, the Bureau conducted a laboratory study (Kim 1988) in which duplicate continuous heating tests were completed on samples of six bituminous coals, one bituminous waste, two anthracite coals and two anthracite wastes. Gas samples were taken during heating and cooling phases of the tests and analyzed for oxygen, nitrogen, carbon dioxide, carbon monoxide, hydrogen, and the hydrocarbons: methane, ethane, ethylene, acetylene, propane, propylene, butanes and pentanes. The concentration of total hydrocarbons for each of the bituminous samples, including the waste, was between 100 and 1000 ppm.

The results obtained from the continuous heating tests were dependent to some degree upon the rank of the coal. For bituminous samples, the concentration of methane and the concentration of higher molecular weight hydrocarbons both increased during heating and decreased during cooling. Using these data, a concentration ratio was defined which relates both factors in a single parameter suitable for graphical presentation. By definition, this ratio is:

$$R1 = \frac{1.01[THC] - [CH4]}{[THC] + c} \times 1000$$

where [THC] = concentration of total hydrocarbons, ppm
[CH4] = concentration of methane, ppm.
c = constant, 0.01 ppm

The ratio R1 was designed so that it would (1) equal zero when and only when the concentration of total hydrocarbons was zero, (2) have a unique value when methane was the only hydrocarbon, (3) eliminate the possibility of division by zero (a mandatory consideration in computer processing of data), and (4) increase as temperature increased. As defined, the ratio, R1, is an increasing function of the percentage of higher hydrocarbons, has a value of zero when no hydrocarbons are detected, a value of 10 when methane is the only hydrocarbon, and a limiting value of about 1000. At this time, the only constraint on the use of R1 as an indicator of heated coal is a practical one; namely, that the concentration of methane in the gas samples be greater than 20 ppm. Below this level, the concentration of other C2 to C5 hydrocarbons present could be below the current 1 ppm detection limit of standard GC analytical techniques, and thus, changes in the hydrocarbon concentrations with temperature might not be detected.

Through the laboratory studies, characteristic values of R1 could be defined for the bituminous coal samples. (Unfortunately, it was found that the anthracite coal samples had a completely different behavior; namely, that few $C_2$ to $C_5$ hydrocarbons were liberated at elevated temperatures. This could be due to the very tight micropore structure of anthracite coal relative to bituminous coal.) Generally, the values of the ratio R1 for the bituminous coal can be interpreted on an empirical scale as:

| R1 | Relative Coal Temperature |
|---|---|
| 0 to 50 | Normal (<30° C) |
| 50 to 100 | Possible Heated Coal (30° to 100° C) |
| > 100 | Heated Coal (>100° C) |

It should be emphasized that the absolute value of the R1 ratio is not necessarily uniquely defined by the coal temperature. R1 values during heating are not equal to the R1 values at the same temperature during cooling. The value is influenced by the rank and internal structure of the coal, and by the amount of gas adsorbed on the coal surface. It is however, indicative of an average temperature condition (elevated values of R1 are due only to the presence of heated coal), and over a period of time will increase with increasing temperature and decrease with decreasing temperature.

The use of a hydrocarbon ratio as a fire signature (for bituminous coals) has several advantages: (1) Hydrocarbon desorption occurs at relatively low temperatures; therefore, the mass of the coal from which the hydrocarbons are desorbed is much larger than that directly affected by combustion. Because the source is larger, the signature is more sensitive. (2) Because a ratio is used, and if the concentration is above the detection limit, dilution with air or combustion products is not a factor. The hydrocarbon ratio is independent of the concentration of other components, such as oxygen and carbon dioxide, and temperature dependent desorption is the major mechanism by which hydrocarbons are produced in an abandoned mine. The use of this hydrocarbon ratio produces a positive and sensitive indicator of remote subsurface combustion.

Remote Fire Detection using Communication Testing

The Mine Fire Diagnostic methodology is based on the assumption that a sufficiently large negative pressure (vacuum) applied to underground regions will cause the gases in the mine atmosphere to flow from some distance toward the point of suction. Controlling the movement of underground gases overcomes the limitations of a point source measurement by allowing gases from a large region to be sampled through one borehole. Repeated sampling at all points in a borehole pattern provides data to determine the presence or absence of fire along pathways between the sampling points. Differences in borehole temperature, pressure, combustion products, and desorbed hydrocarbon gases, are assumed to be related to the radial flow of mine gases from the surrounding underground areas toward the point at which the suction is being applied (fig. 1). Figure 2 is a simplified, one-dimensional depiction of this concept with the exhaust fan attached to the left-most hole. If there is communication between the exhaust hole and the neighboring holes, a decrease in static pressure will be measured at the bottom of each

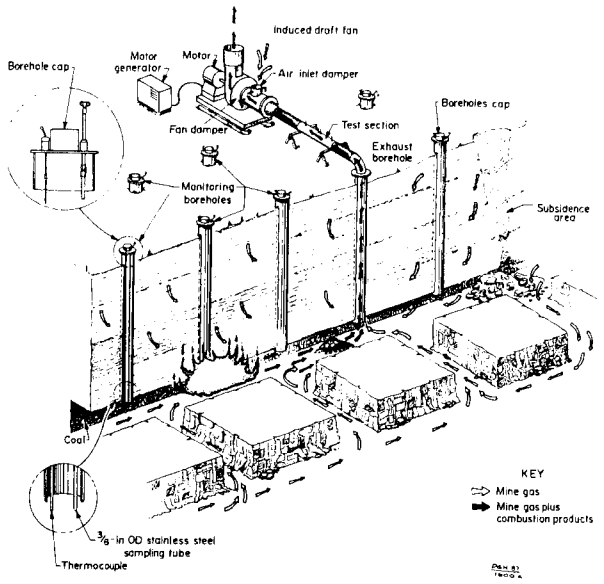



Figure 1. Artist's Representation of Mine Fire Diagnostic Technique.

casing. If a fire exists to the right of the right-most hole (Case A), fire signature gases will be detected in gas samples taken at each borehole. Similarly, a heating between the exhaust hole and its nearest neighbor (Case C), would be detected only at the exhaust hole. To verify the location of heating in an actual mine, the fan is moved to various boreholes set in a pattern and the communication test is repeated. The specific fire location is then determined by overlapping the signature data in a manner similar to triangulation.

The successful determination of combustion zones depends upon the ability to force and detect the movement of gases from these zones. Radially induced pressure effects are detectable as far as several hundred feet from a point of suction. The surrounding boreholes are capped during suction to control dilution by air. If the region is highly fractured or faulted, air will be drawn in from the surface. In most cases however, the use of concentration ratios, which are independent of dilution, will still be reliable indicators of the presence or absence of fire, except when the incoming air reduces the hydrocarbon concentration below the detection limit.

In the first step of the testing procedure, a series of boreholes is drilled and cased to depths consistent with coal bearing strata. Borehole locations are based on surface evidence of the fire, mine maps if available, and efforts to achieve a regular pattern on approximately 100 ft. centers. Although various sizes of boreholes have been used, 8-in. I.D. casing is preferred. With smaller diameter casings, frictional resistance to air flow can cause most of the fan's capacity to be used to move air from the bottom to the top of the casing. With larger diameter pipe, the lower resistance to air flow allows the fan-induced vacuum to affect a larger underground area. Also, there is more room in the larger pipe for lowering monitoring equipment such as water sample bottles, cameras, pressure transducers, sampling lines and thermocouples.

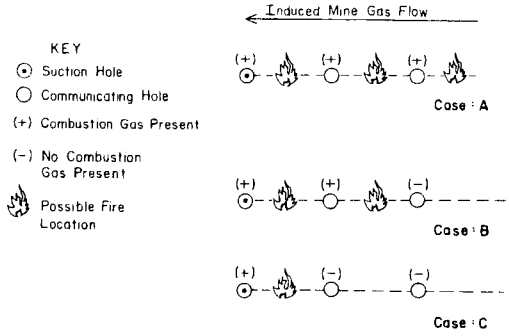



Figure 2. Schematic of Mine Fire Diagnostics

Tests in the first series of holes are used to define heated areas in general. The results from the first round of testing are then used to determine the placement of additional boreholes. A second round of tests is performed to verify the location of heated zones and to establish the presence of a cold boundary. The number of tests in each round is determined by the number of boreholes used as fan attachment points. Depending on the site conditions, some holes may not be used as suction points, and some may be used more than once.

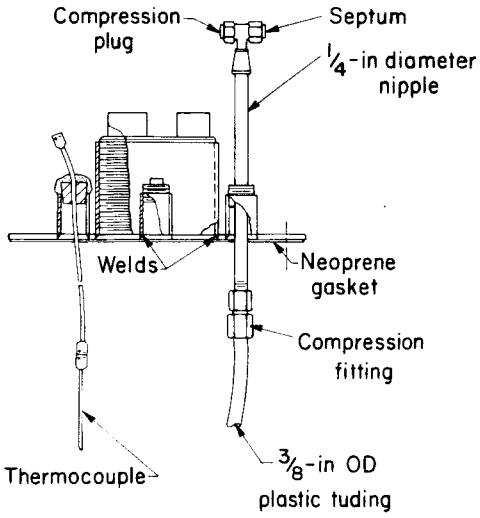



Figure 3. Instrumented Borehole Cap.

The borehole instrumentation cap (fig. 3) holds temperature and pressure probes and a gas sampling tube at or near the bottom of the borehole. To measure the borehole pressure and to obtain gas samples, a 3/8-in. O.D. plastic sampling tube is connected to a tee and septum arrangement at the borehole cap. This tube extends approximately 6 to 12-in. below the bottom of the casing. One side of the tee is capped with a compression plug which is removed for attachment to the gas sampling pump; the other end contains a septum to allow for needle connection to a pressure gauge. Pressure measurements are made with a Magnehelic[3] pressure gauge. Gauge measurements are accurate to +/- 0.005-in. w.c. One port of the pressure gauge is attached to the sampling tube, via a length of 1/4-in. tubing and a hollow needle arrangement, at the borehole cap's septum port. The septum allows

measurement of the pressure in the sample tube without disturbing or contaminating the gas within the tube. The other gauge port is left open to the atmosphere. Changes in pressure during communication tests establish whether the suction fan is affecting the area around the borehole. Normally, pressure changes in the range of 0.005 to several inches w.c. are measured. A diaphragm pump (MSA model S) capable of 0-10 l/min. at 20 to 40-in. w.c. is used to sample the mine gases. An inlet line with a Y-connector and needle arrangement is attached to the sampling port on the tee in the borehole cap. Once the sample line has been purged, an evacuated tube (Vacutainer) is used to collect the gas sample on the outlet side of the pump. It should be noted that the commercially available Vacutainers are not completely evacuated, and must be completely evacuated prior to their use for collecting field gas samples. The use of this type of evacuated tube to store gas samples is preferred over other methods because of the ease in sampling, storage, and transport. The use of the diaphragm pump and the small evacuated glass tubes for gas sample collection has proven to be an easy and efficient method of obtaining uncontaminated gas samples from subsurface areas.

Data Interpretation

Before the fan is turned on, baseline data and samples are taken. When the fan is turned on, at least one hour is allowed to elapse for subsurface conditions to reach equilibrium; then, temperatures and pressures are recorded and gas samples taken at all boreholes. A communication test generally requires three to four hours to complete, depending on the number of boreholes to be sampled.

A set of communication studies produces baseline data (ambient conditions) and communication data (suction applied); both data subsets include static pressure, temperature and gas concentrations at each borehole. The static pressure differences at the bottom of the boreholes of at least 0.01 in. w.c. that occur as a result of suction indicate communication between the boreholes and the suction hole. They are used to construct a pressure contour for each communication test. This contour provides a "window" of good communication and data taken outside this region are disregarded due to probable insufficient gas movement. Temperature measurements provide minimal information with this method, but may indicate the amount of cooling or heating caused by air injection. A flow probe in the intake piping of the fan is used to calculate the total flow through the fan. Pressure and flow in the duct and the pressure changes measured at the boreholes are indicators of the extent to which the applied suction is effective.

Differences between baseline and communication measurements are calculated for the windowed area. These changes at each borehole can be classified as: (1) an increase in RI (heating), (2) a decrease in RI (cooling) or (3) no change during suction tests. To delineate the heated and cold zones, it is assumed that increased hydrocarbons originated in a heated area which lies on a straight line drawn through the borehole and the suction point. Similarly, a decrease in hydrocarbons indicates a non-combustion or cold zone in the same location. Each communication test provides information for a limited area adjacent to each communicating borehole. The suction fan is assumed to cause mine gas to flow along a straight line between the borehole and the suction hole. For each communicating borehole in each test, the change due to communication (increase, decrease, or no change) is drawn on a scaled site map as a 90-degree

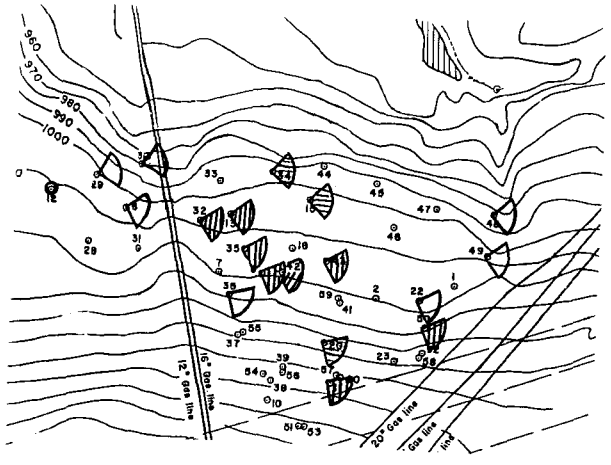



Figure 4a. Mine Fire Diagnostics Communication Test: Exhaust Borehole # 12.

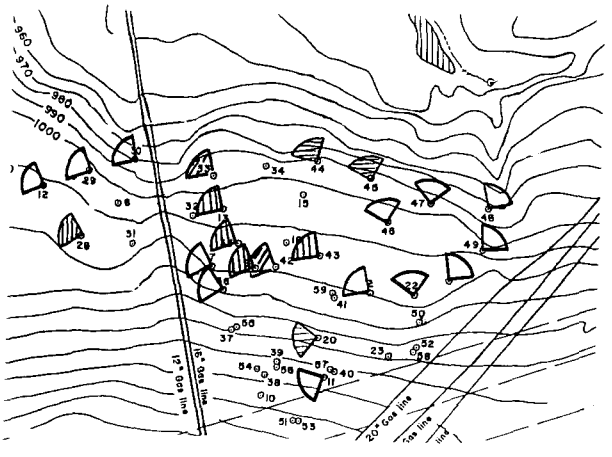



Figure 4b. Mine Fire Diagnostics Communication Test: Exhaust Borehole # 50.

quadrant having a radius of 50-ft.[4] and centered on this line (fig. 4a and 4b). The quadrant apex is centered at the borehole and the quadrant appears to be pointing toward the suction borehole. With this quadrant size, a 2000-sq.-ft. area is defined for each communicating borehole per test. By overlapping the possible combustion and noncombustion regions for all tests, a composite map is made (fig. 5). Alternatively, the quadrants can represent areas that have values of (a) $CH_4$ <= 20 ppm, (b) 9 <= RI < 50, (c) 50 <= RI < 100, or (d) RI > 100, as determined from analyses of the gas samples obtained during the second round of sampling while the fan is operating.

The diagnostic method's accuracy is directly related to the number of communication tests performed. When collating the quadrants from all

[3]References to trade names are given for identification only and do not imply endorsement by the Bureau of Mines.

[4]Represents the average distance between any two adjacent boreholes.

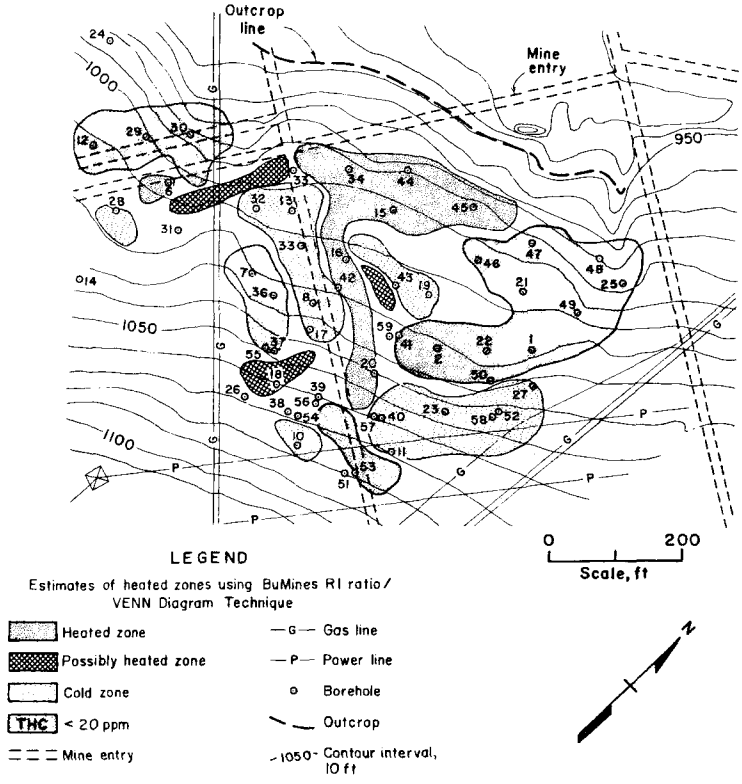



Figure 5. Mine Fire Diagnostics Communication Tests: Composite Results, Large, PA, Mine Fire Site.

the tests on the site map, these guidelines are followed:

(1) When quadrants of the same type (i.e., hot or cold or same RI interval value) overlap, the total area includes both quadrants;
(2) When quadrants of different types overlap, the overlapping area is undefined unless additional tests support either (hot/cold or RI interval value) of the defined quadrants;
(3) When the quadrants do not overlap, but the nearest quadrants are commonly defined, the region between the two is considered to be contiguous. At this point data are rechecked to make sure that no data have been overlooked, and an attempt is made to define and/or explain any area in which the method gives ambiguous results.

Estimates of possible fire zones are mapped for each test; overlapping the results of all tests effectively bounds the possible fire zones through successive approximation. This technique defines probable combustion zones, zones that show no signs of combustion (an important point in defining a fire area), and zones for which the data are inadequate to make a determination.

The diagnostic procedure used to locate underground fire zones is based upon two primary assumptions: (1) changes in the concentration of hydrocarbon gases are due only to the presence or absence of heated coal and (2) the suction fan influences the underground flow of gases to a measurable degree. Corollary assumptions are that the measured pressure gradient can characterize the communication effect at each borehole as good, poor, or none; that underground gas flows follow a straight line path toward the suction point; that the depth of the borehole casing has no effect on the measured pressures or on the gas concentration; that ambient conditions have no effect; and that changes in gas concentration are measurable. In reality, these corollary assumptions of the underground conditions do not always hold. However, if the induced suction creates a measurable pressure gradient, and if changes in gas concentration are measurable, then repetition of the communication tests should minimize the effects of non-ideal subsurface conditions.

## Field Studies

Renton Mine Fire Project. Under the auspices of an interagency agreement with the OSMRE, the Bureau initially investigated the Renton, PA, abandoned mine fire site to determine the location and extent of the combustion. The site encompassed a 60 acre region, and was characterized by venting areas and several large subsidence holes in addition to a large subsidence area (Dalverny 1990). The mine was located in the Pittsburgh coal seam beneath a hill. A one-million gallon water tank was located on the top of the hill, and there were residential dwellings on and near the hill. The overburden varied between 0 and 102 ft., with the average being 42 ft. The mine had been abandoned in 1914, and no mine maps were available. The 129 boreholes (fig. 6) were installed in four rounds of drilling. They were placed primarily for purposes of locating the fire and monitoring the extinguishment, but many were later used as water injection or grout mixture injection points.

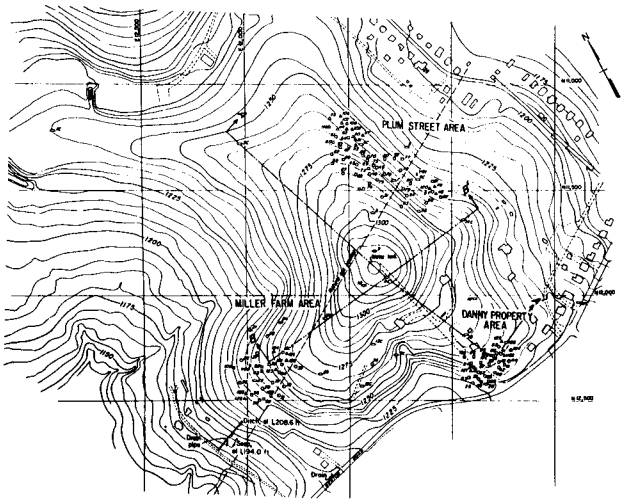



Figure 6. Renton, PA, Project Site Map with Locations of Numbered Boreholes.

At Renton, communicating boreholes, as indicated by a change in downhole pressure of at least 0.02-in. w.c., were separated by distances up to 700 ft. The typical effective distance between the suction point and the boreholes was 250 to 300 ft. The use of the Mine Fire Diagnostic method located three non-contiguous combustion zones, totaling approximately 10 acres. Analysis indicated that the area under the water tank was cold. Subsequently, the Mine Fire Diagnostic method was used to monitor the progress of the extinguishment effort that employed water injection with fume exhaustion as the extinguishment method. In this method (Chaiken 1984), water was injected into the mine to be converted to steam in the combustion areas. A suction fan would move the steam through the mine and remove the moisture and heat. Analysis of the diagnostic data from this time period showed that combustion activity decreased in one area, was unaffected in another area, and increased in the third area (fig. 7).

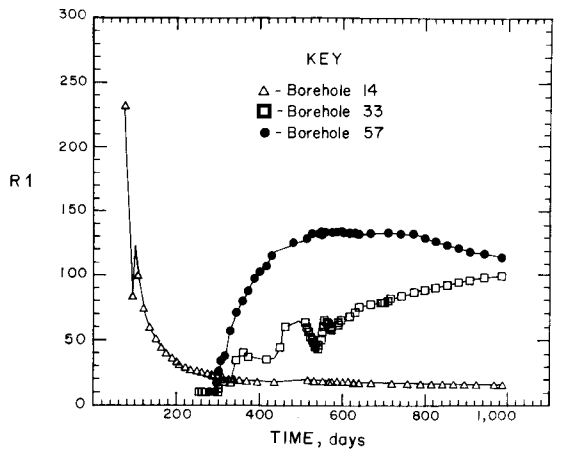



Figure 7. Variation in Baseline Values of R1 with Time at Three Boreholes, Renton, PA, Mine Fire Site.

The Mine Fire Diagnostic technique was important for the Renton project in that it was able to delineate three separate combustion regions. It was also possible to show which areas of the heated zones were being successfully quenched, and as important (or possibly more so), which heated regions were not quenched and but, rather, were spreading.

Large Mine Fire Project. At Large, Pa. a fire is located in an abandoned underground mine in the Pittsburgh coal seam with a maximum overburden of 150 ft. Entries to the mine were located at the base of a steep hill (Justin 1984). A natural gas pumping station is located on the top of the hill and high pressure gas transmission pipes and high voltage electric transmission lines cross the 15 acre site. Because of the terrain and the utilities on the surface, it was considered essential to determine the exact location of the combustion areas. Under a cooperative agreement with PA/DER, the Bureau is using the Mine Fire Diagnostic methodology to delineate heated areas and to establish the location of a cold boundary. Instrumented borehole caps were placed on seven 2-in. boreholes that had been drilled at the site during a previous project. These holes were used for baseline monitoring and as sampling points during communication testing. Seven 8-in. holes were drilled and cased for preliminary communication studies. A second set of 25 8-in. holes was drilled, based on the results of the first communication tests. Suction fan effects were measured as far as 800 ft. from the exhaust borehole.

Surface effects and some historical information implied that the burning was occurring along the buried outcrop. Diagnostic testing, utilizing seven 2-in. boreholes and 32 8-in. boreholes on approximately 100 ft. centers, indicated an L-shaped combustion zone. Figure 5 depicts the composite map drawn from the R1 interval values. The base of the L was near the outcrop; the leg of the L extended into the abandoned mine, probably along a set of main entries. A small isolated combustion zone was located several hundred feet from the primary combustion area. There were several areas in which hydrocarbon concentrations were very low, below the 20 ppm limit considered necessary for the accurate determination of combustion signatures. These areas coincided with areas in which water was noted in some boreholes. If water vapor condensed on the surface of the coal, it could block the desorption of hydrocarbons, thus preventing the determination of a combustion signature. However, if there were sufficient water on the coal to prevent the desorption of hydrocarbons, the temperature of the coal could be assumed to be below the combustion point.

The testing at Large has indicated that long term baseline (without suction) monitoring of R1 will detect changes in subsurface conditions. Hydrocarbon ratio values determined during baseline monitoring have shown that combustion activity is affecting a small isolated area to the left of one gas line right of way. From the baseline data, changes in the hydrocarbon ratio were observable prior to changes in temperature or the concentration of other combustion products (fig. 8). Additional communication testing is planned to establish the location of the cold boundary in the interior of the mine. Eighteen more 8-in. cased boreholes have been drilled and instrumented, as have two 2.5-in. core holes placed toward the interior of the mine. The fact of water pooling in some of the boreholes adds complexity to the

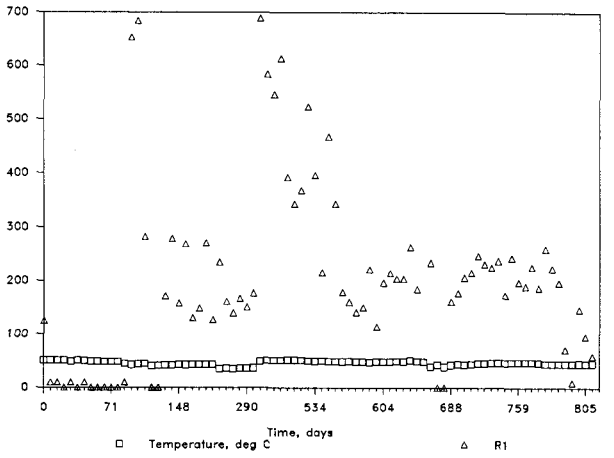



Figure 8. Baseline Values of R1 and Temperature, Borehole # 6, Large, PA, Mine Fire Site.

diagnostics analyses. Although holes bored into coal pillars can still be used during communications testing, water filled holes prevent accessing gases being drawn through the vicinity. Nonetheless, these holes can be monitored for temperature to aid in determining the quantity of heat energy stored in the strata prior to any extinguishment activities. The Large mine fire is an example of the inaccuracy of surface evidence of fire. An observed line of venting may parallel the fire zone along the outcrop, but gives no indication of the combustion areas in the interior portions of the mine. Borehole temperature data (fig. 8) when taken alone would yield misleading results with regard to the location of the combustion zones.

## Summary and Conclusions

Laboratory results and field studies support the effectiveness and applicability of the Bureau's Mine Fire Diagnostic technique to the problems of locating and monitoring abandoned mine fires. For the bituminous coal samples studied, values of the concentration ratio, R1, increased with increasing temperature and decreased during cooling. Although values of R1 do not correspond to a particular temperature, elevated values of R1 are due only to the presence of heated coal. Time dependent monitoring of changes in R1 reflect changes in the average temperature of the coal.

The Mine Fire Diagnostic method incorporates a sampling method that increases the detection zone of normal point source measurements through a gas movement scheme. Measuring one of the characteristics of this moving gas, in this case changes in hydrocarbon concentrations or in a ratio of the concentration values, and plotting the results as vectors (magnitude and direction) rather than point source (magnitude) measurements, expands and bounds the area(s) affected by combustion, as well as the area(s) not affected by combustion. These factors make the Bureau's methodology a significant improvement in locating and monitoring abandoned mine fires.

## Acknowledgements

The development of the Mine Fire Diagnostic methodology was performed by the Bureau of Mines with funding from and in cooperation with the Abandoned Mined Land program of the Office of Surface Mining Reclamation and Enforcement (OSMRE) and with the Pennsylvania Department of Environmental Resources (PA/DER).

## Literature Cited

Chaiken, R.F., R.J. Brennan, B.S. Heisey, A.G. Kim, W.T. Malenka, and J.T. Schimmel. 1983. Problems in the Control of Anthracite Mine Fires: A Case Study of the Centralia Mine Fire (August 1980). BuMines RI 8799, 93 p.

Chaiken, R.F., E.F. Divers, A.G. Kim, and K.E. Soroka. 1984. Calamity Hollow Mine Fire Project (In Five Parts). 4. Quenching the Fire Zone. BuMines RI 8863, 18 p.

Dalverny, L.E. and R.F.Chaiken. 1990. Mine Fire Diagnostics and Implementation of WIFE at Renton, PA. BuMines RI (in press).

Justin, Thomas R. and Ann G. Kim. 1988. Mine Fire Diagnostics to Locate and Monitor Abandoned Mine Fires. p. 348-355. In Proc. Mine Drainage and Surface Reclamation Conf. BuMines IC 9184, V.II.

Johnson, W. and G.C. Miller. 1978. Abandoned Coal-Mined Lands: Nature, Extent, and Cost of Reclamation. BuMines Spec. Publ. 6-79, 29 p.

Kim, A.G. 1973. The Composition of Coalbed Gas. BuMines RI 7762, 9 p.

Kim, A.G. and L.J. Douglas. 1973. Gases Desorbed from Five Coals of Low Gas Content. BuMines RI 7768, 9 p.

Kim, A.G. 1974. Low Temperature Evolution of Hydrocarbon Gases from Coal. BuMines RI 7965, 23 p.

Kim, A.G. 1978. Experimental Studies on the Origin and Accumulation of Coalbed Gas. BuMines RI 8317, 18 p.

Kim, A.G. and F.N. Kissell. 1988. Methane Formation and Migration in Coalbeds. p. 21-25. Ch in Methane Control Research: Summary of Results, 1964-1980. BuMines Bull. 687.

Kim, A.G., L.E. Dalverny, and T.R. Justin. 1989. Mine Fire Diagnostics, Recent Developments. p.151-170. In Proc. The Evolution of Abandoned Mine Land Technologies: A Symposium. (Riverton, WY, June 14-16, 1989).

Kim, A.G. 1989. Laboratory Determination of Signature Criteria for Locating and Monitoring Abandoned Mine Fires. BuMines RI (in press).

Kissell, F.N. and J.C. Edwards. 1975. Two-Phase Flow in Coalbeds. BuMines RI 8066, 16 p.

Maynard, B.R. 1989. Mine Fires. The 11th Annual AML Conference. (Williamsburg, VA, Oct. 16-19, 1989).

McNay. L.M. 1971. Coal Refuse Fires, An Environmental Hazard. BuMines IC 8515, 50 p.

# RELATIVE SELF-HEATING TENDENCIES OF COAL, CARBONACEOUS SHALES AND COAL REFUSE[1]

by

Ann G. Kim and Robert F. Chaiken[2]

Abstract. Fires in abandoned coal mines and waste banks present serious safety and environmental hazards. Often, spontaneous combustion is assumed to be the source of these fires, although some coals have an apparently low susceptibility to self-heating. It has also been noted that in abandoned mines evidence of heating is found in the roof coal and shale. Spontaneous ignition is also a factor in coal waste bank fires. Although the spontaneous combustion of coal has been studied extensively, there is very little information about the self-heating tendency of roof coals and carbonaceous shales.

This study of self-heating tendencies utilized a modified version of a differential thermal analyzer. In this study, a combustion furnace served as the reference temperature against which the sample temperature was measured. A crossing point temperature index was used to relate self-heating tendencies of various samples. Analysis of the effluent gas for $CO_2$, CO, $O_2$ depletion and desorbed hydrocarbons was also used to evaluate the degree of self-heating and the heat generating process. Average sample size was 3 kg; particle size was between -1/4 in and +20 mesh. Furnace temperature was between 20 and 200° C. The air flow rate was 2 scfh, and each sample was heated in moist and dry air. Ten samples, including the Pittsburgh coal, roof slate, carbonaceous shales and western coals were tested. The crossing point index was compared to known self-heating behavior, reaction rates, BTU values and compositional differences.

Additional key words: Spontaneous combustion, Abandoned mine fires, Waste bank fires.

## Introduction

The Bureau of Mines' Environmental Technology Group at the Pittsburgh Research Center is involved in an integrated program of research on the control of fires in abandoned mines and waste banks. These fires present serious safety and environmental hazards. Extinguishing such fires by conventional methods is expensive, difficult and frequently ineffective (Chaiken, 1983). Often, spontaneous combustion is the apparent cause of a fire in an abandoned mine even though the coal in question has a low susceptibility to spontaneous combustion. It has also been noted in many fire control projects that evidence of heating is found in the roof coal, and that the roof coal and shale may serve as continuous paths for spreading the fire. Coal refuse is also known to be susceptible to combustion, often when no apparent ignition source is determined.

The self-heating tendency of roof coals and associated carbonaceous shales is a parameter in determining how fires are spread in abandoned coal mines and has a direct bearing on the application of fire control techniques. If fires start or are spread along roof coals and shales, then any method directed toward controlling a fire in the mine void or collapsed areas will not be effective (Dalverny, 1989). For instance, methods using the heat transfer capabilities of water or other agents depend on delivering the agent to a heated zone. If the heated zone exists in the roof of the mine, any procedure which delivers the agent only to the mine void is biased toward failure.

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2] Ann G. Kin is Research Chemist and Robert F. Chaiken is Supervisory Research Chemist at Pittsburgh Research Center, U.S. Bureau of Mines, Pittsburgh, PA 15236.

Coal is defined as "a readily combustible rock containing more than 50 % by weight and 70 % by volume of carbonaceous material..... (Schopf 1966)." The non-carbonaceous matter in coal, mineral matter, is the inorganic non-combustible compounds which form ash and the sulfur compounds. According to ASTM standards (American Society for Testing and Materials, 1983), the lowest rank coals have more than 48 % fixed carbon and a heating value of less than 8300 BTU. Carbonaceous shales are sedimentary rocks which probably developed from peats containing less than 50 % organic material (Jones and Cameron 1988). Coal refuse consists of waste coal, slate, carbonaceous shales, pyritic shales, clays and other strata associated with the coal seam. The combustible content of this material averages between 2000 and 6000 BTU.

As with any fire, fires in abandoned mines or coal waste banks require three elements: fuel, oxygen and an ignition source. In coal combustion, the fuel is the carbon in the coal. In room and pillar mining as much as 30 to 50 % of the mineable coal remains in the abandoned mine. A layer of coal near the mine roof is frequently left in place for additional support during mining. It eventually fractures and collapses after mining ceases. The material in waste banks may have a combustible content of 50 % or more.

If combustion is considered the exothermic reaction of carbon and oxygen to form carbon dioxide, written as:

$$C + O_2 \quad ----> \quad CO_2 + \text{heat},$$

then the amount of heat liberated is -93.7 kcal/mole. However, coal is not composed of elemental carbon. On a dry, mineral matter free basis, coal contains between 60 and 90 % carbon. The rest of the coal "molecule" is composed of hydrogen, oxygen, nitrogen, and sulfur. The combustion reaction for an average bituminous coal can more accurately be written as (Chaiken 1977):

$$CH_{1.18}N_{0.15}O_{0.35}S_{0.005} + 1.12O_2 \quad --->$$

$$CO_2 + 0.58H_2O + .005SO_2.$$

This reaction is exothermic, producing more energy than it consumes (from 5 to 10 kcal/g of coal) depending on the rank of the coal. The oxidation of coal occurs constantly. The temperature of the coal is a function of the rate of heat generation versus the rate of heat loss. When these processes occur at the same rate, the temperature of the coal remains constant. When the rate of heat generation is greater than the rate of heat loss, the temperature of the reacting system increases. Since the rate of heat generation is an exponential function of temperature and the rate of heat loss is a linear function of temperature, as the temperature increases, the reaction rate increases faster than the heat loss rate. Ignition, or self-sustaining combustion, will occur when the rate of heat generation exceeds the rate of heat loss (Kanury 1975). Ignition is therefore, a function of the amount of energy released by a reaction and the rate at which it is released, as well as the rate at which energy is transferred from the reacting mass to the surroundings. The reaction rate is a function of the concentration of reactants, carbon and oxygen, the surface area, particle size, temperature and activation energy.

In spontaneous ignition, there is no external heat source; natural reactions supply sufficient energy to sustain combustion. Spontaneous combustion in coal, carbonaceous shales or coal refuse is related to the oxidation of the coal to form $CO_2$ and CO. The oxidation of pyrite and the adsorption of water on the coal surface also are exothermic or heat generating reactions which increase the probability of spontaneous combustion. Thermophillic bacteria may also contribute to raising the temperature of the coal. In waste banks, most of the oxygen diffusing from the surface is consumed by bacterial activity within a few feet. However, enough oxygen is available at depth to support combustion.

The normal ignition temperature for coal is between $420^{o}$ and $480^{o}$ C. Under adiabatic conditions (all heat generated is retained in the sample), the minimum temperature at which a coal will self-heat is $35^{o}$ to $140^{o}$ C (Smith & Lazarra 1987). Normal underground temperatures are $11^{o}$ C or less, and ambient air temperatures are usually below $35^{o}$ C. To reach temperatures at which combustion is self sustaining, heat generation must be greater than heat loss. In most abandoned mines and waste piles, conditions favor the retention of heat. As noted above, there are several exothermic reactions which release energy. Heat is lost by convection or conduction. Since abandoned mines and waste piles have an essentially stagnant atmosphere, convection accounts for very little heat loss. Most heat transfer is probably by conduction to surrounding strata. Rocks tend to be good insulators and retain heat within the mine or waste bank. Self-heating as the source of wasted coal fires is the cumulative effect of heat generation and heat removal mechanisms.

The spontaneous combustion of coal has been extensively studied in the United States, Europe and the Soviet Union, for active mines, abandoned mines and for rail and water transportation methods. Studies of self-heating of coal refuse have been primarily limited to methods of reducing the occurrence of fires in active coal waste banks. There is essentially no information dealing with the self-heating tendency of roof coals or carbonaceous shales associated with coals, yet these are the materials that comprise the fuel for spontaneous heating in abandoned mines and waste banks.

The current study is intended to determine the relative combustibility of coal, coal waste, roof coal and carbonaceous shales. Specifically, it is intended to describe some characteristic thermal behavior that can be related to the probability of self-heating. Also the study attempts to relate self-heating to the physical and/or chemical characteristics of the sample, and differentiate heat contributions (+/-) from reactions, physical processes, and heat transfer.

## Materials and Methods

This experiment was designed to determine the cumulative effect of heat generation and heat loss in a relatively short period of time. It used a relatively large sample at controlled temperatures and was not oxygen limited. To determine the relative self-heating tendencies of coals, roof coals and carbonaceous shales, a version of differential thermal analysis was used.

In differential thermal analysis, a temperature difference is measured between a sample and a standard. In conventional differential thermal analysis, two sample compartments within a thermal block, one filled with the sample, the other filled with an inert substance (usually sand), are heated. As the temperature of the block is raised, the difference between the two materials is measured. The point at which the temperature of the sample exceeds the temperature of the standard (the crossing point temperature) is taken as a measure of the sample's tendency to self-heat. Generally, sample size is a few grams, the atmosphere is stagnant and heat loss is minimal.

This study used an 8-in. diameter furnace as the differential thermal analyzer. The furnace temperature was the standard against which the difference in the sample temperature was measured. In this study, the sample size was approximately 3000 g, crushed and screened to -1/4 in. by +20 mesh. The samples were dried overnight under nitrogen at 100° C to remove surface moisture. The samples were packed into the sample holder to create an approximately homogeneous bed. Positive displacement air flow, at a rate of 2 scfh, insured an even distribution of air in the sample. Duplicate tests with wet and dry air were run to determine the effect of water adsorption. Dry air was obtained by passing compressed breathing air through a tube of Drierite[3]. In the moist air tests, the air was bubbled through water immediately prior to the furnace inlet.

Each sample was heated to 100° C, level 1, and held at that temperature for approximately 24 hours. The furnace temperature was then raised to 150° C for 24 hours, level 2, and then to 200° C, level 3. The rate at which the sample heated to the furnace temperature was determined and the temperature differential between the sample and the furnace was measured.

A gas sampling line was installed in the center of the furnace. A small pump was used to purge the line through a T connection, and the gas sample collected in Vacutainers by puncturing the septum in the T. Evolved gases, CO and $CO_2$, were determined by gas chromatographic analysis against a standard. Oxygen consumption was determined as the difference between the concentration in the furnace gas sample and that of the influent air. All solid samples were analyzed for proximate, ultimate and BTU values (Table 1). Forms of sulfur were determined for four of the shale samples.

The F Seam sample was a lower rank Wyoming coal which was known to self-heat. The coal sample, Blue Creek, and the 4 shales labeled Zone A, B, C and D were from a mine which was experiencing self-heating, apparently from the shales in the floor. The shales were channel samples taken from undercasts, with A being closest to the mine floor. The four shale layers were separated by rock. The samples C1, C2 and D1, D2 were samples of the C and D layers from different areas of the mine. Albright (ALB) was a refuse sample from a waste bank which was burning; no apparent cause of the fire had been determined. The Bailey sample was a near surface sample from an active waste bank. The sample of the Pittsburgh coal (PGH) had been obtained from the Bruceton Experimental Mine.

[3]References to trade names are given for identification only and do not imply endorsement by the Bureau of Mines.

## Results and Discussion

In this version of the crossing point method, the extent to which the sample temperature exceeds the furnace temperature is assumed to be indicative of the tendency of the sample to self-heat. Crossing point temperatures were measured at three oven temperatures and a composite index was developed.

For each sample, the furnace temperature and the sample temperature were plotted against elapsed time (Fig. 1). At 100° C, level 1, the temperature of the sample increased until it equaled or exceeded the furnace temperature. The temperature of the sample then would return to the furnace temperature. This indicated sufficient heat generation to raise the sample temperature, followed by heat loss and cooling of the sample. The same pattern was observed at 150° C, level 2. At 200° C, level 3, some samples exhibited the previously observed heating/cooling cycle. For other samples, the temperature increased quickly and did not decrease, indicating that the sample temperature was increasing toward the point of thermal runaway. Normally the temperature of the furnace was held at the designated level for 24 hours. If the sample temperature at level 3 increased by approximately 40 ° C in less than eight hours, the test was terminated to prevent runaway heating and the deposition of coal tars in the furnace.

The maximum temperature differential (Table 2) between the set temperature and the sample temperature was determined at temperature levels 1, 2, and 3 (100°, 150° and 200° C). In those tests which were terminated early due to sustained self heating at level 3, the maximum temperature of the sample was often lower than that of samples which were heated for the standard 24 hours. For example, the maximum temperature of Run # 30 was 281° C at the end of the 24 hour period. Run # 24 was terminated when the sample temperature reached 264° C within 9 hours. For those runs which were ended in less than 24 hours at level 3, an arbitrary 50 ° C was added to the maximum temperature differential and designated $dT_{max}+$.

These values, in Table 2, were used to determine a self-heating rank for levels 1, 2 and 3 (Table 3).

At level 1 (100° C), the maximum temperature differential was between 0 and 9° C. At level 2 (150° C), the maximum temperature differential was between 1 and 56° C, and at level 3 (200° C) it was between 10 and 81° C. The six runs with a "+" in the level 3 column were those which apparently would have reached ignition temperature if the tests had not been terminated. The samples were ranked at levels 1, 2, and 3+ according to the maximum temperature differential at each level. A composite rank from the sum of the values at levels 1, 2 and 3+ was also determined (Table 3). All ranks are given from 1 to 22 with 1 indicating the highest temperature differential, indicative of the greatest degree of self-heating, and 22 being the least.

Table 1: SAMPLE ANALYSIS, As-received, wt pct

| SAMPLE NAME | SAMPLE TYPE | MOIST | ASH | VOLATILE MATTER | FIXED CARBON | TOTAL SULFUR | BTU/LB |
|---|---|---|---|---|---|---|---|
| ALBRIGHT | Refuse | 4.79 | 57.50 | 12.17 | 25.53 | 7.34 | 4606 |
| BAILEY | Refuse | 2.18 | 79.76 | 12.03 | 6.03 | 1.56 | 1573 |
| F SEAM | Coal | 9.14 | 4.04 | 40.86 | 45.96 | 0.51 | 11918 |
| BLUE CREEK | Coal | 0.77 | 36.16 | 19.16 | 43.91 | 0.70 | 9422 |
| PITTSBURGH | Coal | 1.40 | 4.76 | 37.36 | 56.48 | 1.01 | 14205 |
| ZONE A | Shale | 0.31 | 79.96 | 17.66 | 5.07 | 0.29 | 1573 |
| ZONE B | Shale | 0.64 | 59.98 | 15.20 | 27.18 | 0.46 | 5925 |
| ZONE C | Shale | 0.94 | 75.28 | 10.51 | 13.27 | 2.82 | 2725 |
| ZONE D | Shale | 1.04 | 61.58 | 14.26 | 23.12 | 4.74 | 4841 |

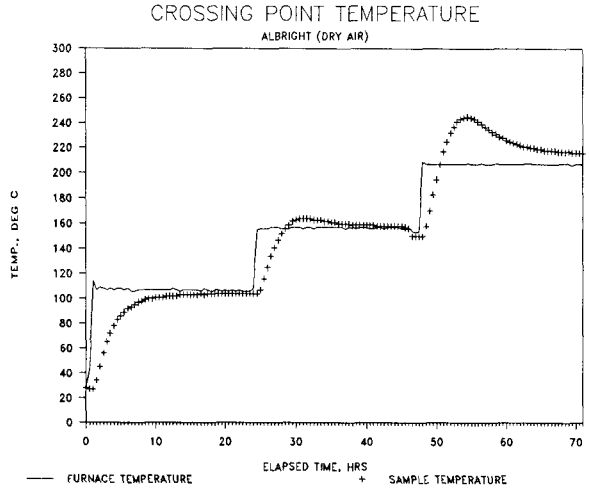



Figure 1. Crossing Point Temperature, Modified

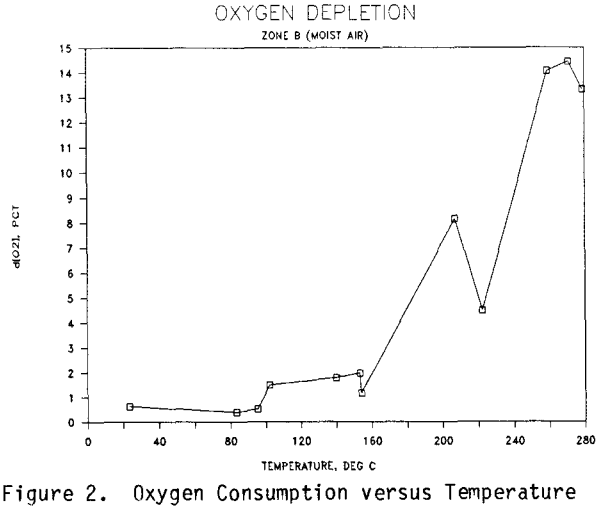



Figure 2. Oxygen Consumption versus Temperature

Table 2: MAXIMUM TEMPERATURE DIFFERENTIAL, Deg C

| RUN # | SAMPLE | AIR D/M[1] | dTmax LEVEL 1 | dTmax LEVEL 2 | dTmax+ LEVEL 3 | dTmax+ TOTAL |
|---|---|---|---|---|---|---|
| 25 | ALB | D | 4 | 14 | 45 | 63 |
| 26 | ALB | M | 3 | 9 | 19 | 31 |
| 11 | BAILEY | D | 0 | 6 | 20 | 26 |
| 12 | BAILEY | M | 3 | 7 | 26 | 36 |
| 23 | F SEAM | D | 9 | 56 | 129+ | 194 |
| 24 | F SEAM | M | 8 | 42 | 114+ | 164 |
| 30 | BLUE CREEK | D | 5 | 25 | 81 | 111 |
| 29 | BLUE CREEK | M | 1 | 15 | 70 | 86 |
| 28 | PGH | D | 1 | 10 | 63 | 74 |
| 27 | PGH | M | 3 | 13 | 63 | 79 |
| 13 | ZONE A | D | 2 | 3 | 16 | 21 |
| 14 | ZONE A | M | 4 | 12 | 54 | 70 |
| 15 | ZONE B | D | 2 | 6 | 25 | 33 |
| 17 | ZONE B | M | 4 | 6 | 23 | 33 |
| 22 | ZONE C2 | D | 2 | 7 | 32 | 41 |
| 16 | ZONE C1 | D | 3 | 10 | 43 | 56 |
| 21 | ZONE C2 | M | 4 | 1 | 10 | 15 |
| 18 | ZONE C1 | M | 4 | 22 | 124+ | 150 |
| 32 | ZONE D2 | D | 3 | 14 | 105+ | 122 |
| 19 | ZONE D1 | D | 2 | 6 | 24 | 32 |
| 20 | ZONE D1 | M | 3 | 11 | 98+ | 112 |
| 31 | ZONE D2 | M | 4 | 25 | 111+ | 140 |

[1]D: Dry air M: moist air

Table 3: SELF-HEATING RANK

| RUN # | SAMPLE | AIR D/M | RANK LEVEL 1 | RANK LEVEL 2 | RANK LEVEL 3+ | RANK TOTAL + |
|---|---|---|---|---|---|---|
| 23 | F SEAM | D | 1 | 1 | 1+ | 1 |
| 24 | F SEAM | M | 2 | 2 | 3+ | 2 |
| 18 | ZONE C1 | M | 4 | 4 | 2+ | 3 |
| 31 | ZONE D2 | M | 7 | 3 | 4+ | 4 |
| 32 | ZONE D2 | D | 10 | 6 | 5+ | 5 |
| 20 | ZONE D1 | M | 12 | 9 | 6+ | 6 |
| 30 | BLUE CREEK | D | 3 | 3 | 7 | 7 |
| 29 | BLUE CREEK | M | 20 | 5 | 8 | 8 |
| 27 | PGH | M | 13 | 7 | 10 | 9 |
| 28 | PGH | D | 21 | 10 | 9 | 10 |
| 14 | ZONE A | M | 8 | 8 | 11 | 11 |
| 25 | ALB | D | 6 | 6 | 12 | 12 |
| 16 | ZONE C1 | D | 14 | 10 | 13 | 13 |
| 22 | ZONE C2 | D | 19 | 12 | 14 | 14 |
| 12 | BAILEY | M | 16 | 12 | 15 | 15 |
| 15 | ZONE B | D | 18 | 13 | 16 | 16 |
| 17 | ZONE B | M | 5 | 13 | 18 | 17 |
| 19 | ZONE D1 | D | 15 | 13 | 17 | 18 |
| 26 | ALB | M | 11 | 11 | 20 | 19 |
| 11 | BAILEY | D | 22 | 13 | 19 | 20 |
| 13 | ZONE A | D | 17 | 14 | 21 | 21 |
| 21 | ZONE C2 | M | 9 | 15 | 22 | 22 |

The F Seam sample and the shales from Zone C and Zone D indicated the greatest overall self-heating. The coal samples also exhibited some self-heating. The Albright and Bailey refuse samples and the shales from Zones A and B exhibited the least self-heating.

The ability to self-heat is apparently not directly related to the amount of carbonaceous material in the sample, since two of the samples which exhibited the greatest degree of self-heating were shales with over 60 % ash. Similarly, heating value (BTU) is not a direct indicator of self-heating.

If self-heating of carbonaceous materials is related to heat generation and heat loss from a single process, it could be expected that increased temperature would simply increase the reaction rate of that process. Under these conditions, the material that was most prone to self-heat at 100° C would be the material most prone to self-heat at 150° C and at 200° C. The relative ranking of the tendency to self-heat should remain the same at all three levels. In this study, the relative rankings at the three levels were different for most of the samples. This may indicate that different reactions with different activation energies are involved in the self-heating process.

The samples were dried under nitrogen at 100° C to remove normal bed moisture, since initial trials indicated that no heating of the sample would occur until the bed moisture was removed. Then, if the

Table 4: COMPARISON OF MAXIMUM TEMPERATURE DIFFERENTIAL, DEG C, IN MOIST AND DRY AIR

| SAMPLE | TOTAL RANK D | TOTAL RANK M | M-D LEVEL 1 | M-D LEVEL 2 | M-D LEVEL 3+ | M-D TOTAL |
|---|---|---|---|---|---|---|
| ALB | 5 | 10 | -1[1] | -5 | -26 | -32 |
| BAILEY | 10 | 8 | 3 | 1 | 6 | 10 |
| F SEAM | 1 | 1 | -1 | -14 | -15 | -30 |
| BLUE CREEK | 3 | 5 | -4 | -10 | -11 | -25 |
| PGH | 4 | 6 | 2 | 3 | 0 | 5 |
| ZONE A | 11 | 7 | 2 | 9 | 38 | 49 |
| ZONE B | 8 | 9 | 2 | 0 | -2 | 0 |
| ZONE C1 | 6 | 2 | 1 | 12 | 81 | 94 |
| ZONE C2 | 7 | 11 | 2 | -6 | -22 | -26 |
| ZONE D1 | 9 | 4 | 1 | 5 | 74 | 80 |
| ZONE D2 | 2 | 3 | 1 | 11 | 6 | 18 |

[1]Negative values indicate higher temperature differentials in dry air.

Table 5: OXYGEN CONSUMPTION, Pct

| RUN # | SAMPLE | DRY/MOIST | $-O_2$[1] LEVEL 1 | $-O_2$ LEVEL 2 | $-O_2$ LEVEL 3 | $-O_2$ t = 0 |
|---|---|---|---|---|---|---|
| 25 | ALB | D | 3.94 | 8.62 | 13.78 | 12.76 |
| 26 | ALB | M | 2.76 | 4.21 | 11.80 | 13.46 |
| 11 | BAILEY | D | 1.21 | 2.21 | 5.38 | 4.70 |
| 12 | BAILEY | M | 1.72 | 2.75 | 11.34 | 1.44 |
| 23 | F SEAM | D | 1.09 | 5.00 | 8.61 | 18.46 |
| 24 | F SEAM | M | 2.30 | 9.09 | 9.96 | 19.50 |
| 30 | BLUE CREEK | D | 1.53 | 4.12 | 11.70 | 3.50 |
| 29 | BLUE CREEK | M | 1.93 | 2.64 | 11.41 | 3.36 |
| 28 | PGH | D | 2.94 | 3.10 | 14.75 | 2.30 |
| 27 | PGH | M | 2.19 | 7.50 | 18.61 | 6.31 |
| 13 | ZONE A | D | 4.60 | 4.53 | 6.56 | 1.80 |
| 14 | ZONE A | M | 4.09 | 4.73 | 12.80 | 3.36 |
| 15 | ZONE B | D | 2.13 | 1.82 | 7.70 | 1.80 |
| 17 | ZONE B | M | 1.49 | 1.78 | 8.16 | 0.62 |
| 16 | ZONE C1 | D | 0.94 | 3.50 | 12.03 | 0.88 |
| 18 | ZONE C1 | M | 1.35 | 4.71 | 13.50 | 5.04 |
| 22 | ZONE C2 | D | 0.24 | 1.48 | 12.09 | 12.83 |
| 21 | ZONE C2 | M | 1.49 | 2.39 | 4.98 | 4.37 |
| 19 | ZONE D1 | D | 0.66 | 1.35 | 3.48 | 2.14 |
| 20 | ZONE D1 | M | 1.68 | 3.50 | 14.50 | 1.70 |
| 32 | ZONE D2 | D | 1.40 | 6.74 | 12.49 | 7.67 |
| 31 | ZONE D2 | M | 0.87 | 3.25 | 13.85 | 5.34 |

[1]Difference between normal air and measured concentration in furnace.

exothermic adsorption of water is a mechanism for initiating self-heating, it would be expected that the maximum temperature differential, particularly at level 1, would be greater in moist air than in dry air. In this study, the effect of moisture at level 1 was minimal. It was more pronounced at levels 2 and 3, but could not be correlated to self-heating behavior (Table 4).

When the total rank for all dry samples is compared, the F seam sample and the Zone D2 shale are the most reactive under both conditions. The Bailey refuse and the Zone B shale had approximately the same reactivity under both conditions. For all other samples the relative reactivity was substantially different under moist and dry conditions.

If self-heating is directly related to the oxidation of the carbonaceous material, the amount of oxygen consumed and the amount of $CO_2$ produced should be proportional to the heat gain. Oxygen consumption (Table 5) does increase with increased temperature for each sample. However, it is not proportional to the temperature increase, and the presence of moisture in the air stream has no consistent effect on the change in the concentration of oxygen.

Oxygen consumption increased at sample temperatures equal to or greater than 100 deg C (Fig. 2), indicating an increase in oxidation rate at these temperatures. For approximately half of the samples, oxygen was apparently adsorbed on the sample when it was initially placed in the furnace(t = 0 - Table 5, Fig. 3). However, if physical adsorption is the only process involved, the consumption of oxygen should decrease at higher temperatures, unless the removal of some carbon moieties is producing more sites for oxygen adsorption.

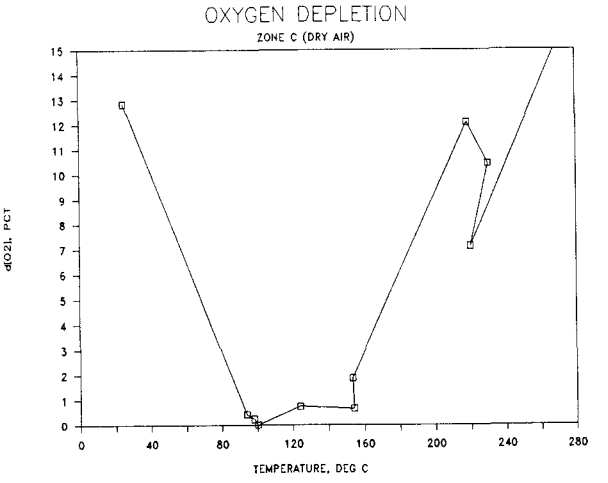



Figure 3. Oxygen Adsorbtion and Consumption versus Temperature

The production of $CO_2$ follows the same pattern as the consumption of oxygen (Table 6, Figs. 4, 5). If the initial adsorption of oxygen is low, the initial production of $CO_2$ is also low; both concentrations increase with temperature. If more oxygen is consumed initially, more $CO_2$ is produced initially. However, it should be noted that the amount of $CO_2$ produced is not equivalent to the amount of $O_2$ consumed. At temperatures above 100° C, the concentration of CO increases as the $CO_2$ concentration increases. At temperatures of 100° C or less, CO is apparently not a reaction product (Table 7).

Table 6: $CO_2$ PRODUCTION, Volume Pct

| RUN # | SAMPLE | DRY/MOIST | $CO_2$ LEVEL 1 | $CO_2$ LEVEL 2 | $CO_2$ LEVEL 3 |
|---|---|---|---|---|---|
| 25 | ALB | D | 0.04 | 0.44 | 1.77 |
| 26 | ALB | M | 0.06 | 0.16 | 1.42 |
| 11 | BAILEY | D | 0.09 | 0.16 | 0.90 |
| 12 | BAILEY | M | 0.03 | 0.19 | 2.22 |
| 23 | F SEAM | D | 0.33 | 1.95 | 2.77 |
| 24 | F SEAM | M | 0.28 | 1.38 | 1.81 |
| 30 | BLUE CREEK | D | 0.04 | 0.35 | 4.13 |
| 29 | BLUE CREEK | M | 0.03 | 0.26 | 2.47 |
| 28 | PGH | D | 0.49 | 2.26 | 1.44 |
| 27 | PGH | M | 0.08 | 0.35 | 4.34 |
| 13 | ZONE A | D | 0.10 | 0.07 | 0.81 |
| 14 | ZONE A | M | 0.06 | 0.48 | 2.25 |
| 15 | ZONE B | D | 0.02 | 0.11 | 1.11 |
| 17 | ZONE B | M | 0.04 | 0.19 | 1.31 |
| 16 | ZONE C1 | D | 0.02 | 0.26 | 1.96 |
| 18 | ZONE C1 | M | 0.05 | 0.36 | 3.00 |
| 22 | ZONE C2 | D | 0.02 | 0.18 | 1.94 |
| 21 | ZONE C2 | M | 0.02 | 0.06 | 1.26 |
| 19 | ZONE D1 | D | 0.02 | 0.13 | 0.72 |
| 20 | ZONE D1 | M | 0.04 | 0.25 | 2.24 |
| 32 | ZONE D2 | D | 0.04 | 0.72 | 2.19 |
| 31 | ZONE D2 | M | 0.03 | 0.30 | 2.50 |

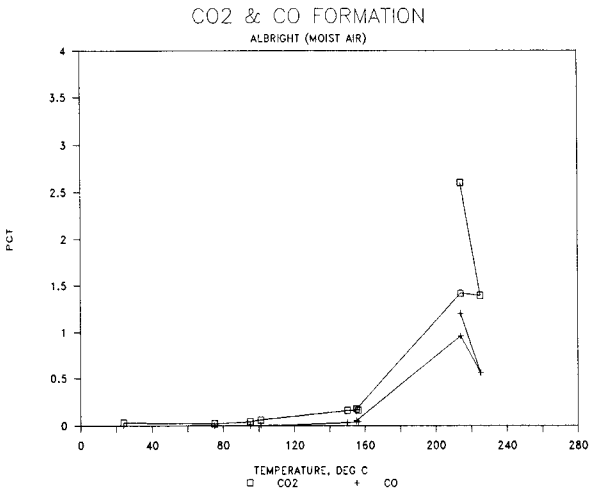



Figure 4. $CO_2$ and CO Generation, Initial Low Oxygen Adsorbtion

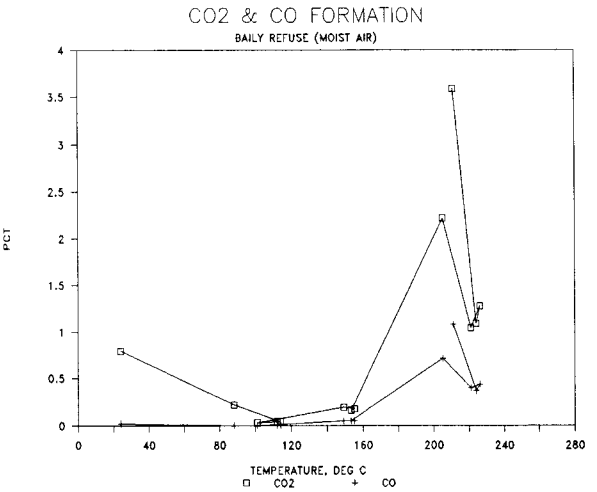



Figure 5. $CO_2$ and CO Generation, Initial High Oxygen Adsorbtion

## Summary

The results of this study of the self-heating characteristics of coal and other carbonaceous materials produced a large volume of data which has not yet been sufficiently analyzed to develop direct predictions of self-heating behavior. This study has, however, demonstrated that measured experimental parameters can be related to potential self-heating. It has also shown that high ash samples, such as coal wastes and carbonaceous shales are capable of generating sufficient heat to initiate fires in abandoned mines and waste banks.

The results have shown that self-heating is not a simple process, and it is apparently more complex than differences in heat generation and heat loss rates. In any sample, more than one heat generating reaction may occur, with one reaction providing the activation energy for subsequent reactions.

It should be noted that the system used did not allow bacterial activity and did not directly measure the possible effect of the oxidation of pyrite. Both of these factors may be significant in the generation of heat under normal conditions.

Table 7: CO CONCENTRATION, Volume Pct

| RUN # | SAMPLE | DRY/MOIST | CO LEVEL 1 | CO LEVEL 2 | CO LEVEL 3 |
|---|---|---|---|---|---|
| 25 | ALB | D | 0.00 | 0.05 | 0.76 |
| 26 | ALB | M | 0.00 | 0.04 | 0.96 |
| 11 | BAILEY | D | 0.00 | 0.06 | 0.28 |
| 12 | BAILEY | M | 0.00 | 0.05 | 0.40 |
| 23 | F SEAM | D | 0.00 | NA[1] | 0.89 |
| 24 | F SEAM | M | 0.01 | 0.51 | 0.45 |
| 30 | BLUE CREEK | D | 0.00 | 0.19 | 1.04 |
| 29 | BLUE CREEK | M | 0.00 | 0.00 | 0.94 |
| 28 | PGH | D | 0.00 | 0.04 | 0.59 |
| 27 | PGH | M | 0.00 | 0.06 | 1.86 |
| 13 | ZONE A | D | 0.00 | 0.00 | 0.21 |
| 14 | ZONE A | M | 0.00 | 0.05 | 1.46 |
| 15 | ZONE B | D | 0.00 | 0.01 | 0.42 |
| 17 | ZONE B | M | 0.00 | 0.00 | 0.41 |
| 16 | ZONE C1 | D | 0.00 | 0.03 | 1.00 |
| 18 | ZONE C1 | M | 0.00 | 0.09 | 1.95 |
| 22 | ZONE C2 | D | 0.00 | 0.04 | 0.87 |
| 21 | ZONE C2 | M | 0.00 | 0.00 | 0.14 |
| 19 | ZONE D1 | D | 0.00 | 0.00 | 0.26 |
| 20 | ZONE D1 | M | 0.00 | 0.04 | 0.94 |
| 32 | ZONE D2 | D | 0.00 | 0.23 | 0.95 |
| 31 | ZONE D2 | M | 0.00 | 0.08 | 1.00 |

[1]NA: Not Available

## Literature Cited


American Society for Testing and Materials. 1983. Standard Classification of Coals by Rank, D388-82 in 1983 Annual Book of ASTM Standards: Petroleum Products, Lubricants, and Fossil Fuels.

Chaiken, R.F. 1977. Heat Balance in In Situ Combustion. BuMines RI 8221, 11 p.

Chaiken, R.F., R.J. Brennan, B.S. Heisey, A.G. Kim, W.T. Malenka, and J.T. Schimmel. 1983. Problems in the Control of Anthracite Mine Fires: A Case Study of the Centralia Mine Fire (August 1980). BuMines RI 8799, 93 p.

Dalverny, L.E. and R.F. Chaiken. 1990. Mine Fire Diagnostics and Implementation of WIFE at Renton, PA. BuMines RI, in press.

Jones, J. R. and B. Cameron. 1988. Modern Coastal Back-barrier Environment: Analog for Coal Basin or Carbonaceous Black Shale? Geol. 16: 345-348.

Kanury, A. Murty. 1975. Introduction to Combustion Phenomena. Gordon and Breach Science Publishers, New York. 411 p.

Kim, A.G. 1977. Laboratory Studies on Spontaneous Heating of Coal: A Summary of Information on the Literature. BuMines IC 8756, 13 p.

Schopf, J.M. 1966. Definitions of Peat and Coal and of Graphite that Terminates the Coal Series(Graphocite). J. of Geol. 74(5): 584-592.

Smith, A.F. and C.P. Lazarra. 1987. Spontaneous Combustion Studies of U.S. Coals. BuMines RI 9079, 28 p.

# Revegetation - AML

This page intentionally left blank.

# TREE SPECIES COMPOSITION, CANOPY COVERAGE, AND IMPORTANCE ON SEVERAL AML SITES IN NORTHERN WEST VIRGINIA[1]

C. D. Johnson and J. G. Skousen[2]

Abstract. Woody plant cover and composition were evaluated on seven Abandoned Mine Land (AML) sites in northern West Virginia during the summer of 1989. Five sites had been mined for the Freeport coal seam and two sites from the Pittsburgh coal seam. The thickness and pH of minesoil A horizons on each site were measured in the field and related to tree composition and importance. A total of 24 tree species occurred across our seven AML sites. All sites, except for one site, included the following tree species: big-toothed aspen (Populus grandidentata Michx.), black birch (Betula lenta L.), black locust (Robinia psuedoacacia L.), Hercules' club (Aralia spinosa L.), pin cherry (Prunus pensylvanica L. f.), red maple (Acer rubrum L.), and sourwood (Oxydendrum arboreum (L.) DC.). Canopy coverage and community structure increased as surface minesoil pH decreased. There were indications that both surface minesoil pH and thickness influenced a few tree species' importance and canopy coverage across our sites.

Additional key words: revegetation, minesoil acidity.

## Introduction

Approximately 34,000 hectares of mined land in West Virginia have been designated as Abandoned Mine Land (AML) (SCS 1979). An AML site is an area of past mining disturbance which was not adequately reclaimed prior to the passage of the Surface Mining Control and Reclamation Act (SMCRA) on August 3, 1977; and where no company, individual, or agency has any reclamation responsibility under state or federal laws.

Natural processes have operated and continue to allow natural reclamation to amend in some degree these inadequately reclaimed sites. Although considerable research has been conducted toward revegetation of coal mine lands (Bradshaw and Chadwick 1980, Chadwick and Goodman 1973, Powell 1988), less research has been done on natural revegetation.

Succession has been studied on many AML sites by assuming that a series of increasingly older AML sites from the same area with similar geology infers a temporal sequence of a single site (Drury and Nisbet 1973, Kimmerer 1984, Skousen et al. 1988). Generally, such sites are first dominated by pioneer plants that are tolerant to extremes in temperature, moisture, and light; and later inhabited by plant species that establish and/or become conspicuous after some amelioration of the extreme condition. Coal mine sites (Bramble and Ashly 1955, Hall 1957) were colonized by

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]C. D. Johnson is graduate research assistant, and J. G. Skousen is Assistant Professor in the Division of Plant and Soil Sciences, West Virginia University, Morgantown, WV 26506-6108. This project is supported by federal funds from the U.S. Bureau of Mines and the National Mine Land Reclamation Center as Cooperative Agreement CO388026, Project #WV08.

those local species which have the capacity to disperse seeds over long distances. Site age or time since abandonment has been shown to affect community composition on disturbed soils in England (Hall 1957, Roberts et al. 1981) and minesoils in Illinois (Lindsay and Nawrot 1981).

Soil acidity was found to suppress the rate of revegetation on Missouri minesoils (Game et al. 1982). On Ohio minesoils (Bell and Ungar 1981), unvegetated areas surrounded vegetation patches of approximately the same low pH, low nutrient levels, and toxic aluminum concentrations. However, the unvegetated areas were unable to support seed emergence due to low soil moisture.

Concurrent with plant community development and succession is the development of a soil profile (Drury and Nisbet 1973). During minesoil development, chemical and physical weathering processes occur which eventually improve physical and chemical properties for enhanced plant establishment and growth.

Depth of the A horizon increased with age in minesoils in Virginia (Daniels and Amos 1981). Minesoil pH of acid sites in Kentucky increased with time while pH of near neutral minesites decreased slightly (Davidson et al. 1988). This indicated that the minesoils were developing toward a soil pH range of 4.0 to 4.9, which is the general pH range of forested areas in Kentucky. On minesoils in Pennsylvania (Pedersen et al. 1978), the pH values were similar to native soils.

The purpose of this paper is to evaluate the thickness and pH of minesoil A horizons on seven AML sites in northern West Virginia and relate these soil properties to tree composition, canopy cover, and importance value. This study is part of a larger project which is attempting to identify and describe minesoil genesis in relationship to plant community development on abandoned mine sites.

## Materials and Methods

A total of seven AML sites in West Virginia were sampled during the summer of 1989. The AML inventory list developed by the West Virginia Department of Energy was used as the pool of available sites for selection. This pool was then reduced to sites with south- to west-facing highwall aspects to control micro-climatic variation (Hicks et al. 1982, Hicks and Frank 1984). These sites were then examined to identify the coal seam that had been mined by using the West Virginia Geological and Economic Survery's General and Economic Geology maps (Hennen and Reger 1912, 1913, 1914). After identifying which coal seam was mined on each site, the Freeport and Pittsburgh sites were each assigned a number. A computer random number generator was then utilized to select five sites which had been mined for the Freeport coal seam and two sites from the Pittsburgh coal seam. Blocking by coal seam was used to reduce the variation between parent materials from the different coal seams. Time since abandonment was not selected but through random sampling the selected sites constituted sites of different age.

### Vegetation Sampling

Vegetation was sampled at each AML site at five stations. Each station measured 10 m x 10 m. The stations were sited by placing a grid over a map of each site and randomly selecting the five stations with a computer coordinate generator. All tree species in each station were identified and each species' canopy cover was measured by the line intercept method on a 10 m transect. The importance of each tree species was assessed by the Importance Value (IV) (Curtis and McIntosh 1951) by the following formula:

$$IV = \text{Relative density} + \text{Relative frequency} + \text{Relative dominance} \quad (1)$$

$$RDen = \frac{\text{number of individuals of Species} \times 100\%}{\text{total number of individuals}} \quad (2)$$

$$RFre = \frac{\text{frequency of a species} \times 100\%}{\text{(sum frequency of all species)}} \quad (3)$$

$$RDom = \frac{\text{dominance of a species} \times 100\%}{\text{dominance of all species}} \quad (4)$$

### Soil Sampling

Three soil pits were dug to a depth of 1 m next to three of the stations (randomly selected). The pH and thickness of the minesoil A horizons were measured in the field.

While other vegetation measurements were taken, this paper reports only on tree species: composition, importance value, and cover. Additional minesoil analyses of chemical and physical properties are ongoing.

## Results and Discussion

Table 1 depicts the presence or absence of 24 tree species across our seven AML sites. Most sites included the following species: big-toothed aspen (Populus grandidentata Michx.), black birch (Betula lenta L.), black locust (Robinia psuedoacacia L.), Hercules' club (Aralia spinosa L.), pin cherry (Prunus pensylvanica L.), red maple (Acer rubrum L.), and sourwood

Table 1. Presence (P) or Absence (-) of 24 tree species found on Freeport (F) and Pittsburgh (P) AML sites in West virginia.

| Species | AML sites | | | | | | |
|---|---|---|---|---|---|---|---|
| | BR-F | JS-F | JP-F | LG-F | VP-F | HU-P | FR-P |
| Bigtooth Aspen *Populus grandidentata* Michx. | P | - | P | P | P | P | - |
| Black Birch *Betula lenta* L. | P | P | P | P | P | P | - |
| Black Locust *Robinia psuedoacacia* L. | P | P | P | P | P | - | - |
| Hercules' Club *Aralia spinosa* L. | - | - | - | P | P | P | - |
| Pin Cherry *Prunus pensylvanica* L. f. | P | P | P | P | P | - | P |
| Red Maple *Acer rubrum* L. | P | P | P | P | P | P | - |
| Red Oak *Quercus rubra* L. | P | - | P | P | P | P | - |
| Sassafras *Sassafras albidum* (Nutt.) | - | - | P | P | P | P | - |
| Sourwood *Oxydendrum arboreum* (L.) DC. | - | P | P | - | - | P | - |
| Tuliptree *Liriodendron tulipifera* L. | P | - | P | - | P | P | - |
| Other Species | | | | | | | |
| Autumn Olive *Elaeagnus umbellata* Thunb. | - | - | - | P | P | - | - |
| Beech *Fagus grandifolia* Ehrh. | - | - | - | - | - | P | - |
| Black Cherry *Prunus serotina* Ehrh. | - | - | - | - | - | P | - |
| Chestnut *Castanea dentata* (Marsh.) Borkh. | - | - | - | P | - | - | - |
| Cucumber Tree *Magnolia acuminata* L. | - | - | - | P | P | - | - |
| Dogwood *Cornus florida* L. | P | - | - | - | - | P | - |
| Mountain Laurel *Kalmia latifolia* L. | - | - | P | - | - | - | - |
| Pignut Hickory *Carya glabra* (Mill.) | - | - | - | - | - | P | P |
| Red Elm *Ulmus rubra* Muhl. | P | - | - | - | - | - | P |
| Smooth Alder *Alnus serrulata* (Ait.) Willd. | - | - | P | - | P | - | - |
| Smooth Arrowwood *Viburnum recognitum* Fernald | - | - | - | P | P | - | - |
| Sycamore *Platanus occidentalis* L. | - | - | - | - | - | P | P |
| White Oak *Quercus alba* L. | - | - | - | P | P | - | - |
| Yellow Birch *Betula alleghaniensis* Britt. | - | - | - | - | P | - | - |

Table 2. Canopy coverage (%) of major tree species on seven AML study sites in northern West Virginia.

| | AML SITES | | | | | | |
|---|---|---|---|---|---|---|---|
| Species | BR | JS | JP | LG | VP | HU | FR |
| Aspen | 51.2 | 0.0 | 3.8 | 30.2 | 15.8 | 20.0 | 0.0 |
| Birch | 0.0 | 15.0 | 71.8 | 0.4 | 98.6 | 25.5 | 0.0 |
| Black locust | 39.2 | 52.6 | 21.4 | 6.6 | 40.0 | 0.0 | 0.0 |
| Cherry | 0.0 | 39.0 | 1.6 | 89.2 | 3.8 | 0.0 | 0.0 |
| Hercules' club | 0.0 | 0.0 | 0.0 | 40.7 | 0.5 | 0.0 | 0.0 |
| Red maple | 0.0 | 7.2 | 32.3 | 19.8 | 49.4 | 26.8 | 0.0 |
| Sourwood | 0.0 | 0.0 | 10.7 | 0.0 | 0.0 | 0.5 | 0.0 |
| Other species | 9.8 | 0.0 | 4.8 | 2.2 | 3.8 | 11.5 | 0.0 |
| Total species | 100.2 | 113.8 | 135.7 | 189.1 | 211.9 | 84.3 | 0.0 |

(Oxydendrum arboreum (L.) DC.). The AML sites were also vegetated by several other tree species including red oak (Quercus rubra L.), sassafras (Sassafras albidum (Nutt.) Nees.), and tuliptree (Liriodendron tulipifera L.). These three species along with several others were combined into one group and will hereafter be termed "other species". Red oak, sassafras, and tuliptree, while occurring on 4 or 5 sites, were not prominent species on our sites.

Table 2 depicts the percent canopy coverage of several tree species on our AML sites. The FR site had no canopy coverage while the BR site had a low level of canopy coverage. This varying amount of canopy coverage was due to the respective community types across these AML sites. The FR site was predominately a meadow with a few scattered small trees, BR site was a relatively open forest dominated by small trees, and the other AML sites we studied were forests. Percent canopy coverage was negatively correlated with minesoil pH in the surface horizon (r = -.76) (Tables 2 and 3). The two high pH sites had 0 and 100% canopy coverage (FR and BR sites, respectively), while the low pH sites had from 85 to 200% canopy coverage. Canopy coverage was also negatively correlated with thickness of the A horizon (r = -.81) (Figure 1). This indicates that A horizon development was greater under herbaceous cover than woody cover likely due to differences in biomass accumulation and incorporation into the soil (Leisman 1957).

There was a negative correlation between number of tree species and surface minesoil pH (r = -.61) (Figure 2) and also with the thickness of the A horizon (r = -.74) (Figure 3).

Linear correlations indicated marginal relationships between either thickness and pH of the surface minesoil horizon with individual tree species importance values (Table 4) across our AML sites (Table 5). Only black birch, Hercules' club) and red maple had absolute r values greater than .50. However, several relationships could be drawn between minesoil surface properties and IV for different tree species. For example, red maple had moderate IVs on the low pH sites followed by its highest IV on the moderate pH site (BR), but had no IV on the highest pH site (FR). Red maple also steadily increased in IV with increased thickness of the A horizon, but had a sharp decrease in IV on sites with A horizons greater than 7 cm (JS and FR). Black birch and sourwood only had large IVs on the sites with low pH

Table 3. Thickness (cm) and pH of minesoil A horizons on AML sites in northern West Virginia.

| | AML SITES | | | | | | |
|---|---|---|---|---|---|---|---|
| Parameter | BR | JS | JP | LG | VP | HU | FR |
| Thickness (cm) | 6.7 | 8.0 | 4.2 | 0.0 | 1.5 | 6.8 | 7.3 |
| pH | 5.9 | 4.3 | 4.2 | 4.3 | 4.6 | 4.5 | 7.3 |

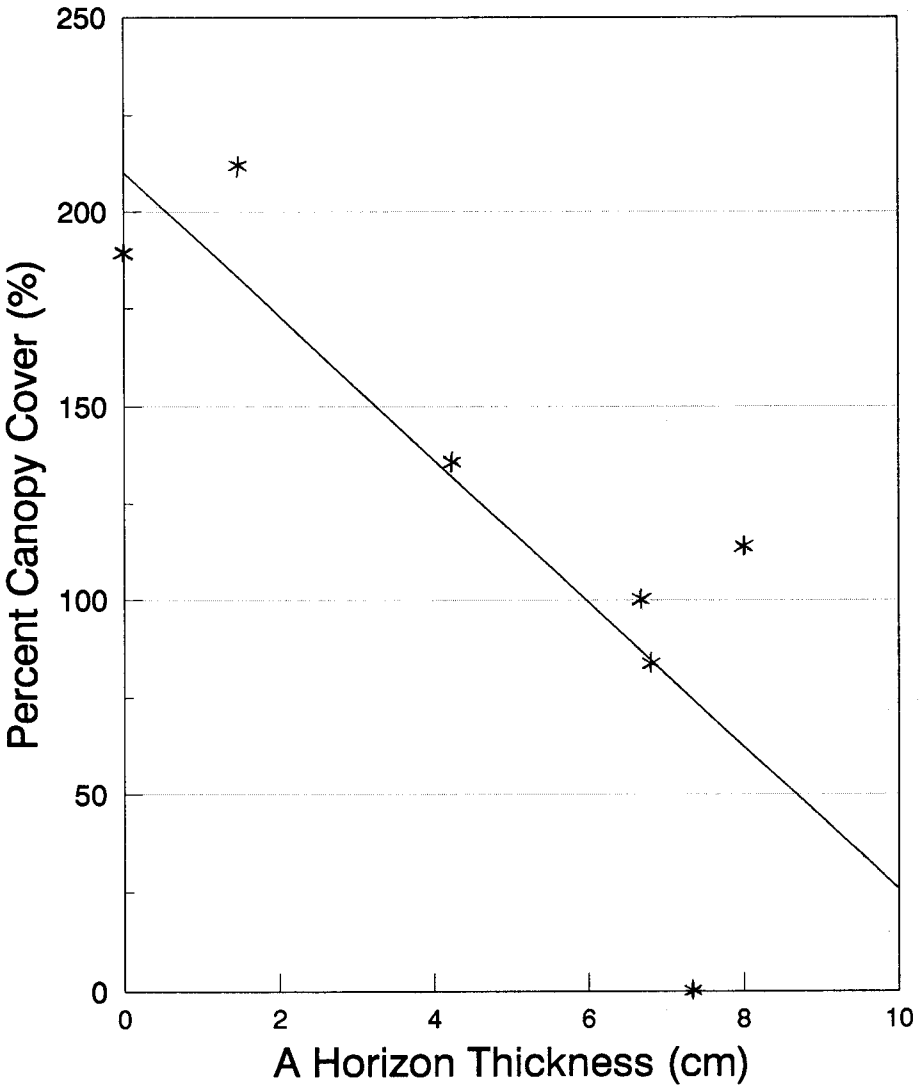



Figure 1. Percent canopy cover of tree species vs. thickness (cm) of minesoil A horizons on AML sites in northern West Virginia.

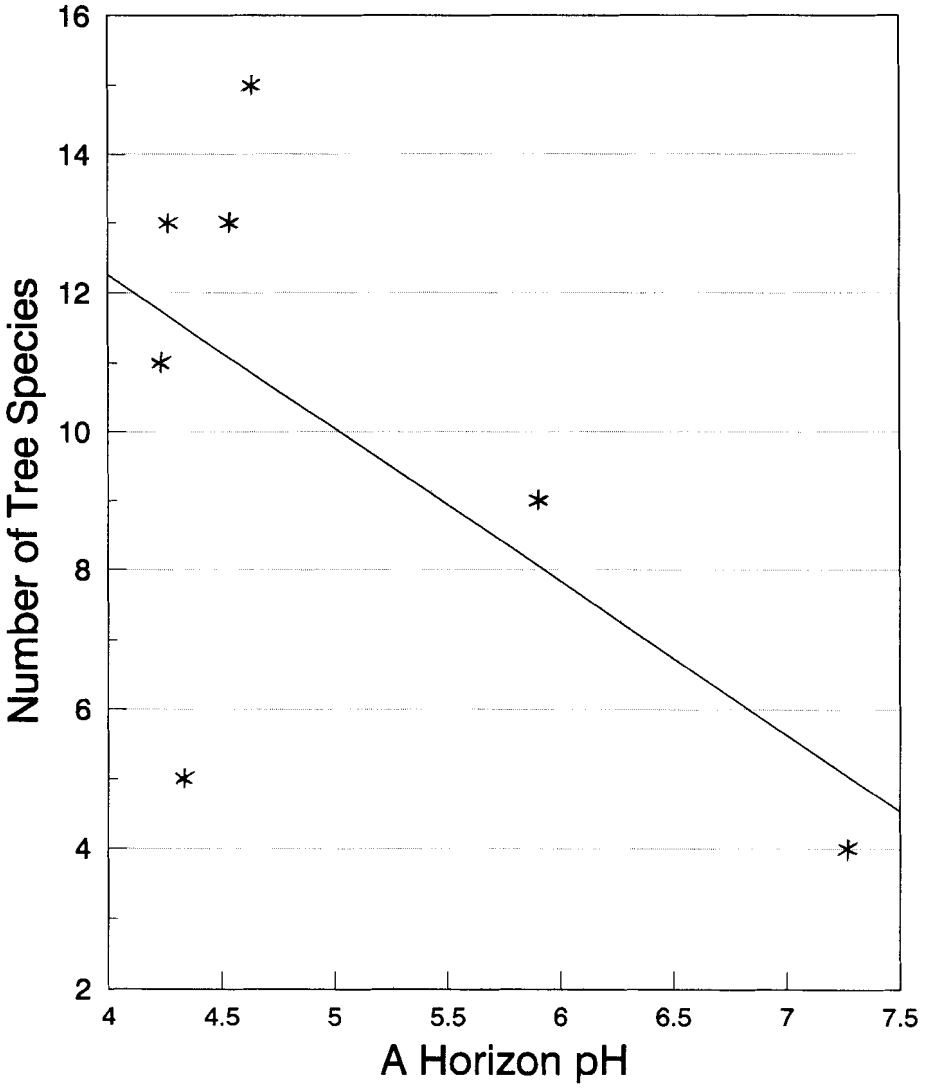



Figure 2. Number of tree species vs. pH of minesoil A horizons on AML sites in northern West Virginia.

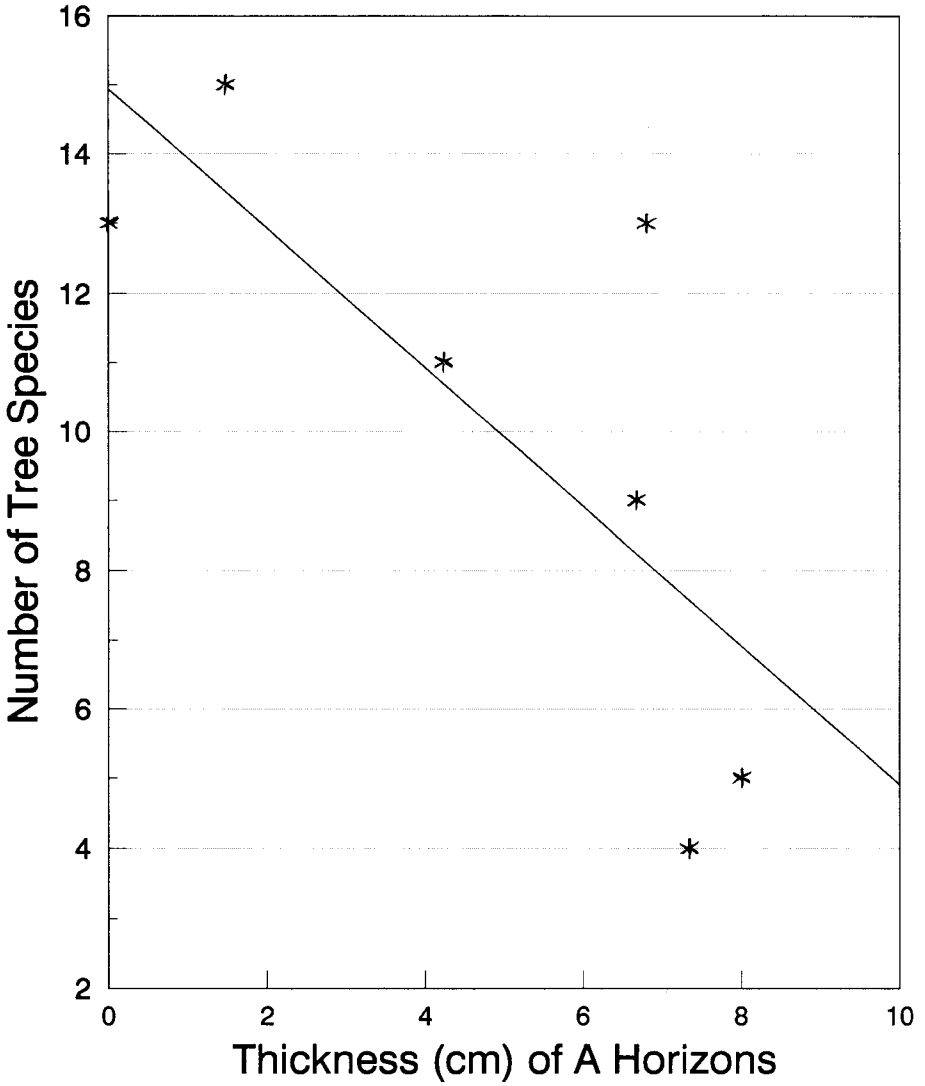



Figure 3. Number of tree species vs. thickness (cm) of minesoil A horizons on AML sites in northern West Virginia.

Table 4. Importance value (IV) of major tree species on seven AML study sites in northern West Virginia.

| | AML SITES | | | | | | |
|---|---|---|---|---|---|---|---|
| Species | BR | JS | JP | LG | VP | HU | FR |
| Aspen | 100.8 | 0.0 | 8.3 | 31.2 | 11.5 | 19.3 | 0.0 |
| Birch | 19.8 | 20.9 | 115.9 | 9.6 | 122.5 | 76.0 | 0.0 |
| Black locust | 20.6 | 148.5 | 18.0 | 28.5 | 28.2 | 0.0 | 0.0 |
| Cherry | 7.6 | 78.0 | 9.9 | 57.0 | 13.9 | 0.0 | 96.7 |
| Hercules' club | 0.0 | 0.0 | 0.0 | 68.0 | 7.2 | 16.5 | 0.0 |
| Red maple | 78.5 | 39.6 | 64.9 | 46.8 | 59.3 | 71.4 | 0.0 |
| Sourwood | 0.0 | 13.0 | 49.8 | 0.0 | 0.0 | 45.4 | 0.0 |
| Other species | 32.7 | 0.0 | 33.3 | 58.9 | 57.3 | 71.5 | 123.3 |
| Total species | 260.0 | 300.0 | 300.0 | 300.0 | 300.0 | 300.0 | 220.0 |

and moderately thick A horizons. Black locust dominated the canopy cover on two moderately deep A horizon sites (BR and JS) but only had a high IV on the low pH JS site. Whereas, big-toothed aspen dominated the canopy and had a high IV on the BR site. Trends can also be seen for Hercules' club and pin cherry.

Analysis of other minesoil properties (e.g. particle size distribution, surface bulk density, water-holding capacity, electrical conductivity, potassium, calcium, magnesium, sulfur, aluminum, phosphorous, and copper, iron, manganese, and zinc may provide more insight into the reasons for tree establishment and dominance on particular AML sites in this study. From this work we can better understand the relationships between minesoil properties and species adaptations to specific conditions.

## Literature Cited


Bell, T. J., and I. A. Ungar. 1981. Factors affecting the establishment of natural vegetation on a coal strip mine spoil bank in southeastern Ohio. Am. Midl. Nat. 105:19-31.

Bradshaw, A. D. and Chadwick, M. J. 1980. The Restoration of Land. Blackwell Scientific, Oxford.

Bramble, W. C. and R. A. Ashly. 1955. Natural revegetation of spoil banks in central Pennsylvania. Ecology 36:417-423.

Chadwick, M. J. and G. T. Goodman. 1973. The Ecology of Resource Degradation and Renewal. Blackwell Scientific, Oxford.

Curtis, J. T. and R. P. McIntosh. 1951. An upland forest continuum in the prairie-forest border region of Wisconsin. Ecology 32:476-498.

Daniels, W. L. and D. F. Amos. 1981. Mapping, characterization and genesis of mine soil on a reclamation research area in Wise County, Virginia. Symp. on Surface mIning, Hydrology, Sedimentology and Reclamation, Lexington, Kentucky. 7-11 December, 1981.

Davidson, W. H., W. C. Ashby, and W. G. Vogel. 1988. Progressive changes


Table 5. Correlation coefficient (r) between each major tree species' importance value (IV) and thickness (cm) and pH of the surface minesoil horizon across seven AML sites in northern West Virginia.

| Species | Thickness (cm) | pH |
|---|---|---|
| Aspen | 0.018 | 0.18 |
| Birch | -0.32 | -0.51 |
| Black locust | 0.25 | -0.37 |
| Cherry | 0.20 | 0.42 |
| Hercules' club | -0.70 | -0.36 |
| Red maple | -0.18 | -0.57 |
| Sourwood | 0.21 | -0.46 |
| Other sp. | -0.034 | 0.67 |

in minesoil pH over three decades. pp. 89-92. In: Vol. II: Mine Reclamation, Abandoned Mine Lands and Policy Issues.

Drury, W. H. and I. C. T. Nisbet. 1973. Succession. Journal of the Arnold Arboretum 54:331-368.

Game, M., J. E. Carrel, and T. Hotrabhavandra. 1982. Patch dynamics of plant succession on abandoned surface coal mines: a case history approach. Journal of Ecology 70:707-720.

Hall, I. G. 1957. The ecology of disused pit heaps in England. Journal of Ecology 45:691-720.

Hennen, R. V. and D. B. Reger. 1912. Map II of Doddridge and Harrison counties showing general and economic geology. West Virginia Geological and Economic Survey. A. Hoen and Co., Baltimore, Md.

Hennen, R. V. and D. B. Reger. 1913. Map II of Monongalia, Marion, and Taylor counties showing general and economic geology. West Virginia Geological and Economic Survey. A. Hoen and Co., Baltimore, MD.

Hennen, R. V. and D. B. Reger. 1914. Map II of Preston county showing general and economic geology. West Virginia Geological and Economic Survey. A. Hoen and Co., Baltimore, MD.

Hicks, R. R., Jr. and P. S. Frank, Jr. 1984. Relationship of aspect to soil nutrients, species importance and biomass in a forested watershed in West Virginia. Forest Ecology and Management 8:281-291.

Hicks, R. R., Jr., P. S. Frank, Jr., H. V. Wiant, Jr., and K. L. Carvell. 1982. Biomass productivity related to soil-site factors on a snall watershed. p. 9-12. W.V. For. Notes, W.V. Agric. Exp. Stn. Circ., 121.

Kimmerer, R. W. 1984. Vegetation development on a dated series of abandoned lead and zinc mines in southwestern Wisconsin. Am. Midland Naturalist 111:332-341.

Leisman, G. A. 1957. A vegetation and soil chronosequence on the Mesabi iron range spoil banks, Minnesota. Ecol. AMonographs 27:221-245.

Lindsay, R. E. and J. R. Nawrot. 1981. Evaluation of natural revegetation of problem spoil banks. p. 367-375. Symp. of Surface Mining Hydrology, Sedimentology and Reclamation, Lexington, Kentucky. 7-11 December, 1981.

Pedersen, T. A., A. S. Rogowski, and R. Pennock, Jr. 1978. Comparison of morphological and chemical characteristics of some soils and minesoils. Reclamation Review 1:143-156.

Powell, J. L. 1988. Revegetation options. p. 49-92. In L. R. Hossner (ed.) Reclamation of Surface-Mined Lands. Vol 2. CRC Press, Inc., Boca Raton, Florida.

Roberts, R. D., R. H. Marrs, R. A. Skeffington, and A. D. Bradshaw. 1981. Ecosystem development on naturally colonized china clay wastes. I. Vegetation changes and overall accumulation of organic matter and nutrients. Journal of Ecology 69:153-161.

Skousen, J. G., C. A. Call, and R. W. Knight. 1988a. A chronosequence of vegetation and minesoil development on a Texas lignite surface mine. p. 79-88. In: Mine drainage and surface mine reclamation Volume II: Mine reclamation, abandoned mine lands and policy issues. Proc. Conference ASSMR, BOM, and OSM, Pittsburgh, Pennsylvania. April 19-21, 1988.

Soil Conservation Service. 1979. The status of land disturbed by surface mining in the United States. SCS-TP-158. USDA, Soil Conservation Service, Washington, DC. 124 p.

This page intentionally left blank.

# REVEGETATION OF ABANDONED MINE LANDS IN PENNSYLVANIA WITH CONTAINERIZED SEEDLINGS AND SOIL AMENDMENTS[1]

by

Russell J. Hutnik and H. Glenn Hughes[2]

Abstract. In 1977, a study was initiated in central Pennsylvania on the feasibility of revegetating barren abandoned mine lands with containerized red pine (Pinus resinosa) seedlings in combination with various soil amendments and herbaceous seed mixes. The sites selected had low nutrient availability, high acidity (pH 3.0 to 3.3), and high exchangeable Al. Vegetation was lacking even though tree seedlings had been planted following completion of mining in 1968-69. Variables tested were location (3 sites), soil amendment (none, lime, lime + fertilizer, lime + fertilizer + mulch), seeding mix (none, deertongue grass + birdsfoot trefoil, KY-31 tall fescue + birdsfoot trefoil), and seedling type (2 + 0 bare-root seedlings, Japanese paperpot seedlings). There were 144 plots with 20 trees per plot. In 1989, 12 growing seasons following planting, over-all survival was 71.5%. The only significant differences in survival involved seedling type; bare-root seedlings had significantly higher survival than Japanese paperpot seedlings at one location (Renovo), but not at the other two. There were significant differences in tree height among amendments, with mean heights of 3.0, 3.7, 4.4, and 4.4 m for no amendment, lime, lime + fertilizer, and lime + fertilizer + mulch, respectively. No other variables and no interactions were significant with respect to tree height. Supplemental studies in subsequent years showed that: (1) other types of containers had survival and height growth rates similar to those of the bare-root and Japanese paperpots in the main study; and (2) survival was high after 10 years for all plantings done during three consecutive years.

Additional Key Words: Pinus resinosa Ait., reclamation, bare-root seedlings, herbaceous seed mixes, survival, tree height.

## Introduction

In 1977, the Surface Mining Control and Reclamation Act was passed to regulate surface mining and reclamation of mine lands in the U.S. Title IV of the act specified funds for the reclamation of abandoned mine land through a levy on each ton of coal mined. This provision was of special importance to Pennsylvania where an estimated 159,000 acres had been abandoned and were in need of some reclamation activity (Grim and Hill 1974). It was apparent the funds generated from the tax on coal produced would not be nearly enough to reclaim all of the abandoned mine land according to the standards in the federal law. Therefore, a priority system was established which provided for complete reclamation only on those sites posing a threat to public safety and welfare. Less emphasis and money were allocated for sites with environmental problems (e.g. those that were barren and/or producing acid mine drainage).

A vegetative cover would reduce environmental problems associated with these low-priority abandoned mine lands. Erosion would be reduced, aesthetics would be improved, food and habitat for wildlife would be created, and possibly wood products would eventually be produced. Therefore, inexpensive methods to produce or generate some vegetative cover needed to be developed. This was the overall objective of a series of studies begun in 1977 in the bituminous region of

[1] Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2] Russell J. Hutnik is Professor Emeritus of Forest Ecology, The Pennsylvania State University, University Park, PA 16802 and H. Glenn Hughes is Associate Professor of Forest Resources, The Pennsylvania State University, DuBois, PA 15801.

Pennsylvania. The underlying considerations of the studies and justification for them are:

1. Species. The choice was red pine (Pinus resinosa), since it performs well on a variety of mine sites in Pennsylvania. Including other species in the study would increase its complexity and cost and probably yield little information in addition to that obtained in past species trials on abandoned mine lands, such as that by Horn and Ward (1969).

2. Seedling type. Almost all of the red pine seedlings previously planted on mine lands had been two-year-old seedlings raised in conventional nursery beds and planted in a bare-root condition. However, in various parts of North America, use of containerized seedlings was growing in popularity. The qualities possessed by such seedlings (Tinus 1974) might enable them to survive and grow better than the bare-root stock when planted on adverse sites. Therefore several types of container-grown seedlings as well as bare-root nursery stock were considered for use in the study.

3. Soil amendment. Abandoned mine lands in Pennsylvania are characterized by high acidity, low available nutrients, and, in many cases, toxic levels of aluminum. They are low in soil-moisture capacity and are often subject to high surface temperatures. Therefore, various combinations of lime, fertilizer, and mulch were included in the study.

4. Herbaceous seed mixes. Planted tree seedlings provide negligible cover for several years following planting. In order to achieve a more immediate vegetative cover, it is necessary to apply an herbaceous seed mix, preferably containing at least one legume species. Cool-season grasses have usually been used, although there are indications that warm season grasses are less competitive with newly established tree seedlings and, once established, provide a longer-lasting cover. Therefore a mix containing a cool-season grass, KY-31 tall fescue (Festuca arundinacea), was compared to one containing a warm-season grass, deertongue grass (Panicum clandestinum). Each contained a legume, birdsfoot trefoil (Lotus corniculatus).

## Main Study

### Procedures

In 1976, three study sites were selected in various parts of the bituminous region of Pennsylvania (Figure 1). All three sites had previously been planted with tree species, but for one reason or another, survival was nil. The sites, named for the closest large town, were:

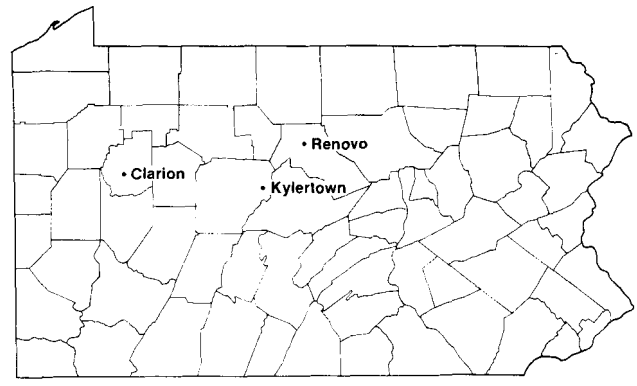



Fig. 1. Location of abandoned mine land study sites.

1. Kylertown. Approximately 10.5 km north-northwest of Kylertown; Lower Kittanning seam; mining completed in 1969; average pH 3.3.

2. Clarion. Approximately 14.5 km southwest of Clarion; Upper and Lower Kittanning seams; mining completed in 1968; averaged pH 3.2.

3. Renovo. Approximately 7.2 km northwest of Renovo; Lower Kittanning seam; mining completed in 1968; average pH 3.0.

A randomized complete block design with split plots provided a factorial experiment at each location. There were two replications, each containing four major plots, with six split plots in each major plot. The major plot factor was amendment (control, lime, lime + fertilizer, lime + fertilizer + mulch). The split plots contained all possible combinations of two factors, seeding mix (control, KY-31 tall fescue + birdsfoot trefoil, deertongue grass + birdsfoot trefoil) and seedling type (2 + 0 bare-root nursery stock, seedlings grown in Japanese paperpots (Hodemaker 1974), with individual containers of 3.8 cm in diameter by 15 cm in length, in the greenhouse for an extended growing season). Each split plot contained 20 trees planted on a 7-ft x 7-ft (2.1 m x 2.1 m) spacing.

The study was originally designed to include two additional types of containers, the Tinus Spencer-Lemaire book planter (Spencer 1974) and multi-seedling pots, a relatively new concept in container technology. The multi-seedling pots had a top diameter of 18.5 cm and a height of 19.0 cm. Each had a maximum of 12 seedlings per container. The seedlings are bare-rooted just before planting in the field; thus, they have similarities to both nursery-grown stock and conventional containerized seedlings. However, the Spencer-Lemaire and multi-seedling pot seedlings were of very poor quality at the time of planting. They were small, chlorotic, and low in vigor.

Within 8 weeks following planting, 71.5% of the Spencer-Lemaire seedlings and 99.4% of the multi-seedling pot seedlings were dead. Data from these types of containers have therefore been omitted from the statistical analyses.

Spoil samples were taken in late summer and early fall 1976 and analyzed for texture, pH, lime requirement, available P, exchangeable K, Mg, Ca, Al, and Mn, and specific conductivity.

Ground agricultural limestone was applied in accordance with the soil test recommendations in March 1977 at the following rates: Kylertown and Clarion, 4 t/acre (9.0 mt/ha); and Renovo, 5 t/acre (11.2 mt/ha). All plots, including controls, were scarified by disking or ripping following application of the lime.

Tree seedlings were planted during the period of April 1 through 3, 1987. Because of a heavy rain on April 2, the spoils were at or near field capacity during or soon after planting. Herbaceous seed and fertilizer were broadcast applied in mid-April. Each mix contained 10 lb/acre (11.2 kg/ha) of birdsfoot trefoil and 20 lb/acre (22.4 kg/ha) of a grass--either KY-31 tall fescue or "Tioga" deertongue grass. An 8-32-10 fertilizer, with 70% of the N in a slow-release form (sulfur-coated urea), was applied at a rate of 600 lb/acre (670 kg/ha). About one week later, hay mulch was applied by hand at a rate of 2 t/acre (4.5 mt/ha) and tacked down with an asphalt emulsion. Details on the study sites, soil analyses, and plot and treatment are available in Goodman (1978).

Data on tree seedling survival and growth have been collected at various time since planting. Our paper deals primarily with the most recent data collection, made in spring 1989, 12 years following plot establishment. These data were analyzed using conventional analysis of variance techniques for randomized blocks with split plots.

## Results and Discussion

Chemical properties of the minesoils before treatment are given in Table 1. The values did not vary greatly among locations and were similar to those reported by Beyer and Hutnik (1969) and Horn and Ward (1969) for the same seams, but in other locations in Pennsylvania. The spoils were coarse in texture, indicating possible problems with water stress, high surface temperatures, and low nutrient availability. The pH values were low and available Al was high, indicating a potential for toxicity.

In spite of the adverse conditions existing on the untreated spoil, differences in mean survival among amendments were not significant after the first growing season (Goodman 1978) nor after the twelfth. However, there was a difference in pattern of survival with time (Table 2). Mortality has occurred between every measurement date on the untreated control plots. In contrast, mortality on those receiving soil amendments leveled off after the first few growing seasons. Survival decreased by 12 percentage points between the first and twelfth seasons for the untreated control, but only 5, 4, and 1 percentage points for the lime, lime + fertilizer, and lime + fertilizer + mulch treatments, respectively. Evidently, some of the trees which initially survived on the untreated controls were too weak to withstand subsequent environmental stresses, which were more severe on the control plots than on treated plots. The treatments resulted in an increase in nutrient availability and a decrease in toxicity problems (Goodman 1978). Also, the herbaceous cover increased as additional soil amendments were applied, thereby making the microclimate more suitable for tree survival and growth.

In the analysis of variance for the 1989 survival data, only container type and the container by location interaction were significant. Bare-root nursery stock had an overall mean survival of 75% compared to 68% for the Japanese paperpots (Table 3). However, the Renovo plots were almost completely responsible for this difference. There, the mean survival for bare-root stock was 75%, well above the 55% survival for Japanese paperpots. In fact, the Renovo plots planted with Japanese paperpots and receiving no amendment were the only ones with a mean survival of less than 50%.

Table 1. Selected chemical characteristics of minesoils at study sites (from Goodman 1978).

| Site | pH | Bray No. 1 P | Exchangeable Cations | | | | | Specific Conductivity |
|---|---|---|---|---|---|---|---|---|
| | | | K | Mg | Ca | Mn | Al | |
| | | kg/ha | - - - - - - me/100 g - - - - - - | | | | | mmhos/cm |
| Kylertown | 3.3 | 0.96 | 0.077 | 0.47 | 0.23 | 0.088 | 2.95 | 0.45 |
| Clarion | 3.2 | 2.41 | 0.078 | 1.12 | 0.89 | 0.066 | 3.35 | 0.76 |
| Renovo | 3.0 | 0.74 | 0.085 | 0.61 | 0.69 | 0.012 | 2.95 | 0.62 |

Table 2. Mean survival of red pine trees by date and amendment.

| Date | Soil Amendment* None | L | L+F | L+F+M | Mean |
|---|---|---|---|---|---|
| | % | % | % | % | % |
| May, 1977 | 91 | 86 | 88 | 84 | 87 |
| Aug, 1977 | 76 | 76 | 78 | 77 | 77 |
| May, 1978 | 72 | 74 | 76 | 76 | 74 |
| Aug, 1978 | 70 | 72 | 74 | 76 | 72 |
| Aug, 1979 | 69 | 72 | 74 | 76 | 72 |
| Dec, 1980 | 67 | 71 | 74 | 76 | 72 |
| May, 1989 | 64 | 71 | 74 | 76 | 71 |

* L = lime; F = fertilizer; M = mulch.

Possibly the lower pH and available P levels at Renovo (Table 1) as compared to the other locations were responsible for the lower survival rate of Japanese paperpot seedlings. If so, this would indicate that containerized seedlings may be less resistant to environmental stresses than bare-root stock. This needs to be verified in other studies before any changes in planting recommendations are made.

The seed-mix treatments were very similar with respect to both mean survival (71, 73, and 70%) and mean height (3.8, 3.9, and 3.8 m) for the control, KY-31 tall fescue + birdsfoot trefoil, and deertongue grass + birdsfoot trefoil, respectively. Furthermore, the analyses of variance revealed no significant interactions involving seed mix. Consequently, data for the different seed-mix treatments were combined in Table 2. The lack of a seed-mix effect on tree survival and growth is not surprising since the lime + fertilizer + mulch treatment had, at the end of the first growing season following planting, the best mean cover, which was only 32%. Such a sparse cover would provide little competition to the established tree seedlings.

The mean tree height for all plots combined was 3.9 m (Table 3). In 1981, four growing seasons following planting, the mean tree height was only 0.5 m, indicating that the annual height growth in recent years has been considerably better than in the early years.

Soil amendments have had a significant impact on tree height (Table 3). Trees on plots receiving both lime and fertilizer were almost 50% taller than those on untreated plots. Trees on plots receiving lime only were intermediate in height.

During the first year, when height growth averaged only 3.6 cm, there were very small differences among amendments (Goodman 1978). Mean height growth doubled in the second year, with trees on limed and fertilized plots having more than twice the height growth of those on the untreated control. Over the next few years, height growth continued to increase, and the limed and fertilized plots maintained their two-to-one advantage over the control plots with respect to height growth. Although annual height growth was not measured in 1989, it appears from visual observations that the trees are still benefiting from the lime and fertilizer treatment.

By the end of the first growing season following planting, mean herbaceous cover was 0% in the non-amended control plots, less than 5% in the lime, 21% in the lime + fertilizer, and 32% in the lime + fertilizer + mulch plots (Goodman 1978). These values were very similar to those of Vandevender and Sencindiver (1982) for acid mine spoils in West Virginia treated with various combinations of lime and fertilizers.

The fescue mix had a better mean cover (33%) than the deertongue mix (20%). There was some natural invasion on the lime + fertilizer and lime + fertilizer + mulch plots by both herbaceous and woody species. In 1989, these plots contained considerable numbers of trees, mainly aspen (Populus spp.) at the Clarion and Kylertown sites and black locust (Robinia pseudoacacia) at Renovo. This aspect of the study will be covered more fully in a subsequent paper.

## Supplemental Studies

The almost complete failure of the multi-seedling pot phase provided an opportunity for two supplemental studies. The objectives were: (1) to determine how representative 1977 was with respect to seedling survival and subsequent height growth on non-amended sites; and (2) to determine how types of container systems other than Japanese paperpots would perform in comparison to 2 + 0 nursery-grown seedlings planted in a bareroot condition.

### Annual Variability Study

This study was conducted at the Kylertown and Clarion sites on the control and lime split plots that had initially been planted with seedlings from multi-seedling pots. Vegetative cover on these covers was less than 5%. Twenty nursery-grown 2 + 0 red pine seedlings were planted in each split plot in the springs of 1978 and 1979 to complement those planted in 1977. Thus there was a total of 480 seedlings in 24 split plots.

Mean survival of these plantings in 1989 was 76% for the 1977 planting, 82% for the 1978 planting, and 90% for the 1979 planting. This high survival for three consecutive years even on untreated plots indicates that many abandoned mine

Table 3. Mean survival and height in spring 1989 of red pine trees for each location, amendment, and container type*.

| Location | Amendment** | Mean Survival | | | Mean tree height | | |
|---|---|---|---|---|---|---|---|
| | | BR*** | JPP*** | All | BR*** | JPP*** | All |
| | | % | % | % | m | m | m |
| Kylertown | None | 78 | 74 | 76 | 2.8 | 2.8 | 2.8 |
| | L | 88 | 80 | 84 | 3.3 | 3.3 | 3.4 |
| | L+F | 87 | 79 | 83 | 4.3 | 4.1 | 4.2 |
| | L+F+M | 84 | 80 | 82 | 4.4 | 4.4 | 4.4 |
| | All | 84 | 78 | 81 | 3.7 | 3.7 | 3.7 |
| Clarion | None | 54 | 78 | 66 | 3.4 | 3.5 | 3.4 |
| | L | 60 | 57 | 58 | 3.5 | 4.0 | 3.7 |
| | L+F | 82 | 66 | 74 | 4.6 | 4.4 | 4.5 |
| | L+F+M | 70 | 82 | 76 | 4.3 | 4.6 | 4.5 |
| | All | 67 | 71 | 69 | 4.0 | 4.1 | 4.0 |
| Renovo | None | 60 | 42 | 51 | 2.4 | 3.3 | 2.8 |
| | L | 80 | 61 | 70 | 3.5 | 4.1 | 3.8 |
| | L+F | 74 | 58 | 66 | 4.5 | 4.4 | 4.4 |
| | L+F+M | 83 | 58 | 71 | 4.2 | 4.4 | 4.3 |
| | All | 74 | 55 | 65 | 3.7 | 4.1 | 3.9 |
| All | None | 64 | 65 | 64 | 2.8 | 3.2 | 3.0 |
| | L | 76 | 66 | 71 | 3.5 | 3.3 | 3.7 |
| | L+F | 81 | 68 | 74 | 4.5 | 4.3 | 4.4 |
| | L+F+M | 79 | 73 | 76 | 4.3 | 4.4 | 4.4 |
| | All | 75 | 68 | 71 | 3.8 | 4.0 | 3.9 |

* Value for different seed mixes have been combined for simplicity.
** L = lime; F = fertilizer; M = mulch.
*** BR = bare-root nursery stock; JPP = Japanese paperpot.

lands in Pennsylvania can be successfully revegetated using conventional nursery-grown seedlings of species adapted to the site, such as red pine. However, spring weather conditions during the 1977-1979 period were generally favorable for seedling establishment. Droughts did occur during the spring of 1977, but the seedlings had been planted when soil conditions were near optimum.

As expected, tree height varied significantly with year of planting. However, mean height per year is nearly the same for the three different years. Results with respect to location and to application of lime were similar to those found in the main study.

Container Comparison Study

This supplemental study, located only at the Kylertown site, consisted of two parts: (1) planting of five different seedling types on a non-seeded split plot within each of the main soil amendment plots; and (2) planting of the same five seedling types on the various herbaceous seed mix split plots within two of the main soil amendment plots (lime + fertilizer and lime + fertilizer + mulch).

The seedling types were: (1) 2 + 0 nursery-grown seedlings; (2) multi-seedling pot seedings; (3) Tinus Spencer-Lemaire book container seedlings; (4) Colorado container seedlings; and (5) tarpaper container seedlings. The first three were the same as those involved in the main study. The Colorado container is a styroblock with 30 cavities each 5 x 5 x 19 cm in size. The tarpaper container is made by folding and stapling a sheet of asphalt-based roofing paper to form a container 5 x 5 cm in cross-section and 18 cm in length. Its advantages and disadvantages are discussed by Strachan (1974).

In both trials, a row of four trees of each seedling type were planted in each split plot in the spring of 1978. Survival and tree height data were collected in the spring of 1989 and analyzed by analysis of variance techniques.

Mean survival was good for all seedling types in both trials (Table 4)

Table 4. 1989 mean survival and tree height for supplemental studies of seedling types.

| Seedling Type | Amendment trial Survival | Amendment trial Tree Height | Seed mix trial Survival | Seed mix trial Tree Height |
|---|---|---|---|---|
| | % | m | % | m |
| Nursery bare root | 91 | 3.3 | 83 | 4.0 |
| Multi-seedling pot | 91 | 3.1 | 88 | 3.8 |
| Spencer-Lemaire book | 88 | 3.3 | 83 | 3.7 |
| Colorado styroblock | 91 | 3.3 | 92 | 4.0 |
| Tarpaper | 94 | 3.1 | 83 | 3.8 |

and did not vary significantly from one another. Within each trial, mean tree heights were also uniform among seedling types. Survival was much better than reported by Davidson and Sowa (1974) in earlier trials with container-grown seedlings on acid mine spoils in Pennsylvania. However, the container types we used were less susceptible to frost heaving, a major problem in their study.

The results with respect to soil amendment for all five container types and two replications combined were:

| Amendment | Mean Survival | Mean Height |
|---|---|---|
| | % | m |
| None | 78 | 2.5 |
| Lime | 100 | 3.0 |
| Lime+fertilizer | 92 | 3.8 |
| Lime+fertilizer+mulch | 92 | 3.8 |

Mean survival for the non-amended plots was significantly less than for the treated plots. Mean heights increased with each additional amendment, with the exception of the mulch application. These patterns were similar to those found in the main study and confirmed the need for lime and fertilizer use in mine land reclamation as emphasized by Mays and Bengtson (1978).

One difference between the main study and the supplemental study was the difference in mean survival between the lime + fertilizer and lime + fertilizer + mulch treatments. Survival was greater for the lime + fertilizer + mulch treatment in the main study but less in the supplemental study. The probable cause was that the supplemental study planting was made after a vegetative cover, albeit sparse, had been established. As shown below, there seems to be an inverse relationship between mean survival and percent cover.

| Amendment | Seed Mix | Aug. 1977 Mean Cover | Mean 1989 Survival |
|---|---|---|---|
| | | % | % |
| L+F | None | 0 | 92 |
| L+F | Deertongue | 14 | 95 |
| L+F | Fescue | 21 | 85 |
| L+F+M | None | 0 | 92 |
| L+F+M | Deertongue | 23 | 80 |
| L+F+M | Fescue | 37 | 70 |

## Conclusions

1. Many of the abandoned mine lands in Pennsylvania can be revegetated by merely planting species adapted to the site, such as red pine.

2. Application of lime and fertilizer, however, will increase tree growth substantially for at least a decade following planting. Long-term survival may also be increased.

3. On abandoned mine lands, application of lime and fertilizer will probably be required to obtain an herbaceous cover. Even then, the cover will likely be sparse. The application will also promote natural invasion of the sites by woody and herbaceous species and result in greater biotic diversity.

4. The competition for water and nutrients, and possibly allelopathic effects, resulting from an established vegetative cover, even a sparse one, on abandoned mine lands may result in decreased survival of newly planted seedlings.

5. The performance of containerized seedlings on abandoned mine lands in Pennsylvania is similar to that of conventional nursery-grown stock planted in a bare-root condition. No one type of container provides substantially better survival and growth than the others. The choice of seedling type for planting on abandoned mine lands in Pennsylvania therefore depends on factors such as availability, cost, ease of planting, and care needed during planting. These factors were not investigated in our study.

6. Seedling quality is probably more important than seedling type in the successful establishment of trees on abandoned mine lands.

## Acknowledgments

We are indebted to Stephen D. Goodman, a former M.S. candidate in the School of Forest Resources, The Pennsylvania State University. He was instrumental in locating and laying out the study areas, making the treatments,

planting the tree seedlings and herbs, and collecting and analyzing performance data during the first year following planting.

During the first few years, the project was funded in part through U.S.D.A. Cooperative State Research Service Grant No. 684-15,9.

## Literature Cited

Beyer, Lawrence E. and Russell J. Hutnik. 1969. Chemical properties of toxic strip-mine spoil banks in Pennsylvania. Preprints of Papers Presented at the 157th National Meeting of the American Chemical Society, Division of Fuel Chemistry, Symposium on Pollution Control in Fuel Combustion, Processing, and Mining (Minneapolis, MN. April 13-18) 13(2):120-123.

Davidson, Walter H. and Edward A. Sowa. 1974. Early attempts to vegetate coal-mine spoils with container-grown seedlings p. 372-376. In Tinus, Richard W., William I. Stein, and William E. Balmer (Eds.). Proceedings North American Containerized Forest Tree Seedling Symposium, (Denver, CO. Aug. 26-19) Great Plains Agricultural Council Publication No. 68, 458 p. Lincoln, NE.

Goodman, Stephen D. 1978. Container-grown red pine seedlings on strip-mine spoils in Pennsylvania. M.S. thesis. 115 p. The Pennsylvania State University, University Park, PA.

Grim, E. C. and R. D. Hill. 1974. Environmental protection in surface mining of coal. U.S. Environmental Protection Agency, 277 p. Cincinnati, OH.

Hoedemaker, E. 1974. The Japanese paperpot system. p. 214-216. In Tinus, Richard W., William I. Stein, and William E. Balmer (Eds.). Proceedings North American Containerized Forest Tree Seedling Symposium, (Denver, CO. Aug. 26-19) Great Plains Agricultural Council Publication No. 68. 458 p. Lincoln, NE.

Horn, M. L. and W. W. Ward. 1969. The revegetation of highly acid spoil banks in the bituminous coal region of Pennsylvania. Special Research Report SR-71. 65 p. The Pennsylvania State University, University Park, PA.

Mays, D. A. and G. W. Bengtson. 1978. Lime and fertilizer use in land reclamation in humid regions. p. 307-328. In Schaller, Frank W. and Paul Sutton (Eds.). Proceedings Reclamation of Drastically Disturbed Lands. Symposium (Wooster, OH. Aug. 9-12, 1976) American Society of Agronomy. 742 p. Madison, WI.

Spencer, Henry A. 1974. To "Engineer" the container, p. 229-232. In Tinus, Richard W., William I. STein, and William E. Balmer (Eds.). Proceedings North American Containerized Forest Tree Seedling Symposium, (Denver, CO. Aug. 26-19) Great Plains Agricultural Council Publication No. 68. 458 p. Lincoln, NE.

Strachan, Marvin D. 1974. Tar paper containers. p. 209-210. In Tinus, Richard W., William I. Stein, and William E. Balmer (Eds.). Proceedings North American Containerized Forest Tree Seedling Symposium, (Denver, CO. Aug. 26-29). Great Plains Agricultural Council Publication No. 68. 458 p. Lincoln, NE.

Tinus, Richard W. 1974. Characteristics of seedlings with high survival potential. p. 276-282. In Tinus, Richard W., William I. Stein, and William E. Palmer (Eds.). Proceedings North American Containerized Forest Tree Seedling Symposium, (Denver, Co. Aug. 26-29) Great Plains Agricultural Council Publication No. 68. 458 p. Lincoln, NE.

Vandevender, John C. and John C. Sencindiver. 1982. The effects of three forms of nitrogen fertilizer, phosphorus and hydrated lime on abandoned mine land reclamation. p. 497-502. In Graves, Donald H. and R. William DeVore. Proceedings 1982 Symposium Surface Mining Hydrology, Sedimentology and Reclamation. (Lexington, KY. Dec. 5-10) 728 p. University of Kentucky, Lexington, KY 40506-0046.

This page intentionally left blank.

# AN EVALUATION OF LIME REQUIREMENT TESTS ON PYRITIC MINESOILS[1]

Peter J. Cagnetta and Everett M. Jencks[2]

Abstract. Soil samples were collected from 6 abandoned minelands. All were extremely acidic with pH values ranging from 2.79 to 4.51. On each soil, nine lime requirement tests were performed. The tests included the Ca(OH)2 titration, Woodruff buffer, Mehlich buffer, the Shoemaker, McLean, and Pratt double buffer, KCl exchangeable acidity, acid-base account (total-S), acid-base account (reduced-S), hydrogen peroxide oxidation, and soxhlet extractions. The soils were then incubated in the greenhouse, each with different application rates of a highly reactive limestone. There were five liming rates per soil with each replicated four times. The moisture content was kept at approximately 33 kPa for six weeks followed by two weeks in which water was not added. This cycle continued for 15 months. At the end of six months and twelve months each treatment was leached with 2L of distilled water. The leachate was analyzed for pH. For this paper, only the six month leachate data was available. Incubation-leaching lime requirement values were based on the near maximum reactivity of the limestone. Laboratory values were related to the incubation values through correlation and regression analysis. The Ca(OH)2 test showed the strongest correlation with the incubation value (r=.95). The soil property most closely correlated with the incubation value was total-S (r=.85).

Additional key words: Acid-base account, buffer, exchangeable aluminum and pyrite.

## Introduction

There are many existing methods which are used to determine the lime requirement (LR) of agricultural soils (Dunn 1943 and Kamprath 1970). Several of the quick-test methods incorporate the use of a buffer solution adjusted to a known pH (Woodruff 1948; Mehlich 1976; McLean et al. 1978). These tests, with varying degrees of accuracy, account for the active and reserve acidity that results from hydrogen and the hydrolysis reactions of aluminum and its polymers.

Most buffer methods are based on the principle that when a buffer solution of a known pH is mixed with an acid soil the pH will decrease linearily with respect to the H content of the soil.

The Woodruff (1948) buffer method incorporates the use of a highly buffered solution that has been shown to be accurate on soils with differing exchange capacities and high lime requirements. The Shoemaker, McLean, and Pratt (1961) (SMP) method is ideally suited for soils with significant amounts of exchangeable Al and little organic matter. The SMP double buffer (McLean et al. 1978) (SMPDB) takes into

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]P. J. Cagnetta is a Graduate Teaching Assistant, Division of Plant and Soil Sciences and E. M. Jencks is an Assoc. Professor of Agronomy, Division of Plant and Soil Sciences, West Virginia University, Morgantown, WV 26505.

account the buffering capacity of individual soils. For sandy soils the Yuan (1974) method is often employed while Adams and Evans (1962) devised a buffer for Red-Yellow Podzolic Ultisols.

Numerous comparison studies have been conducted on various types of soils. Alabi et al. (1986) evaluated eight buffer methods on coarse-textured soils of Nebraska. Kenney and Corey (1963) conducted an investigation on 26 Wisconsin soils. Trans and van Lierop (1981) determined the accuracy of buffer methods on Canadian soils. Lateritic soils of India were utilized in a study by Savant and Kibe (1971). When these methods were applied to acidic minesoils Mays and Bengston (1978) have indicated that these methods varied substantially in their prediction of lime requirements. Crews and Vogel (1986) found that the Woodruff and SMP methods underestimate the LR of minesoils containing pyrite.

Often with minesoils there is the added component of acidity generated from the oxidation of pyrite, FeS2, which yields sulfuric acid (Carruccio 1968 and Smith et al. 1970). The potential acidity from the unoxidized pyrite in the soil fraction must be considered in the total acidity when formulating an LR for these soils.

The acid-base account (Smith 1974) has been a widely accepted method for determining the LR of pyritic materials. The estimation of total S by hydrogen peroxide has also been employed, although it is more suited for fresh overburden material (Sobek et al. 1978).

The objectives of this study were to (1) determine the relationship of LR values predicted by nine methods with those by an incubation-leaching technique based on the maximum reactivity of the limestone, (2) determine the extent of correlation between LR values, and (3) determine the correlation of all LR values with various soil chemical properties. This is a preliminary report based on the first set of leachate data collected from an ongoing study.

## Methods and Materials

### Soil Analysis

Samples from six strongly acidic minesoils were collected from abandoned strip mine sites in Monongahela and Preston counties of north central West Virginia. Four sites were completely void of vegetation while the remaining two had little invading vegetation. Samples were collected to a depth of 15 cm. All soils were air-dried at approximately 28 C in the greenhouse and then were passed through a 2 mm sieve. Only the less than 2 mm size fraction was used in the study.

Soil pH was determined with a glass electrode pH meter using a 1:1 soil:water ratio. Organic carbon was determined according to the Walkley-Black method (Allison 1965), cation exchange capacity and exchangeable bases by the NH40Ac (pH 7) method, and exchangeable acidity and Al by KCl extraction as outlined by the Soil Survey Staff (1984). Moisture retention at 33 kPa of pressure was determined using pressure plates and the pipette method was employed to determine particle size distribution. A Leco induction furnace with an automatic titrater was used to determine percent total - S. Sulfate-S was leached from the samples (Smith et al. 1978) and the percent reduced-S was determined again using the induction furnace. Soil characterization results can be found in Table 1.

### Lime Requirement Tests

The Ca(OH)2 titration (Dunn 1943) in conjunction with a five-minute boiling step, (Abruna and Vincente 1955) was used to determine the lime requirements of all soils. Exchangeable acidity was also used as a basis for liming. Buffer pH methods employed

Table 1. Selected physicochemical measurements of the minesoils.

| | | Minesoil Number | | | | | |
|---|---|---|---|---|---|---|---|
| Measurement | Units | 1 | 2 | 3 | 4 | 5 | 6 |
| pH | | 2.79 | 2.80 | 3.36 | 3.49 | 4.39 | 4.51 |
| Organic Matter | % | 0.43 | 0.31 | 0.20 | 0.29 | 0.58 | 0.47 |
| CEC | cmol/kg | 10.50 | 13.30 | 9.30 | 16.20 | 9.50 | 8.60 |
| Exch Acidity | cmol/kg | 10.93 | 13.72 | 9.38 | 24.57 | 5.50 | 10.53 |
| Exchangeable Al | cmol/kg | 6.30 | 6.80 | 3.70 | 10.10 | 1.58 | 4.60 |
| Total S | % | 1.51 | 0.71 | 0.13 | 0.26 | 0.03 | 0.07 |
| Reduced S | % | 0.41 | 0.20 | 0.04 | 0.04 | 0.03 | 0.03 |
| Clay | % | 29.00 | 18.00 | 24.00 | 39.00 | 26.00 | 39.00 |

Table 2. Lime requirement determination by various methods.

| Measurement | Minesoil Number 1 | 2 | 3 | 4 | 5 | 6 |
|---|---|---|---|---|---|---|
| | Mg/ha | | | | | |
| Ca(OH)2 | 22.06 | 17.47 | 12.10 | 20.38 | 12.54 | 14.90 |
| Woodruff | 33.50 | 33.15 | 23.41 | 46.66 | 13.22 | 18.08 |
| SMP-DB | 37.32 | 28.36 | 20.76 | 54.30 | 13.06 | 25.56 |
| Mehlich | 17.45 | 18.39 | 13.69 | 25.63 | 7.91 | 12.97 |
| Ex Acidity | 22.56 | 30.87 | 23.23 | 60.50 | 14.95 | 25.07 |
| Soxhlets | 75.18 | 31.08 | 20.65 | 21.56 | 9.24 | 15.12 |
| ABA-TS | 124.23 | 69.78 | 21.37 | 38.75 | 7.71 | 15.57 |
| ABA-RS | 47.17 | 33.67 | 15.21 | 23.27 | 7.08 | 12.66 |
| H202 | 20.14 | 10.51 | 3.23 | 4.95 | 0.35 | 4.91 |
| Incubation | 34.27 | 23.74 | 12.10 | 24.64 | 6.50 | 18.37 |

included the Woodruff (1948), the Mehlich (1976), and the Shoemaker, McLean and Pratt double-buffer (1978). The acid-base account utilizing both total and reduced-S also served to develop an LR, as was the hydrogen peroxide method. The use of soxhlet extractors (Stiller et al. 1988) was employed to develop an LR based on the sulfate-S concentration in the leachate. Due to time restraints this procedure was conducted after the incubation began. LR values as determined by the various methods can be found in Table 2.

Incubation Procedure

Each soil was limed at five different rates with each rate replicated four times. The rates encompassed the range of LR values as determined by the laboratory tests and can be found in Table 3. Each treatment contained 0.5 kg of soil. Polyethylene containers fitted with rubber spouts were used to contain the soils. The spouts were sealed with polyethylene clamps. Glass wool was used to line the bottom of the containers. A layer of acid-washed sand was placed on top. The soil-lime mixtures were then added. The soils were moistened to approximately 33 kPa for six weeks. This was followed by two weeks in which no water was added. Alternate wetting and drying periods were chosen to simulate natural conditions. This cycle continued for fifteen months and at the end of the sixth and twelfth months each treatment was leached with 2L of distilled water. This value was chosen inorder to remove many of the salts that accumulated during the incubation.

Leachate and Limestone Analysis

The leachate was analyzed for pH with a glass electrode pH meter. The lime used in the study was a pulverised agricultural limestone obtained from German Valley Limestone in Riverton, WV. Total Ca and Mg were determined through atomic adsorption spectrometry. Its CaCO3 equivalent was determined and its particle-size distribution was defined using 20, 35, 60, 140, 200, 270, and 300 mesh sieves. The activity was derived from a chart for evaluating agricultural limestones devised by Schollenberger and Salter (1943).

## Results and Discussion

Lime Requirement Test Comparisons

The correlation coefficients among

Table 3. Incubation lime application rates.

| Lime Rate | Minesoil Number 1 | 2 | 3 | 4 | 5 | 6 |
|---|---|---|---|---|---|---|
| | Mg/ha | | | | | |
| 1 | 0 | 0 | 0 | 0 | 0 | 0 |
| 2 | 17.0 | 11.9 | 6.1 | 12.3 | 6.5 | 10.3 |
| 3 | 34.3 | 23.7 | 12.1 | 24.6 | 9.6 | 14.3 |
| 4 | 53.1 | 32.7 | 20.2 | 44.4 | 13.0 | 18.4 |
| 5 | 132.8 | 74.6 | 25.1 | 58.0 | 14.1 | 27.3 |

Table 4. Correlation coefficients among laboratory LR values.

| Method | WD | SMPD | MEH | ACID | SOX | ABAT | ABAR | H202 |
|---|---|---|---|---|---|---|---|---|
| CA | 0.81 | 0.84 | 0.80 | 0.53 | 0.75 | 0.81 | 0.84 | 0.79 |
| WD | | 0.94 | 0.98 | 0.86 | 0.41 | 0.52 | 0.62 | 0.44 |
| SMPD | | | 0.97 | 0.89 | 0.36 | 0.43 | 0.51 | 0.39 |
| MEH | | | | 0.89 | 0.37 | 0.46 | 0.57 | 0.40 |
| ACID | | | | | -0.07 | 0.03 | 0.15 | -0.03 |
| SOX | | | | | | 0.97 | 0.93 | 0.97 |
| ABAT | | | | | | | 0.99 | 0.98 |
| ABAR | | | | | | | | 0.96 |

laboratory LR tests can be found in Table 4. The Woodruff, SMP-DB, and Mehlich methods were all significantly related with coefficients > .94. The three buffers were all highly correlated with the Ca(OH)2 titration and exchangeable acidity as evidenced by coefficients greater than .80. The relationship was not as strong between the Ca(OH)2 and exchangeable acidity (r=.53). The ABA-TS, ABA-RS, soxhlet extractions, and the H202 oxidation were all significantly correlated (r's > .93). These methods all account for the oxidation of residual pyrite. There was a limited realtionship between the three buffers and the four sulfur methods as the coefficients ranged from .36 to .62. There was virtually no relationship between these tests and exchangeable acidity (r's < .15). The relationships between the four sulfur tests and the Ca(OH)2 were strong (r's > .75).

Limestone Analysis Results

The CaCO3 equivalent was determined to be 96.7%. The calcitic limestone contained 42% Ca and 0.4% Mg. The sieve analysis showed that 100% passed a 60 mesh sieve, 58% a 140 mesh sieve, 35% a 200 mesh sieve, 23% a 270 mesh sieve and 3% passed a 300 mesh sieve. The activity was calculated to be .96, indicating a highly reactive limestone.

Determination of Incubation Leaching LR Values

The LR values were based on the near maximum reactivity of the limestone. This was determined by plotting the leachate pH values for a particular soil against the lime application rates. The plot for soil 1, found in Figure 1, is typical of the results obtained for all soils. All soils reached a point at which increasing rates did not increase the pH significantly. The four most acidic soils reached a peak reactivity with the third liming rate while soils 5 and 6 peaked with the second and fourth rates. Near maximum reactivity pH values ranged from 7.28 to 8.05. They are referred to as near maximum reactivity values because they are below the theoretical limits on pH for pure CaCO3.

LR Test Values vs Incubation-Leaching LR Values

Table 5 shows the correlation coefficients and linear regression equations relating the laboratory LR values (LLR) and incubation-leaching LR values (ILR). The regression equations utilized the LLR values as the dependent variables and the ILR values as the independent variable. Regression lines can be found in Figures 2 and 3. If all lab methods were to precisely predict the LRs of the soils the intercept of the equations would be 0 and the slopes would be equal to 1. The data indicates that the intercepts vary from 0 and the slopes are not equal to 1.

The strongest relationship with the ILR was with the Ca(OH)2 (r = .95). The ABA-RS, H202, ABA-TS, and the soxhlet extractions were all significantly correlated with the ILR (r's = .92, .90, .89, and .84). The three buffer tests all exhibited the same strong relationship with the ILR (r = .73). The exchanageable acidity was the least correlated (r = .41). All correlations were significant at the 0.01 probability level except the exchangeable acidity.

Despite having correlation coefficients of similar magnitude the regression equations were substantially different. This may be attributed to the strong possibility of continuing acidification at different rates of the incubated samples due to pyrite oxidation. There are also differences in the nature of the acidity in the soils and in the different abilities of the LLR tests to react with these acids.

In the Ca(OH)2 test, the base reacts with both acidity in soil solution and on exchange sites. The three buffer solutions react with acidity in a similar manner. They neutralize both solution and adsorbed H ions and also contain displacing cations which remove Al compounds from exchange sites thus allowing for their neutralization with the buffer

# Figure 1. Soil 1

## Liming rate vs Leachate pH

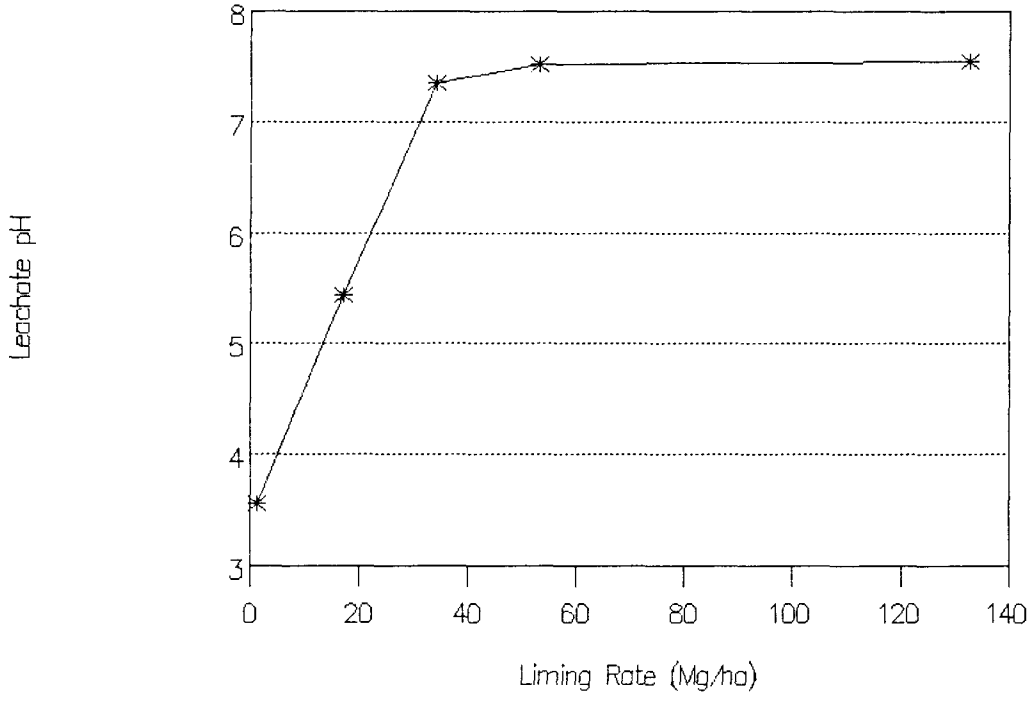



Table 5. Correlations and regressions for Lab LRs vs Inc LRs.

| Method | r | Linear Regression Equation |
|---|---|---|
| Ca(OH)2 | *.95 | y = 8.71 + .39x |
| ABA-RS | *.92 | y = -4.67 + 1.4x |
| H202 | *.90 | y = -5.53 + .65x |
| ABA-TS | *.89 | y = -32.98 + 3.97x |
| Soxhlet | *.84 | y = -11.67 + 2.03x |
| Woodruff | *.73 | y = 10.08 + .90x |
| SMP-DB | *.73 | y = 8.58 + 1.07x |
| Mehlich | *.73 | y = 7.13 + .45x |
| Exch. Acid | .41 | y = 12.87 + .24x |

* Significant at the 0.01 probability level.

Figure 2. Fitted Regression Equations

Relating LLR and ILR

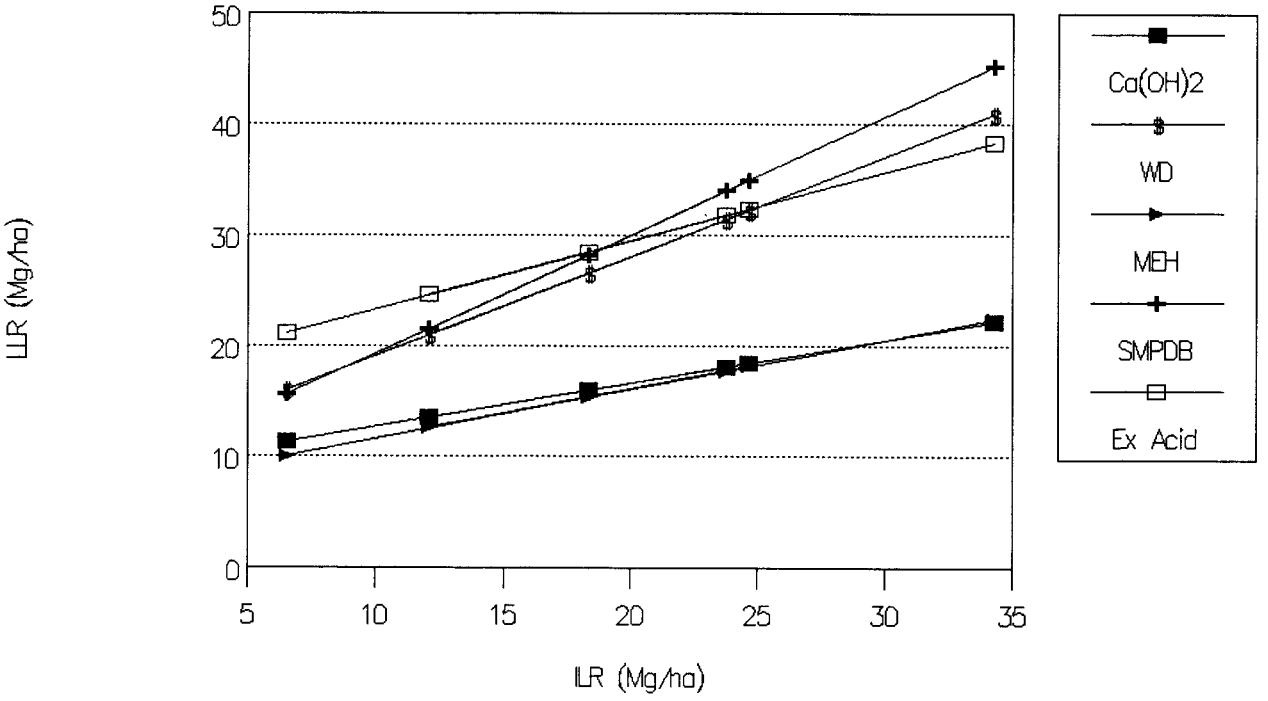
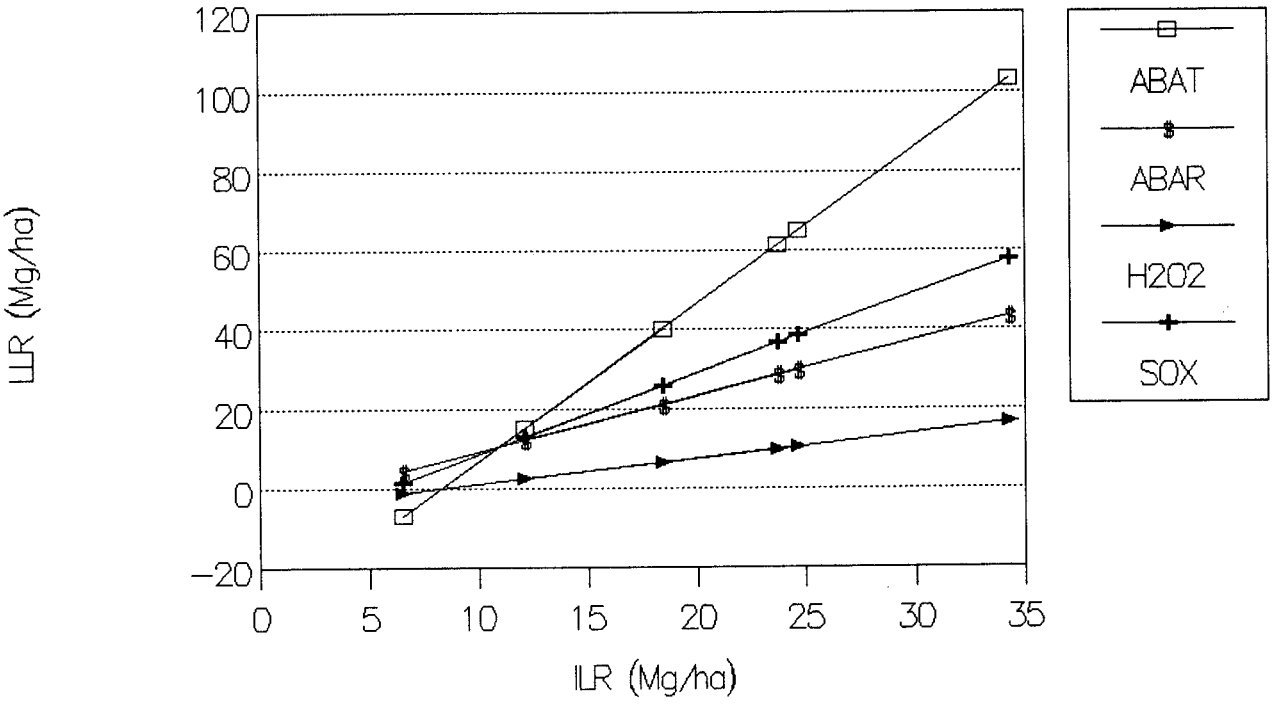


Figure 3. Fitted Regression Equations

Relating LLR and ILR



solutions.

Exchangeable H and Al are displaced by K ions in the KCl extraction method. An amount of CaCO3 equivalent to the acidity generated from the full hydrolysis of the Al and displaced H in the extract is used as a basis for liming.

The acid-base account balances the maximum potential acidity resulting from the complete oxidation of iron sulfides against a soil's ability to neutralize this acid. Using reduced S measurements to predict the potential acidity, as opposed to total S, is more applicable to weathered minesoils since the SO4-S will no longer generate acidity. Acidic minesoils quite commonly lack basic minerals which create an intrinsic CaCO3 equivalent deficit. As a result, free acidity is often detected in determining the neutralization potential. By combining an amount of CaCO3 equivalent to the total acidity resulting from the complete oxidation of reduced S with an amount needed to neutralize the free acids, an LR is determined. In the H202 method an LR was based on the potential acidity from pyrite oxidation. Through the use of soxhlet extractors which simulate geochemical weathering of the minesoils alternate heating and leaching cycles maximize pyrite oxidation. An LR is formulated based on the amount of SO4-S released.

## LLR and ILR Test Values vs Soil Properties

Correlation coefficients of all LR values with selected soil properties can be found in Table 6. The LR values from the three buffer methods and the Ca(OH)2 titration were significantly correlated with CEC values and levels of exchangeable Al. There was also a strong relationship between CEC and exchangeable acidity (r = .90). The Ca(OH)2 titration was also highly correlated with total and reduced-S (r = .77 and .70). The four S-tests showed the strongest correlations with levels of total and reduced-S, and showed strong negative correlations with soil pH. The ILR displayed the strongest correlation with the level of total-S (r = .85).

The reactivity of the limestone may not only have been controlled by the levels of acidity present, but also by the amount of SO4-S present. The precipitation of gypsum, a CaSO4 compound, with its subsequent establishment of an equilibrium with the soil solution may have slowed the dissolving of the CaCO3 due to the common ion effect.

## Conclusions

Based on the data available from this study the Ca(OH)2 LR test was the most strongly correlated with the ILR (r = .95). The four LR tests which accounted for the maximum potential acidity from pyrite were all strongly correlated (r's > .93). There were also strong correlations among the three buffer tests and exchangable acidity (r's > .86). The four S tests and the buffers were less well correlated. The soil property most strongly correlated with the ILR was total-S (r =.85). The SO4-S content of the soils appeared to be one of the factors controlling the solubility fo the limestone. At the conclusion of the incubation, soil analysis data will be available and additional conclusions will be drawn.

## References Cited


Adams, F., and C. E. Evans. 1962. A rapid method for determining lime requirements of Red-Yellow Podzolic soils. Soil Sci. Soc. Am. Proc. 26:355-357.

Alabi, K. E., R. C. Sorenson, D. Knudsen, and G. W. Rehm. 1986. Comparison of several lime requirement methods of coarse-textured soils of northeastern Nebraska. Soil Sci. Soc. Am. J. 50:937-941.


Table 6. Correlation coefficients of LR values and soil properties.

| Method | Clay | CEC | pH | Al | Tot S | Red S |
|---|---|---|---|---|---|---|
| Ca(OH)2 | 0.22 | 0.82 | -0.60 | 0.58 | 0.77 | 0.70 |
| Woodruff | 0.18 | 0.90 | -0.66 | 0.80 | 0.43 | 0.33 |
| SMP-DB | 0.51 | 0.93 | -0.42 | 0.91 | 0.35 | 0.24 |
| Mehlich | 0.33 | 0.89 | -0.57 | 0.86 | 0.37 | 0.26 |
| Exch. Acid | 0.48 | 0.90 | -0.21 | 0.90 | -0.06 | -0.17 |
| Soxhlets | -0.19 | 0.41 | -0.72 | 0.05 | 0.98 | 0.97 |
| ABA-TS | 0.27 | 0.47 | -0.81 | 0.08 | 0.99 | 0.98 |
| ABA-RS | 0.26 | 0.51 | -0.86 | 0.16 | 0.96 | 0.94 |
| H202 | -0.19 | 0.37 | -0.73 | 0.22 | 0.98 | 0.97 |
| Incubation | 0.13 | 0.64 | -0.69 | 0.41 | 0.85 | 0.79 |

Allison, L. E. 1965. Organic carbon. In C. A. Black et al. (ed.) Methods of soil analysis, part 2. Agronomy 9:1376-1379. American Society of Agronomy, Madison, Wisconsin.

Caruccio, F. T. 1968. An evaluation of factors affecting acid mine drainage and the groundwater interactions in selected areas of western Pennsylvania. p. 107-151. Second Symposium on Coal Mine Drainage Research. (Pittsburgh, PA).

Crews, J. T. and W. G. Vogel. 1985. Evaluation of lime requirement tests for minesoils. Environmental Geochemistry and Health. 7(2):51-55.

Dunn, L. E. 1943. Lime requirement determinations of soils by means of titration curves. Soil Sci. 56:341-351.

Kamprath, E. J. 1970. Exchangeable aluminum as a criteria for liming acid leached soils. Soil Sci. Soc. Am. Proc. 34:252-254.

Keeney, D. R., and R. B. Corey. 1963. Factors affecting lime requirements of Wisconsin soils. Soil Sci. Soc. Am. Proc. 27:277-280.

Mays, D. A. and G. W. Bengston. 1978. Lime and fertilizer use in land reclamation in humid regions. In Schuler, F. W. and Sutton, Paul (Editors) Reclamation of Drastically Disturbed Lands. Am. Soc. of Agr. p. 743.

McLean, E. O., D. J. Eckert, G. Y. Reddy, and J. F. Trierweiller. 1978. An improved soil lime requirement method incorporating double-buffer and quick-test features. Soil Sci. Soc. Am. J. 42:311-316.

Mehlich, A. 1976. New buffer pH method for rapid estimation of exchangeable acidity and lime requirement of soils. Comm. in Soil Sci. and Plant Anal. 7:637-652.

Savant, N. K., and M. M. Kibe. 1971. Acidity in Lateritic rice soils as estimated by various methods. Soil Sci. Soc. Am. Proc. 35:845-846.

Schollenbrger, C. J. and R. M. Salter. 1943. A chart for evaluating agricultural limestone. Am. Soc. Agr. Proc. 35:955-966.

Shoemaker, H. E., E. O. McLean, and P. F. Pratt. 1961. Buffer methods for determination of lime requirements of soils with appreciable amounts of exchangeable aluminum. Soil Sci. Soc. Am. Proc. 25:274-277.

Smith, E. E., and L. S. Shumate. 1970. Sulfide to sulfate reaction mechanism. Water Pollution Control Series. 14010 FRS os/70, E.P.A. Superintendent of Documents. 115 p. Wash. D.C.

Smith, R. M., W. E. Grube, Jr., T. Arkle, Jr., A. Sobek. 1974. Mine spoil potentials for soil and water quality. EPA-670/2-74-070. USEPA, Cincinnati, Ohio.

Sobek, A. A., W. A. Schuller, J. R. Freeman and R. M. Smith. 1978. Field and Laboratory Methods APplicable to Overburdens and Minesoils. EPA-600/2-78-054. U.S.E.P.A., Cincinnati, Ohio.

Soil Survey Staff. 1984. Procedures for collecting soil samples and methods of analysis for soil survey. Soil Survey Investigations Report No. 1, USDA Soil Conservation Service, Wash., D.C.

Stiller, A. H., J. J. Renton, and T. E. Rymer. 1988. A laboratory procedure to evaluate the acid producing potential of coal associated rocks. Mining Science and Technology. 7:227-235.

Trans, T. S. and W. van Lierop. 1982. Lime requirement determination for attaining pH of 5.5 and 6.0 of coarse-textured soils using buffer-pH methods. Soil Sci. Soc. Am. J. 46:1008-1014.

Woodruff, C. M. 1948. Testing soils for lime requirement by means of a buffered solution and the glass electrode. Soil Sci. 66:53-66.

Yuan, T. L. 1974. A double buffer method for the determination of lime requirement of acid soils. Soil Sci. Soc. Am. Proc. 38:437-441.

This page intentionally left blank.

# SURVIVAL AND GROWTH OF BIGTOOTH ASPEN ON ACIDIC SURFACE-MINE SOILS AS INFLUENCED BY PISOLITHUS TINCTORIUS AND NITROGEN AND PHOSPHORUS FERTILIZATION[1]

G. DeMuro, E. M. Jencks, and D. F. Hindal[2]

Abstract. Bigtooth aspen seedlings, Populus grandidentata, non-inoculated or inoculated with Pisolithus tinctorius (Pt) were transplanted to an acidic, infertile abandoned minesoil in May 1984. One year later 71 percent of the seedlings were surviving with no seedling losses between May and August of 1985. Nitrogen-fertilized Pt-inoculated seedlings had the lowest survival rate. The only significant growth increase occurred with treatment of N and P together for non-inoculated seedlings. Growth of aspen after 15 months showed no difference between uninoculated and Pt inoculation.

Additional key words: Aluminum, bulk density, iron, overburden, Populus grandidentata.

## Introduction

Many areas surface mined for coal in Appalachia before the enactment of the Surface Mining Control and Reclamation Act of 1977 (PL95-77) have been abandoned and are now barren; hence, they are unsightly, desolate, and subject to severe erosion. Approximately 6100 hectares of such land are present just in West Virginia (B. Midcap, West Virginia Department of Natural Resources, 1984, personal communication).

Rather than revegetating abandoned minesoils with grass-legume mixtures which require periodic treatments with lime and fertilizer, a better alternative might be to return them to hardwood forests, a condition prevalent in Appalachia. Unfortunately, abandoned and unamended minesoils are acidic and low in nutrients, conditions that slow the establishment of most forest tree seedlings. Success in establishing native forest species, except black locust, (Robinia pseudoacacia, on minelands in Appalachia has been limited (Bramble 1952; Bramble and Ashley 1955; Schramm 1966; Marx 1980) without the application of lime and fertilizer at planting (Limstrom 1948; Austin and Strand 1960; Mays and Bengston 1978). Significant improvements in survival and growth of pine seedlings on disturbed soils have been obtained, however, when their roots were inoculated with symbiotic ectomycorrhizal fungi before transplanting (Marx and Artman 1979; Marx 1980). These symbionts, by extending hyphae into the soil, supplement nutrient absorption by plant feeder roots. Thus, mycorrhizal symbiont inoculation of hardwood seedlings prior to outplanting on low-fertility minesoils might reduce the need for lime and fertilizer. Little work has been reported on the use of mycorrhizal hardwood species for mineland reclamation (Schultz et al. 1981), although some success has been reported for red oak (Quercus rubra) (Beckjord and McIntosh 1983; Beckjord and McIntosh 1984).

Species for forestation of abandoned minesoils should be selected from those present before mining or from those species, such as bigtooth aspen, a pioneer species that readily volunteers on minesoil. Numerous sites in northern West Virginia and western Pennsylvania

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]G. DeMuro is Project Manager, Environmental Resources Management, Inc., 855 Springdale Drive, Exton, PA 19341, E. M. Jencks is Assoc. Professor and D. F. Hindal is Professor, Division of Plant and Soil Sciences, West Virginia University, Morgantown, WV 26506.

where bigtooth aspen has naturally colonized abandoned minesoils have been observed by the authors and others (Bramble and Ashley 1955; Schramm 1966; Medve and Shan 1981). Up to 95 percent cover and good soil stabilization had occurred after within 25 years following replacement of the coal overburden. At these sites, climax species such as oak (Quercus sp.), hickory (Carya sp.), cherry (Prunus sp.), and yellow poplar (Liriodendron tulipifera) are becoming established. In addition to providing good ground cover, bigtooth aspen is also an excellent food source and cover for wildlife, and is valued by the wood and pulp industry.

Woody plant species volunteering on abandoned minesoils have been reported (Schramm 1966; Medve 1973; Marx 1980; Medve and Shan 1981) to require an ectomycorrhizal association for survival and growth. Iskra and Hindal (1985) found that Pisolithus tinctorus (Pt) was widely distributed on roots of volunteer bigtooth aspen on various minesoils in West Virginia. This fungus forms mycorrhizae on a number of tree species, including bigtooth aspen (Marx 1977), growing on coal minesoils or other drastically disturbed sites (Schramm 1966; Medve and Shan 1981). DeMuro (1986) showed that bigtooth aspen seedlings could be successfully inoculated with Pt and that this mycorrhizal fungus improved growth of greenhouse-grown bigtooth aspen seedlings on an acidic, infertile minesoil. "Starter" amounts of soluble N and P (as rock phosphate) enhanced the effect of this mycorrhizal fungus. The objectives of the present study were to determine if inoculating the roots of bigtooth aspen seedlings with Pt prior to transplanting would improve plant survival and growth on an abandoned, infertile minesoil, and to examine the effects of "starter" amounts of N and P on the development of Pt-inoculated and non-inoculated seedlings.

## Materials and Methods

Seeds were collected from a volunteer bigtooth aspen stand in the vicinity of Morgantown, West Virginia. Five hundred plastic pots (5.5 cm x 8.0 cm x 6.0 cm) were sterilized with 10% chlorox solution, and then filled with steam-sterilized peat-vermiculite (PV) (1:1 v/v) to a depth of 5 cm. A 1 cm thick band of the fungus in PV (1:1 v/v) mixture (Isolate #288, Sylvan Spawn Laboratories, Inc., Worthington, Pennsylvania) was placed on the surface of sterilized PV in 250 of the pots and covered with another 2 cm of sterilized PV. Sterilized PV was substituted for the Pt-PV inoculum in 250 pots in order to maintain uniform physical conditions. Ten seeds were placed in each pot. All plants were grown in the greenhouse under incandescent lights for a 16 hour photoperiod with the temperature being maintained at about 24° C. The cultures were watered daily with distilled-deionized water. Two weeks after germination, seedlings were thinned to one per pot. Four, six, and eight weeks after germination, $NH_4NO_3$, $Na_2HPO_4$, and KCl were added in amounts to supply 262 mg $kg^{-1}$ of N, 114 mg $kg^{-1}$ of P, and 243 mg $kg^{-1}$ of K. On May 10, 1984, eight weeks after germination, seedlings were placed outdoors for three weeks to harden. Plants were watered every third day, unless it rained.

On May 30, 1984, 112 mycorrhizal and 112 non-mycorrhizal seedlings, whose heights were within the standard deviation of the mean of each population, were transplanted to the minesoil. At the time of transplanting seedling roots were examined to confirm the presence or absence of Pt infection. There were few root mantles on plants in Pt-infested containers, but at least 30% of the roots had Pt hyphae growing along them (long wall infection). Seedlings in noninfested containers showed no evidence of infection.

The minesoil selected for this study was 26 years old, had not been topsoiled, and had never been limed (pH 3.7), fertilized, or seeded. It was developing in Mahoning sandstone overburden replaced after surface mining of Upper Freeport coal in Preston County, West Virginia. Mahoning sandstone is typical of coal overburden in Appalachia. It is generally massive, but is often interspersed with thin-bedded shale and coal partings (Grube et al. 1971). The lower portion of the Mahoning is low chroma (gray) and often pyritic (up to 1.0% S), while the upper portion is highly weathered to a depth of about 6 m as evidenced by high chroma (red-brown), and is free of pyrite (Grube et al. 1972). Because no attempt was made to separate rock types when the overburden was replaced, it is a heterogenous mixture of acid and non acid-forming sandstone and shale. Although minesoils developing in this material are generally loamy, they contain from 35 to 70 percent coarse fragments on a volume basis. Unless these minesoils receive fertilizer, they quickly become infertile because of acidic weathering and leaching. They also have relatively high bulk densities, and low porosities. The present minesoil had a bulk density of 1.48 Mg $m^{-3}$.

The field experiment consisted of a factorial (2 X 2 X 2) of eight treatments - Pt inoculated and non-inoculated seedlings with and without "starter" amounts of N and/or P in all combinations which were replicated four times. Plots, each 1 m x 7 m, were arranged in an eight x four plot layout and were separated by 1 m buffer strips.

Seven bigtooth aspen seedlings, including the PV rootball, were planted in each plot at 1 m intervals. Because of the root bound-condition of seedlings at the time of transplanting, root balls were loosened which promotes root branching and growth (McLean 1976). Each seedling received 500 ml of water at planting.

Two weeks before transplanting, "starter" amounts of N and P were broadcast on the plots selected for fertilization. Nitrogen, as ammonium nitrate, was added at the rate of 76 kg $ha^{-1}$. Rock phosphate is moderately soluble in acidic media; thus it was expected to serve as a slow release form of P in the acidic minesoil. Slow release of P was desirable because it would prevent rapid fixation of applied P by the high levels of soluble Al and Fe in this minesoil (4.3 cmol(+) $kg^{-1}$ of KCl-extractable Al and 90.2 mg $kg^{-1}$ of DTPA-extractable Fe). To prevent deer browsing, a dome-shaped structure of 5 cm hex-mesh wire fencing was placed over each plot.

In August 1984, May 1985, and August 1985, one randomly selected seedling plus a 15 cm diameter and 15 cm deep portion of its root system was collected from each plot. The roots were separated from the stems and adhering soil was removed from the roots by gentle and repeated emersions in water. Tops were rinsed with distilled water. Both roots and tops were oven-dried at 68° C for four days. Data on percentage survival and top and root dry weight were analyzed statistically using the Statistical Analytical System (SAS) and Duncan's New Multiple Range Test (Snedecor and Cochran 1967).

## Results and Discussion

Transplanting was followed immediately by a hot, dry period. The US Weather Bureau Station at Brandonville, West Virginia (14 km from the study stie), reported that rainfall for June 1984 was 3.63 cm below the normal of 5.97 cm. Eight days after transplanting, 47 percent of the seedlings were severely defoliated. However, with the addition of 500 ml of water per seedling on two occasions during the dry period and with above-normal rainfall (+5.77 cm) occuring during July and August of 1984, most of these seedlings produced new leaves. In August 1984, an average of 76 percent of the seedlings were still surviving (Table 1). Although at this date percentage survival did not appear to be related to treatment, the lowest rate of survival occurred in the Pt inoculated-N fertilizer treatment. This same difference was observed in May 1985. The reduction in survival associated with the N-fertilized, Pt-inoculated trees may have been caused by increased plant succulency induced by Pt-enhanced N uptake. This may have caused seedlings to be more vulnerable to heat and moisture stress, diminishing their ability to survive. Beckjord (1978) and Beckjord et al. (1984) found that greenhouse-grown, N-fertilized, Pt-inoculated red oak seedlings suffered greater heat damage than seedlings receiving no N.

Seventy-one percent of all seedlings still survived one year after transplanting, and no further losses occurred between May and August of 1985 (Table 1). Additional observations in September 1987 (not tabulated) indicated that 67 percent of the seedlings alive in August 1985 had survived and several trees were greater than 150 cm in height. This was a high rate of survival compared to results of other studies involving tree seedlings planted on abandoned minesoils (Medve 1973; Marx and Artman 1979). The high survival rate is of considerable interest since this site had a low pH, high Al and Fe contents, and high bulk density.

In May 1985, one year after transplanting, wide differences in growth as expressed as root and top weight were present among treatments (Table 2). However, few of the differences were significant. The minesoil studied varied considerably over short distances with regard to relative proportions of soil material and coarse fragments. Consequently, plant growth within treatments was quite variable which increased the differences required for significance. Since top dry weight in most cases was related to root dry weight, discussion will only consider the relationship between root growth and treatment.

Rather than improve root growth, inoculation with Pt appeared to have reduced root growth in August 1984 and in May 1985 when compared with the non-inoculated, unfertilized control. This apparent growth reduction seemed to have been counteracted by the application of both N and P, which statistically increased seedling growth. Application of both N and P also increased the growth of non-Pt seedlings. It may be inferred that during the first year of growth, a mycorrhiza had not become well established and that the fungus competed with its host plant for nutrients, especially N and P. Growth of Pt seedlings that had received N and P in combination was increased about the same degree as Pt seedlings receiving N and P individually. In contrast, growth of non-Pt seedlings that had been fertilized with N and P in combination was the greatest among treatments, in both August 1984 and in May 1985. These latter results would, again, suggest a competition for nutrients between the fungus and the host plant.

Table 1. Percentage survival of bigtooth aspen seedlings as influenced by Pisolithus tinctorius, N, and P at three dates after transplanting (May 30, 1984).

| Treatment[1] | Harvest Date | | |
|---|---|---|---|
| | 8/84 | 5/85 | 8/85 |
| | % survival[2] | | |
| I | 71.2 | 84.2 | 100.0 |
| U | 89.3 | 91.3 | 100.0 |
| PI | 82.3 | 81.2 | 100.0 |
| PU | 89.3 | 95.0 | 100.0 |
| NI | 50.0 | 71.4 | 100.0 |
| NU | 75.0 | 80.0 | 100.0 |
| NPI | 71.5 | 88.9 | 100.0 |
| NPU | 81.3 | 86.4 | 100.0 |
| Overall[3] | 171/224=76% | 137/192=71% | 105/160=66% |

[1]I = inoculated (Pt), U = uninoculated, P = phosphorus, N = nitrogen.

[2]Percentage survival for 5/85 and 8/85 based on number of seedlings remaining in each treatment after 8/84 and 5/85 harvests respectively. One seedling was removed from each of 4 replicates in each treatment at each harvest.

[3]Percentage overall survival for 5/85 and 8/85 based on number of seedlings remaining in all treatments after 8/84 and 5/85 harvests respectively. A total of 32 seedlings were removed from the experiment at each harvest (1 seedling/each of 4 replicates X 8 treatments = 32).

Based on the one-year growth data it may be concluded that inoculation with Pt is ineffective in promoting growth of bigtooth aspen on abandoned minesoils such as the one studied, and that "starter" amounts of both N and P are needed to enhance growth of uninoculated Pt-bigtooth aspen. It also may be inferred from the one-year data that minesoil N and P were growth-limiting factors and that more emphasis should be placed upon these nutrients in forestation programs designed for old, acidic, abandoned minesoils, since most of them are commonly known to be low in N and P. However, these conclusions may be modified after further growth.

In August 1985, 15 months after transplanting, root growth (and top growth) had increased several-fold in every treatment when compared to the May 1985 growth (Table 2). However, there were wide variations among treatments. The increases in root growth can be attributed to the normal climatic conditions that prevailed during the period beginning in the early spring and extending through August, and to some carry-over effect of the N and P added in May 1984. The one hundred percent survival rate that occurred over the May-August period (Table 1) indicates that favorable growing conditions existed.

The apparent adverse effect of Pt on root growth noted in the May 1985 data seemed to have diminshed by August 1985, at which time the Pt seedlings without N or P seemed to have produced root growth greater than the control. Root growth of Pt seedlings that had been treated with both N and P were little different than the control, which again, supports the inference made earlier that the fungus competed with the plant for N and P.

## Conclusions

The results of this study, although not substantiated by statistical evidence and long-term observations, imply that N and P fertilization is necessary for encouraging good growth of bigtooth aspen seedlings transplanted to minesoils similar to the one reported

Table 2. Effect of Pt inoculation, N, P, and harvest date upon bigtooth aspen root and top growth.

| Treatment[1] Harvest Date | | Root Weight[2] g | Top Weight[2] g |
|---|---|---|---|
| 8/84 | I | 0.91 a[3] | 0.65 a |
| | U | 1.10 a | 0.63 a |
| | PI | 1.19 a | 0.75 a |
| | PU | 1.68 a | 0.88 a |
| | NI | 2.20 a | 3.20 a |
| | NU | 1.47 a | 1.25 a |
| | NPI | 1.47 a | 1.25 a |
| | NPU | 2.52 a | 3.24 a |
| 5/85 | I | 0.90 b | 1.79 b |
| | U | 1.49 b | 2.34 ab |
| | PI | 1.28 b | 2.27 b |
| | PU | 2.38 ab | 4.68 ab |
| | NI | 2.02 ab | 2.57 ab |
| | NU | 4.15 ab | 6.11 ab |
| | NPI | 2.23 ab | 3.51 ab |
| | NPU | 5.25 a | 7.52 a |
| 8/85 | I | 9.63 ab | 15.23 ab |
| | U | 4.87 b | 7.75 b |
| | PI | 13.13 ab | 22.18 ab |
| | PU | 11.78 ab | 16.67 ab |
| | NI | 18.38 ab | 27.63 a |
| | NU | 18.42 ab | 23.65 ab |
| | NPI | 6.40 ab | 11.57 ab |
| | NPU | 24.75 a | 27.00 a |

[1]I = inoculated (Pt), U = uninoculated, P = phosphorus, N = nitrogen.

[2]Data are averages of 4 trees, one was randomly selected from each of the four replicate plots.

[3]Data in each column for each harvest date followed by the same letter are not significantly different at p = 0.05 (Duncan's New Multiple Range Test).

here. Unfortunately, the use of N and P on abandoned minesoils has economic restrictions. Application of "starter" amounts of these two nutrients may fall within economically acceptable limits, but periodic applications thereafter do not unless economic returns are expected. Also, single applications of N and P may not suffice for long periods because of leaching losses of N and of eventual fixation of P by the high concentrations of Al and Fe often occurring in acidic minesoils.

With time it is possible that fungal mycelia will develop to a degree sufficient to offset low N and P levels by supplementing the roots in absorbing these nutrients from the minesoil. Data did not show any evidence in 15 months that inoculation of aspen with Pt was of any benefit over non-inoculated plants.

## References


Austin, R. C. and R. F. Strand. 1960. The use of slowly soluble fertilizers in forest planting in the Pacific Northwest. J. For. 58:619-627.

Beckjord, P. R. 1978. The incidence of ectomycorrhizae by Pisolithus tinctorius on Quercus rubra seedlings fertilized with sodium nitrate and ammonium chloride. Ph.D. Dissertation. Virginia Polytechnic Institute and State University, Blacksburg, Virginia.

Beckjord, P. R. and M. S. McIntosh. 1983. Growth and fungal retention by field planted Quercus rubra seedlings inoculated with several ectomycorrhizal fungi. Bull. Torry Bot. Club. 110(3):355-359.

Beckjord, P. R. and M. S. McIntosh. 1984. Growth and fungal persistence by Quercus rubra inoculated with ectomycorrhizal fungi and planted on a clear cutting and strip mine. Can. J. Bot. 62(8):1571-1574.

Beckjord, P. R., D. W. Smith, and M. S.

McIntosh. 1984. Effects of nitrogen fertilizer and Pisolithus tinctorius on Quercus rubra seedling root and top development. For. Sci. 30:124-128.

Bramble, W. C. 1952. Reforestation of strip-mined bituminous coal land in Pennsylvania. J. For. 50:308-314.

Bramble, W. C. and R. H. Ashley. 1955. Natural revegetation of spoil banks in Central Pennsylvania. Ecology 36:417-423.

DeMuro, G. 1986. The effect of Pisolithus tinctorius on the growth of bigtooth aspen on abandoned minesoil. M.S. Thesis, West Virginia University, Morgantown, West Virginia.

Grube, W. E., Jr., E. M. Jencks, R. N. Singh, R. M. Smith, and H. A. Wilson. 1971. Mine spoil potentials for water quality and controlled erosion. E.P.A. Proj. 10410 EJE. Environmental Protection Agency, Cincinnati, Ohio.

Grube, W. E., Jr., R. M. Smith, E. M. Jencks, and R. N. Singh. 1972. Significance of weathering in a Pennsylvania sandstone to pollution from strip mines. Nature (London) 236:70-71.

Iskra, A. and D. F. Hindal. 1985. Ectomycorrhizal associations of Populus grandidentata colonizing coal mine refuse. Phytopath. 75:625 (Abstract).

Limstrom, G. A. 1948. Extent, character, and forestation possibilities of land stripped for coal in the Central States. USDA Forest Service Tech. Paper. CS-109, USDA. Central States Forest Exp. Stn., Columbus, Ohio.

McLean, E. O. 1976. Chemistry of soil aluminum. Comm. Soil Sci. Plant Anal. 7:619-636.

Marx, G. D. 1977. Tree host range and world distribution of the ectomycorrhizal fungus Pisolithus tinctorius. Can. J. Microbiol. 23:217-223.

Marx, G. D. 1980. Role of mycorrhizae in forestation of surface mines. p. 109-116. In Proc. Trees for reclamation. Interstate Mining Compact Comm., USDA Forest Service. (Lexington, Kentucky, Oct. 27-29, 1980).

Marx, G. D. and J. D. Artman. 1979. Pisolithus tinctorius ectomycorrhizae improve survival and growth of pine seedlings on acid coal spoils in Kentucky and Virginia. Reclamation Rev. 2:23-31.

Mays, D. A. and G. W. Bengston. 1978. Lime and fertilizer use in land reclamation in humid regions. p. 308-328. In F. W. Schaller and P. Sutton (ed.). Reclamation of Drastically Disturbed Lands. Amer. Soc. Agronomy, Crop Sci. Soc. Am., Madison, Wisconsin.

Medve, R. J. 1973. Tree seedling survival on reclaimed bituminous strip mine spoils in Moraine State Park, Pennsylvania. Proc. Pa. Acad. Sci. 47:129-132.

Medve, R. J. and G. Shan. 1981. Distribution and plant associates of Pisolithus tinctorius on disturbed area in Western Pennsylvania. p. 72. In Abstracts of the Fifth North American Conference on Mycorrhizae.

Schramm, J. R. 1966. Plant colonization studies on wastes from the anthracite mining region in Pennsylvania. Trans. Amer. Philos. Soc. 56. 194 p.

Schultz, R. C., P. P. Kormanik, and W. C. Bryan. 1981. Effects of fertilization and vesicular arbuscular inoculation on growth of hardwood seedlings. Soil Sci. Soc. Am. J. 45:961-965.

Snedecor, G. W. and W. G. Cochran. 1967. Statistical Methods. Iowa State Univ. Press. Ames, IA.

# LONG-TERM EFFECTS OF A SINGLE APPLICATION OF MUNICIPAL SLUDGE ON ABANDONED MINE LAND[1]

BY

William E. Sopper and Eileen M. Seaker[2]

Abstract. In 1977, digested and dewatered municipal sludge was applied and incorporated in spoil material at a rate of 184 Mg/ha on a 0.4 ha experimental plot on an abandoned strip mine site in Pennsylvania. Data were collected for a five-year period (1977-1981) to determine the effects of the sludge application on the quality and growth of the herbaceous vegetation, the chemical properties of the soil, and the chemical quality of groundwater. In 1989, 12 years after sludge application, the site was again resampled to determine the long-term residual effects of the sludge application. Results of the re-evaluation indicated that the single high application of sludge facilitated the rapid development of a vegetative cover which has persisted over the 12 years with no apparent adverse effects on vegetation, soil, or groundwater.

Additional key words: Reclamation, trace metals, sludge utilization, revegetation, groundwater quality.

## Introduction

It is estimated that more than 7.7 million dry metric tons of municipal sludge are currently produced each year by the 15,300 public-owned treatment works in the United States. Approximately 25% of this is being land-applied for its fertilizer and organic matter value (Federal Register 1989). One of the most efficient uses for sludge is the reclamation of disturbed lands, such as those abandoned after coal mining which are acidic, droughty, and devoid of organic matter. Sludge has been shown to improve spoil structure, water holding capacity, and bulk density in addition to adding N, P, K, and other plant nutrients (Sopper et al. 1982; Sopper and Seaker 1983).

Approximately 121,000 hectares of land in Pennsylvania, strip mined prior to the federal Surface Mining Control and Reclamation Act of 1977, were abandoned after the coal was removed, leaving vast areas of barren spoil (USDA 1980). These sites have remained barren for years due to the difficulty of establishing and maintaining vegetation on the highly acidic material.

In 1977, a project was initiated in Pennsylvania that introduced the concept of using municipal sludge for revegetation of mined land to the general public in order to gain public acceptance and support. The specific objective of the project was to demonstrate that municipal sludge could be used to reclaim strip mined land and return it to potential agricultural use or to a wildlife habitat in an environmentally acceptable manner, without adverse effects on the quality of the vegetation, soil, or water. Vegetation, soil, and groundwater samples were collected over a five-year period (1977-1981) and results of these studies have been reported by Seaker and Sopper (1984).

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990. The research described in this article has been partially funded by the U.S. Environmental Protection Agency through Grant No. S-804511-020 and CR807408010.

[2]William E. Sopper is Professor of Forest Hydrology, School of Forest Resources, The Pennsylvania State University, University Park, PA 16802 and Eileen M. Seaker is Environmental Consultant, 1917 E. Branch Road, State College, PA 16801.

The issue of the long-term effect of applying single, large amounts of sludge in order to revegetate mine land often arises. What happens after all the sludge has been mineralized and all the nutrients and trace metals have been released to the soil and are potentially available for plant uptake and leaching? Will the vegetative cover persist or deteriorate?

One of the sites used in the 1977 project was an abandoned strip mine bank located in Venango County that had been backfilled and recontoured after mining without top soil replacement. Several revegetation attempts were unsuccessful. Dewatered digested sludge was applied in May 1977 to a 0.2-ha plot. In August, 1989, 12 years after sludge application, the site was revisited and samples of vegetation, soils, and groundwater were collected to evaluate the long-term effects. The project was originally designed as a demonstration to the public, rather than an experiment. Subsampling was employed, but statistical analyses could not be performed on the data. Instead, general trends are discussed.

## Materials and Methods

### Sludge Application

The surface soil was compacted, stony, and extremely acid (pH 3.8). The 0.2 ha plot was scarified with a chisel plow to loosen the surface spoil material and agricultural lime was applied at 12.3 Mg/ha to raise the spoil pH to 7.0. Sludge for the project was obtained from three local wastewater treatment plants. The sludge was applied at 184 Mg/ha with a manure spreader. The average concentrations of nutrients and trace metals and amounts applied in the sludge are given in Table 1. The amounts of nutrients applied were equivalent to applying an 11 (N) -9 ($P_2O_5$) -0 ($K_2O$) chemical fertilizer at 22,400 kg/ha.

Table 1. Chemical analysis of dewatered sludge applied and amounts of elements applied at 184 Mg/ha rate (Dwt Basis)

| Constituent | Average Concentration | Amount Applied |
|---|---|---|
| | mg/kg | kg/ha |
| Total P | 4624 | 918 |
| Total N | 12188 | 2388 |
| K | 93 | 18 |
| Ca | 9970 | 1834 |
| Mg | 2082 | 383 |
| Zn | 811 | 147 |
| Cu | 661 | 129 |
| Pb | 349 | 55 |
| Ni | 69 | 12 |
| Cd | 3.2 | 0.6 |
| pH | 7.9 | |

The amounts of trace metals applied are given in Table 2 along with the U.S. Environmental Protection Agency (EPA) and Pennsylvania Department of Environmental Resources (PDER) interim guideline recommendations (United States Environmental Protection Agency 1977; Pennsylvania Department of Environmental Resources 1977). It is quite obvious that the amounts of trace metals applied were well below the recommended lifetime limits except for copper, which slightly exceeded the Pennsylvania guidelines.

Immediately after sludge application and incorporation, the site was broadcast seeded with a mixture of two grasses (Kentucky-31 tall fescue, Festuca arundinacea Schreb., 22 kg/ha, Pennlate orchardgrass, Dactylis glomerata L., 22 kg/ha) and two legumes (Penngift crownvetch, Coronilla varia L., 11 kg/ha, and Empire birdsfoot trefoil, Lotus corniculatus L., 11 kg/ha). Then the site was mulched with straw and hay at the rate of 3.8 Mg/ha.

### Sampling and Analyses

A complete monitoring system was installed on the plot to evaluate the effects of the sludge applications on water quality, vegetation, and soil. Two groundwater wells were drilled (up-gradient and down-gradient) to sample the effects of the sludge application on groundwater quality. After sludge application, groundwater samples were collected bi-weekly for the first two months and monthly thereafter. Samples were analyzed for pH, nitrate-N by ion-selective electrode (Ellis 1976), dissolved Cu, Zn, Cr, Pb, Co, Cd, and Ni by atomic absorption spectrophotometry (EPA Methods of Chemical Analysis 1974).

Minesoil samples were collected at the 0 to 15, and 15 to 30 cm depth, passed through a 2 mm sieve, and analyzed for pH, Kjeldahl-N, Bray-P, exchangeable K, Ca, and Mg by ammonium acetate extraction, and dilute hydrochloric acid extractable Cu, Zn, Cr, Pb, Cd, and Ni (Jackson, 1958). Exchangeable cation and extractable metal concentrations were determined by atomic absorption.

At the end of each growing season vegetation growth responses were determined by measurements of percentage areal cover, and dry matter production. No crops were harvested over the 12-year period. Individual samples of tall fescue, orchardgrass, crownvetch, and birdsfoot trefoil from each plot were collected for foliar analyses. Plant samples were analyzed for Kjeldahl-N; P, K, Ca, Mg, by plasma emission spectrometry (Baker et al. 1964), and Cu, Zn, Cr, Pb, Co, Cd, and Ni by atomic absorption (Jackson 1958), after dry ashing and digestion.

## Results and Discussion

### Vegetation

The site was completely vegetated by August 1977, three months after sludge application, which has persisted throughout the 12-year period. Average annual dry matter production for the first five years and in 1989 was as follows:

| Year | Yield |
|---|---|
| | Mg/ha |
| 1977 | 6.0 |
| 1978 | 9.3 |
| 1979 | 11.3 |
| 1980 | 31.2 |
| 1981 | 22.6 |
| 1989 | 15.5 |
| AHY | 4.0 |

Table 2. Trace metal loadings of the sludge application and lifetime loadings recommended by the EPA and PDER.

| Constituent | Sludge Application 184 Mg/ha | EPA[1] (CEC 5-15) | PDER |
|---|---|---|---|
| | kg/ha | | |
| Cu | 129 | 280 | 112 |
| Zn | 147 | 560 | 224 |
| Cr | 74 | NR[2] | 112 |
| Pb | 55 | 800 | 112 |
| Ni | 12 | 280 | 22 |
| Cd | 0.6 | 11 | 3 |
| Hg | 0.09 | NR[2] | 0.6 |

[1] Average CEC of site ranged from 11.6 to 15.2 meq/100g

[2] No recommendation given by EPA

Dry matter production increased during the first four years, leveling off in 1981. In 1989 it was slightly lower but still well above the average hay yield (AHY) for undisturbed farmland soils in the county. During the first two years the two grass species dominated the site, but by the third growing season, the two legume species predominated and persisted through the fifth year (1981). However, by 1989 the birdsfoot trefoil had almost disappeared and now the dominating vegetative cover consists mostly of crownvetch and orchardgrass.

For brevity, only the foliar analyses for crownvetch and orchardgrass will be discussed. Foliar concentrations of macronutrients are given in Table 3. Nutrients (N and P) were all generally higher in the sludge-grown plants. Potassium and Ca were higher in the sludge-grown orchardgrass than in control plants. Potassium and Ca were only slightly lower in the sludge-grown birdsfoot trefoil plants than in the control plants. Foliar Mg concentrations were similar in both sludge-grown and control plants. Nutrient levels in the sludge-grown plants in 1989 were about the same level as the first year when sludge was applied. There appears to be little depletion of nutrients from the site over the 12-year period. Birdsfoot trefoil data are given in Table 3 because no crownvetch plants were present on the control plot for comparison. Macronutrient concentrations in crownvetch on the sludge-amended plot are given in Table 4. Concentrations were quite similar to those of birdsfoot trefoil.

Foliar concentrations of Zn, Cu, Pb, Ni, and Cd in orchardgrass and crownvetch are shown in Figures 1 to 5. Concentrations of Zn (Fig. 1), and Ni (Fig. 4) tended to be higher in crownvetch than in orchardgrass; whereas, concentrations of Cu (Fig. 2) tended to be higher in orchardgrass. Concentration of Pb (Fig. 3) and Cd (Fig. 5) were variable and showed no distinct trends. In general, trace metal foliar concentrations tended to be highest the first year and then decrease over time. Except for Ni, foliar concentrations of trace metals in the sludge-grown orchardgrass plants were higher than in control plants. The 1989 values for Cu (Fig. 2) and Cd (Fig. 5) were quite similar to those of 1981. Foliar concentrations of Zn, Ni, and Pb showed a slight increase from 1981 to 1989. Although sludge application appeared to increase some trace metal concentrations in the foliage, these increases were minimal and well below the suggested tolerance levels for agronomic crops (Melsted 1973). No phytotoxicity symptoms were observed during the study. The suggested tolerance levels are not phytotoxic levels but suggest foliar concentration levels at which decreases in growth may be expected.

Spoil Chemical Status

Changes in spoil pH over time are shown in Table 5. Spoil pH tended to increase from 1977 to 1979 and declined thereafter. This may explain why some of the foliar trace metal concentrations showed an increase in 1989. The nutrient status of the spoil seemed to show a general increase in concentrations of Kjeldahl-N up to 1981 and up to 1984 for Bray-phosphorus, K and Ca (Table 6). The application of lime and sludge initially resulted in a decrease in the concentration of Mg; however, since 1978 there has been a steady increase. The 1989 values are lower but still quite adequate to support plant growth.

Concentrations of extractable trace metals in the 0 to 15 cm spoil depth are given in Table 7 and for the 15 to 30 cm spoil depth in Table 8. Concentrations of Cu, Zn, Cr, Pb, Cd, and Ni all show a steady increase for the first five years (1977-81). By this time, most of the sludge organic matter was probably mineralized and most of the trace metals released to the surface spoil. Results of spoil analyses in 1984 and 1989 showed a gradual decrease in concentrations of all trace metals. Although the sludge application seemed to increase the concentrations of extractable trace metals in the 0 to 15 cm spoil depth, these higher concentrations are still within the normal ranges for these elements in U. S. soils (Allaway 1968).

It appears that there is some leaching of trace metals through the spoil profile. Concentrations of trace metals in the 15 to 30 cm spoil depth show a general increasing trend from 1977 to 1989.

Groundwater Quality

Results of the analyses of groundwater well samples are given in Table 9. The values for Well

Table 3. Mean foliar concentrations of macronutrient elements in orchardgrass and birdsfoot trefoil collected from the control and sludge-amended plots.

| Sludge application | Year | Orchardgrass | | | | | Birdsfoot trefoil | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | N | P | K | Ca | Mg | N | P | K | Ca | Mg |
| Mg/ha | | ----------%---------- | | | | | ----------%---------- | | | | |
| 0 | 1977 | [1] | | | | | [1] | | | | |
| | 1978 | 0.92 | 0.18 | 1.51 | 0.36 | 0.23 | 1.03 | 0.24 | 1.93 | 0.61 | 0.26 |
| | 1979 | 1.17 | 0.22 | 2.41 | 0.30 | 0.22 | 2.59 | 0.14 | 1.74 | 1.82 | 0.40 |
| | 1980 | 1.11 | 0.24 | 1.86 | 0.32 | 0.20 | 2.11 | 0.17 | 1.92 | 1.02 | 0.23 |
| | 1981 | 1.22 | 0.18 | 1.62 | 0.68 | 0.22 | 3.32 | 0.17 | 1.89 | 1.52 | 0.29 |
| | 1989 | 1.67 | 0.17 | 1.82 | 0.42 | 0.30 | 2.31 | 0.17 | 1.71 | 0.92 | 0.22 |
| 184 | 1977 | 2.62 | 0.40 | 2.84 | 0.84 | 0.31 | 3.64 | 0.27 | 1.46 | 1.99 | 0.28 |
| | 1978 | 1.26 | 0.37 | 2.01 | 0.49 | 0.28 | 1.27 | 0.36 | 2.30 | 0.59 | 0.25 |
| | 1979 | 1.33 | 0.51 | 2.53 | 0.47 | 0.23 | 3.57 | 0.25 | 1.56 | 0.54 | 0.20 |
| | 1980 | 1.70 | 0.42 | 2.65 | 0.45 | 0.23 | 2.93 | 0.25 | 1.62 | 1.27 | 0.18 |
| | 1981 | 2.57 | 0.37 | 2.38 | 0.53 | 0.26 | 4.03 | 0.26 | 1.69 | 1.14 | 0.23 |
| | 1989 | 2.36 | 0.37 | 2.24 | 0.45 | 0.22 | 2.38 | 0.16 | 1.01 | 0.65 | 0.23 |

[1]No plants available for sampling

Table 4. Mean foliar concentrations of macronutrient elements in crownvetch on the sludge-amended plot.

| Year | Crownvetch | | | | |
|---|---|---|---|---|---|
| | N | P | K | Ca | Mg |
| | ----------%---------- | | | | |
| 1977 | 3.36 | 0.34 | 1.64 | 2.63 | 0.42 |
| 1978 | 2.35 | 0.37 | 3.14 | 0.96 | 0.45 |
| 1979 | 3.35 | 0.22 | 1.29 | 1.68 | 0.25 |
| 1980 | 3.00 | 0.27 | 1.89 | 1.72 | 0.29 |
| 1981 | 3.78 | 0.31 | 1.89 | 1.25 | 0.23 |
| 1989 | 2.62 | 0.21 | 2.20 | 0.91 | 0.26 |

Table 5. Changes in Spoil pH over the thirteen year period

| Depth | Soil pH | | | | | |
|---|---|---|---|---|---|---|
| | May 1977[1] | Sept 1977 | Nov 1978 | Oct 1979 | May 1981 | Aug 1989 |
| cm | | | | | | |
| 0-15 | 3.8 | 6.2 | 6.7 | 7.3 | 5.8 | 5.4 |
| 15-30 | 3.8 | 4.2 | 4.6 | 5.1 | 3.4 | 5.6 |

[1]Pre-treatment samples

1 (control) reflect quality of groundwater for the disturbed mine site. Well 2 reflects the effects of the sludge application on water quality. Depth to the water table was 4.9 m in Well 1 and 3.0 in Well 2 in 1989. During the first five years (1977 to 1981) the water table fluctuated between 4.4 to 5.3 m in Well 1 and between 2.5 and 3.4 m in Well 2. Results indicate that the sludge application did not appear to have any significant effect on groundwater concentrations of nitrate-N. Average monthly concentrations of $NO_3$-N were below 10 mg/l (maximum concentration for potable water) for all months sampled during the five-year period (1977-81). The highest monthly values were 3.0 mg/l for the control well and 2.4 mg/l for Well 2.

Application of lime and sludge and subsequent revegetation appears to have had a positive effect on groundwater pH (Table 9). Groundwater pH increased from 4.6 (1977) to 6.0 by 1981. Results of the 1989 sampling indicated a pH of 6.6. There has also been a gradual increase in pH in the

Table 6. Changes in concentrations of Kjeldahl-nitrogen, Bray-phosphorus and exchangeable cations in the spoil collected at the 0-15 cm depth

| Year | Kjeldahl Nitrogen | Bray Phosphorus | K | Ca | Mg |
|---|---|---|---|---|---|
| | % | | mg/kg | | |
| May 1977[1] | 0.04 | 2 | 12 | 541 | 452 |
| Sept 1977 | 0.05 | 11 | 19 | 1222 | 32 |
| 1978 | 0.09 | 9 | 23 | 2600 | 40 |
| 1979 | 0.16 | 38 | 46 | 3873 | 53 |
| 1981 | 0.34 | 79 | 45 | 1298 | 99 |
| 1984 | --- | 91 | 74 | 1440 | 108 |
| 1989 | 0.12 | 83 | 30 | 733 | 84 |

[1] Pre-sludge samples

Table 7. Changes in concentrations of extractable trace metals from spoil collected at the 0-15 cm depth following sludge application.

| Sampling Date | Cu | Zn | Cr[2] | Pb | Cd | Ni |
|---|---|---|---|---|---|---|
| | | | mg/kg | | | |
| May 1977[1] | 2.5 | 2.9 | 0.2 | 0.5 | 0.02 | 1.1 |
| Sept 1977 | 10.8 | 7.7 | 0.4 | 3.5 | 0.04 | 0.9 |
| 1978 | 8.8 | 7.7 | 0.2 | 2.3 | 0.02 | 1.2 |
| 1979 | 58.7 | 56.9 | 1.7 | 13.0 | 0.27 | 1.5 |
| 1981 | 87.3 | 74.6 | 3.5 | 22.7 | 0.95 | 2.8 |
| 1984 | 57.6 | 59.6 | --- | 14.8 | 0.56 | 2.8 |
| 1989 | 51.9 | 37.8 | --- | 13.5 | 0.42 | 2.0 |
| Normal Range for U.S. Soils | 2-100 | 10-300 | 5-3000 | 2-200 | 0.01-7.00 | 5-500 |

[1] May 1977 values represent pretreatment conditions
[2] Values for Cr are total concentrations.

control well from pH 4.4 to pH 5.8. Since 1980, attempts have been made to reclaim the control area by conventional methods using lime and fertilizer. The amounts of lime and fertilizer applied and frequency of application are not known as the coal company is no longer in business. However, these applications and vegetation growth probably contributed to the increase in groundwater pH in the control well.

There appears to be no significant increase in any of the trace metal concentrations over the initial five-year period (1977-1981) in the groundwater samples from Well 2 compared to the control well (Table 9). From 1977 to 1981 most of the monthly concentrations were within the U.S. Environmental Protection Agency drinking water standards. The only exception was Pb which exceeded the limit of 0.05 mg/l for both the control well and Well 2, probably resulting from solubilization upon weathering after mining. The highest monthly Pb values were 0.28 mg/l in the control well and 0.33 mg/l in Well 2 in 1978, and the mean annual Pb concentrations were 0.19 and 0.20 mg/l for control well and Well 2, respectively. By 1981, however, the mean annual Pb concentrations had decreased to 0.04 and 0.05 mg/l for the two wells. Results of analyses of the groundwater samples collected in 1989 had extremely low concentrations of all trace metals in both wells in comparison to values for the initial five years (1977-81).

## Conclusions

Re-evaluation of an abandoned strip mine spoil bank 12 years after being amended with 184 Mg/ha of

Table 8. Changes in concentrations of extractable trace metals from spoil collected at the 15-30 cm depth following sludge application.

| Sampling Date | Cu | Zn | Cr | Pb | Cd | Ni |
|---|---|---|---|---|---|---|
| | ---------- mg/kg ---------- | | | | | |
| May 1977[1] | 3.0 | 2.4 | 0.10 | 0.6 | 0.020 | 1.0 |
| Sept 1977 | 4.0 | 2.0 | 0.10 | 1.3 | 0.010 | 0.4 |
| 1978 | 2.5 | 1.7 | <0.01 | 1.3 | 0.007 | 0.7 |
| 1979 | 9.2 | 8.7 | 0.28 | 2.4 | 0.026 | 0.2 |
| 1981 | 2.4 | 2.8 | 0.05 | 0.5 | 0.014 | 0.4 |
| 1989 | 13.8 | 10.2 | 0.43 | 3.8 | 0.122 | 1.9 |

[1]May 1977 values represent pretreatment conditions
[2]Values for Cr are total concentrations

Table 9. Mean annual concentrations of nitrate - N and trace metals in groundwater

| Site | Year[1] | pH | $NO_3$-N | Cu | Zn | Cr | Pb | Cd | Ni |
|---|---|---|---|---|---|---|---|---|---|
| | | | ---------- mg/l ---------- | | | | | | |
| Well 1 | 1977 | 4.4 | 1.4 | 0.22 | 4.13 | 0.02 | 0.14 | 0.006 | 3.67 |
| (control) | 1978 | 4.3 | <0.5 | 0.23 | 2.02 | 0.01 | 0.19 | 0.002 | 0.98 |
| | 1979 | 4.6 | <0.5 | 0.17 | 1.48 | 0.03 | 0.13 | 0.001 | 0.50 |
| | 1980 | 5.5 | 0.6 | 0.05 | 0.89 | 0.05 | 0.09 | 0.001 | 0.50 |
| | 1981 | 5.7 | 0.7 | 0.06 | 0.83 | 0.03 | 0.04 | 0.003 | 0.31 |
| | 1989[2] | 5.8 | 0.02 | 0.01 | 0.09 | <0.001 | 0.01 | 0.001 | 0.06 |
| Well 2 | 1977 | 4.6 | 1.1 | 0.10 | 3.39 | 0.03 | 0.09 | 0.001 | 2.67 |
| (sludge) | 1978 | 4.5 | <0.5 | 0.14 | 3.29 | 0.01 | 0.20 | 0.002 | 1.26 |
| | 1979 | 4.4 | <0.5 | 0.18 | 1.49 | 0.03 | 0.13 | 0.001 | 0.97 |
| | 1980 | 5.7 | 0.6 | 0.05 | 1.05 | 0.04 | 0.11 | 0.001 | 0.76 |
| | 1981 | 6.0 | 0.6 | 0.05 | 0.57 | 0.02 | 0.05 | 0.001 | 0.31 |
| | 1989[2] | 6.6 | 0.06 | 0.01 | 0.07 | <0.001 | 0.01 | 0.001 | 0.04 |
| EPA drinking water standard | | | 10 | 1 | 5 | 0.05 | 0.05 | 0.010 | --- |

[1]Values are annual means of monthly samples.

[2]Average of three samples collected in August 1989.

municipal sludge indicates that a single large application of sludge can be used successfully to revegetate mine lands with no apparent adverse effects on vegetation, spoil, or groundwater quality.

Literature Cited


Allaway, W. H. 1968. Agronomic controls over the environmental cycling of trace metals. Adv. In Agron. 20:235-271.

Baker, D. E., G. W. Gorsline, C. B. Smith, W. I. Thomas, W. E. Grube, and R. L. Ragland. 1964. Technique of rapid analyses of corn leaves for eleven elements. Agron. J. 56:133-136.

Ellis, B. G. 1976. Analyses and their interpretation for wastewater application on agricultural land. North Central Regional Research Publication 235-Sec. 6.

Federal Register. 1989. Standards for the Disposal of Sewage Sludge. Proposed Rule Vol. 54(23), Feb. 6, 1989, Part II, 40CFR parts 257 and 503, p. 5746-5902.

Jackson, M. L. 1958. Soil Chemical Analysis. Prentice-Hall, Inc., Englewood Cliffs, NJ.

Joost, R. E., J. H. Jones, and F. J. Olsen. 1981. Physical and chemical properties of coal refuse as affected by deep incorporation of sewage sludge and/or limestone. p. 307-312. In: Proc. Symp. on Surface Mining Hydrol., Sedimentol., and Reclamation. (Univ. of Kentucky, Lexington, KY, 7-11 Dec. 1981).

Melsted, S. W. 1973. Soil-plant relationships. p. 121-128. In: Recycling Municipal Sludges and Effluents on Land, Nat. Assoc. of State Universities and Land Grant Colleges, Washington, DC.

Pennsylvania Department of Environmental Resources. 1977. Interim guidelines for sewage sludge use for land reclamation. In: The Rules and Regulations of the Department of Environmental Resources, Commonwealth of Pennsylvania, Chapt. 75, Subchapt. C, Sec. 75.32.

Seaker, E. M. and W. E Sopper. 1984. Relcamation of bituminous strip mine spoil banks with municipal sewage sludge. Reclam. Reveg. Res., 3:87-100.

Sopper, W. E., E. M. Seaker, and R. K. Bastian (Editors). 1982. Land Reclamation and Biomass Production with Municipal Wastewater and Sludge. The Pennsylvania State Unviersity Press, University Park, PA 16802.

Sopper, W. E. and E. M. Seaker. 1983. A guide for revegetation of mined land in eastern United States using municipal sludge. Institute for Research on Land and Water Resources, The Pennsylvania State University, Universit Park, PA., 93 pp.

United States Department of Agriculture 1980. Soil and Water Resource Conservation Act: Appraisal 80. USDA, Washington, D. C.

United States Environmental Protection Agency 1974. Methods for Chemical Analysis of Water and Wastes. Washington, D. C.

United States Environmental Protection Agency. 1977. Municipal Sludge Management: Environmental Factors. Tech. Bull. EPA 430/9-76-004, MCD-28.

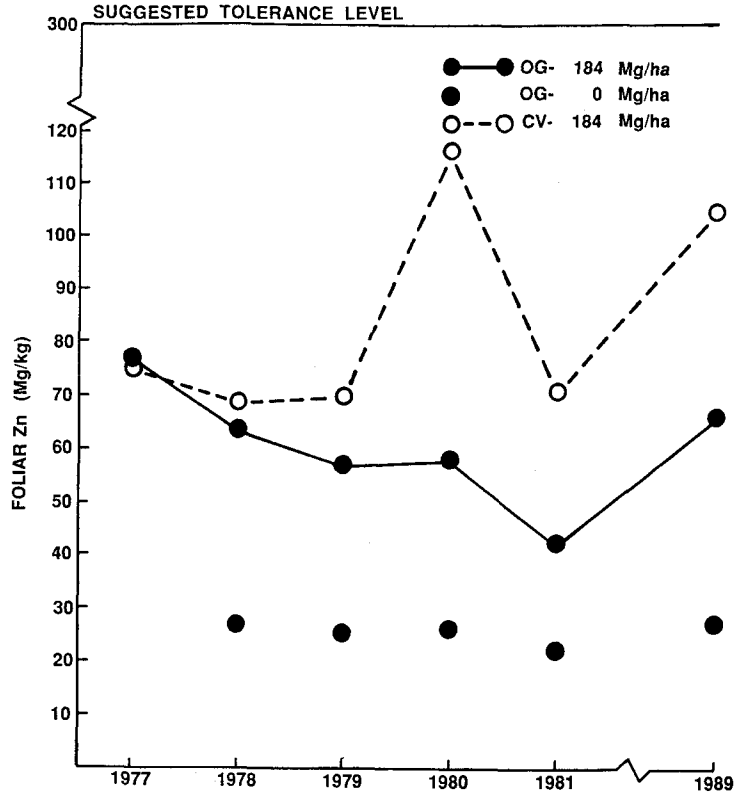



Figure 1. Mean foliar concentration of Zn in orchardgrass and crownvetch collected from the control and sludge-amended plots.

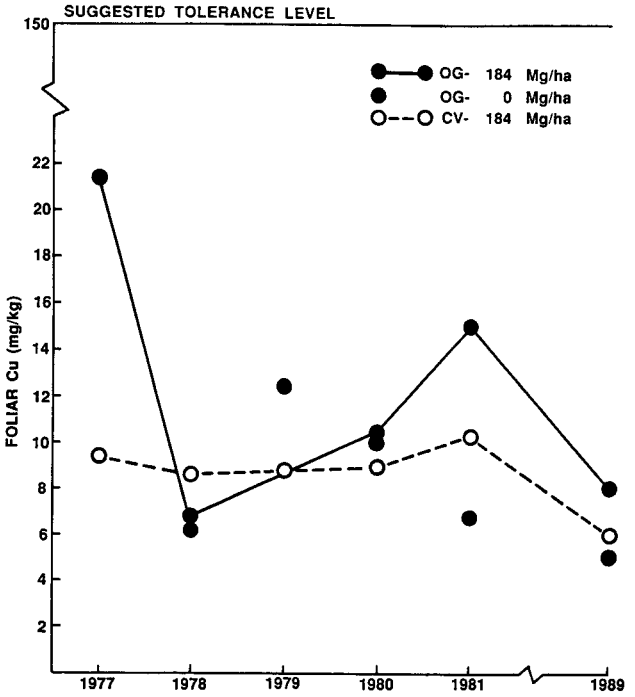



Figure 2. Mean foliar concentration of Cu in orchardgrass and crownvetch collected from the control and sludge-amended plots.

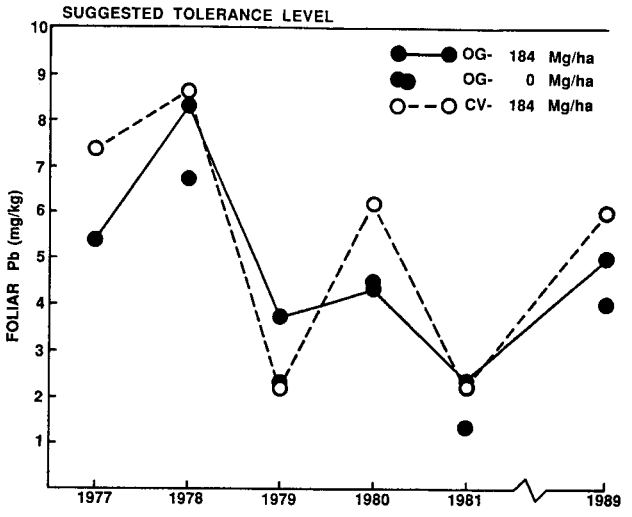



Figure 3. Mean foliar concentration of Pb in orchardgrass and crownvetch collected from the control and sludge-amended plots.

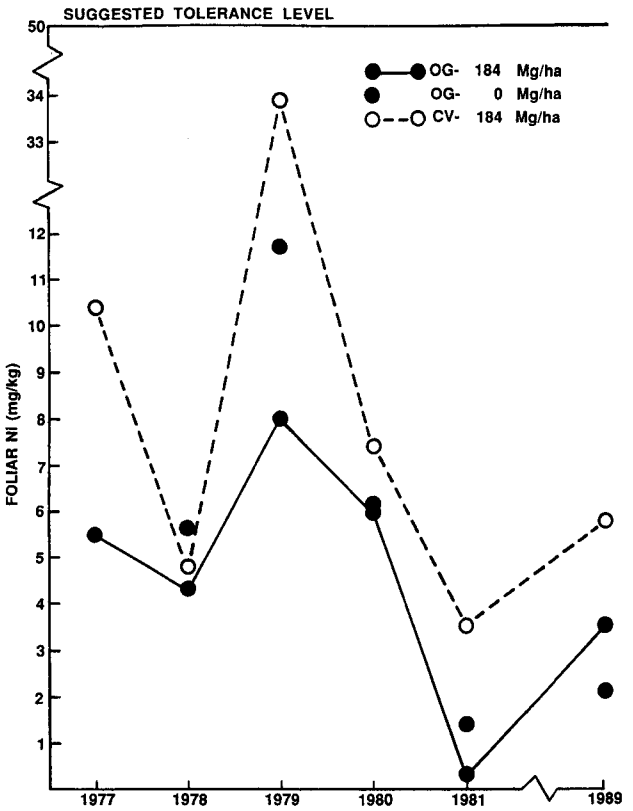



Figure 4. Mean foliar concentration of Ni in orchardgrass and crownvetch collected from the control and sludge-amended plots.

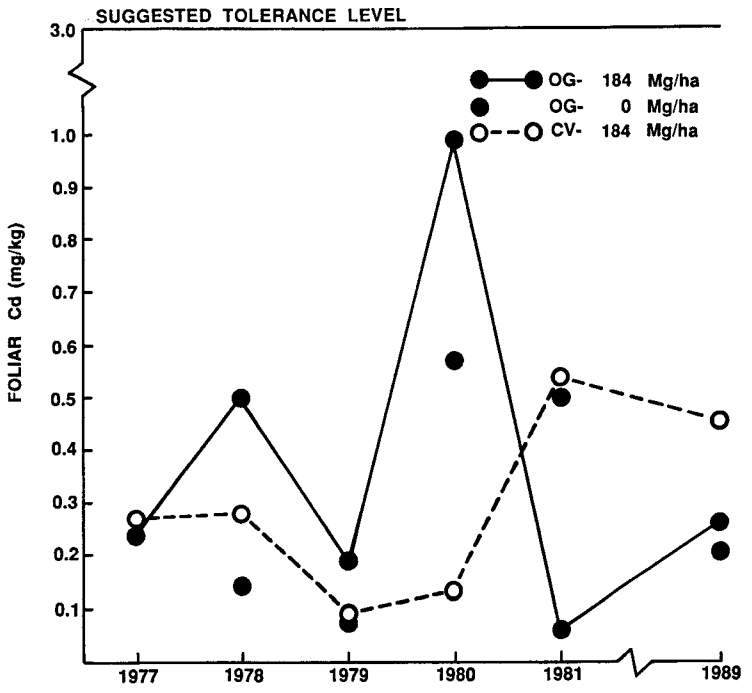



Figure 5. Mean foliar concentration of Cd in orchardgrass and crownvetch collected from the control and sludge-amended plots.

This page intentionally left blank.

# STIMULATION OF ASPEN ESTABLISHMENT ON UNRECLAIMED MINE SPOILS[1]

Robert S. Hedin and Elizabeth Ruch Hedin[2]

Abstract. The effect of surface amendments on the colonization of volunteer trees onto unvegetated coal spoils was tested. In the spring of 1989, barren acidic (pH 3.5) minesoils were treated with limestone (550 g $m^{-2}$), fertilizer (75 g $m^{-2}$ of 5-10-5), and either chipped aspen stems or chipped aspen roots (200 cc $m^{-2}$). The experimental design was a 2 X 2 X 3 factorial with 3 replications. Surrounding the study area were patchy, but vigorous stands of volunteer aspens (Populus tremuloides and P. grandidentata) which produced copious seed in late May. Counts of aspen seedlings in the experimental plots were done at two week intervals from June to September. Germination of aspen seeds was increased by all treatments. The highest numbers of seedlings on June 18 were observed in fertilizer/root chip plots (218 seedlings $m^{-2}$), limestone/root chip plots (140 seedlings $m^{-2}$), and limestone/fertilizer/stem chip plots (121 seedlings $m^{-2}$). Control plots had only 10 seedlings $m^{-2}$. High mortality of seedlings was observed in all plots with the onset of dry, hot weather in July. Survival of seedlings only occurred in plots that received fertilizer. Although hot and dry weather continued through August, only minor further mortality was observed. On September 9, the highest seedling densities occurred in plots that received limestone, fertilizer and either type of wood chip treatments (11 seedlings $m^{-2}$). Whether the chips were derived from roots or stems made no difference. Analyses of surface soil samples collected in September indicated that fertilizer and limestone ammendments still had ameliorative effects on soil chemistry. Survival of aspen seedlings in acid, infertile minesoils cannot yet be determined. Monitoring of plots is continuing.

## Introduction

Volunteer aspens (Populus grandidentata and P. tremuloides) are important determinants of long-term successional trends on unreclaimed acidic minesoils in northern Appalachia (Hedin 1988). On sites where large numbers of aspens colonized the spoils soon after site abandonment, a forested plant community developed within 25 to 35 years. Where low numbers of aspens colonized, a poorly vegetated, open-canopied plant community developed. After 20 to 30 years of abandonment, these sites are typically characterized by groves of aspens separated by large areas of spoil that are either bare or covered with lichens and mosses. Because colonization of these open areas by trees is slow, the spoils appear destined to remain poorly vegetated for many decades.

Poorly vegetated minesoils are typically attributed to a combination of extremely low pH and high concentrations of exchangeable aluminum (Berg and Vogel 1973, Bell and Unger 1982, Ciolkosz et al. 1985). This explanation, however, cannot explain the existence of thriving stands of aspens in soils that are chemically similar to the bare soils. On one poorly vegetated site, the pH of surface soils under aspens averaged 3.9, only 0.4 pH units higher than bare soils (Hedin 1987). Most established aspens on these sites are slowly

[1]Paper presented at the 1990 Mining and Reclamation Conference and Exhibition, West Virginia, April 23-26, 1990.

[2]Robert S. Hedin is a Research Biologist with the U.S. Bureau of Mines, Pittsburgh Research Center and Elizabeth R. Hedin is a horticulturalist with Horticultural Design, Pittsburgh, PA 15228.

invading bare spoils by root sprouting. It is unlikely that aspen roots would invade these areas if the soil chemistry was toxic to aspens.

These observations led us to hypothesize that acidic minesoils remain barren for decades because soil characteristics are toxic to colonizing seedlings, not to established plants. This colonization "bottleneck" prevents the establishment of trees that could otherwise thrive on the site. If the bottleneck hypothesis is correct, then temporary moderation of the chemical conditions of the minesoils should cause colonization by aspens that, once sufficiently established, will survive and grow long after the moderating conditions dissipate. In this paper, we report first year results of an experiment designed to test the bottleneck hypothesis by amending extremely acidic barren minesoils with treatments of limestone, fertilizer and wood chips.

## Study Site

The experiment is being conducted on strip mine spoils three miles west of Emlenton, Pennsylvania. Lower Allegheny coals (Clarion, Brookville) were mined in the early 1960's. Although the site was planted with pine, spruce, and larch tree seedlings before abandonment, mortality appears to have been very high. Currently, most of the spoils are either barren or sparsely vegetated with a mixture of planted trees and volunteer plants. Isolated volunteer trees, typically aspen, black birch (Betula lenta), red maple (Acer rubrum), or oak (Quercus alba, Q. imbricaria, and Q. velutina) are scattered across the site. Where several aspens colonized the same area, root sprouting has sometimes resulted in small groves of trees. The dominant herbaceous plants are the bunch grasses, broomsedge (Andropogon virginicus) and poverty grass (Danthonia spicata). Coverage of grasses is highest in drainage areas. Except for these grassy patches and within dense groves of aspens, the spoils are generally bare. Despite copious annual production of wind born seeds by aspen, broomsedge and poverty grass, successful colonization of the bare minesoils has been minimal during the last several decades.

## Materials and Methods

An experimental area was established on barren, gently sloping spoils in April 1989. A square grid was established that contained 49 plots, each 2m by 2m. On April 30, the plots were treated with surface applications of limestone, fertilizer, and wood chips. Rates of application are shown in Table 1. Limestone raised pH and provide calcium and magnesium. Fertilizer provided nutrients (N, P, K). Root chips were applied as a mycorrhizal inoculant, while stem chips were applied as non-mycorrhizal control. Chip treatments should not be considered a mulch. The application rates were so light (two sandwich bags of chips per 4 $m^2$ plot) that affects on soil moisture or soil temperature likely did not occur.

The wood chips were obtained by cutting branches and digging roots from aspen trees growing on the site and chipping them with a standard brush chipper. Care was taken to ensure that no contamination of the stem chips with roots occurred. Between preparation and application, chips were stored in black plastic bags at 20° C. During this 10-day period, the root chips became covered with fungal mycelia. No growth of mycelia was observed on stem chips.

Table 1. Amendment Applications Rates used in the Experiment

| Amendment | Rate |
|---|---|
| Limestone (86% $CaCO_3$ equiv) | 550 g $m^{-2}$ (5000 lb $A^{-2}$) |
| Fertilizer (5-10-5, granular) | 75 g $m^{-2}$ (680 lb $A^{-2}$) |
| Wood Chips | 500 cc $m^{-2}$ |
| Aspen roots | 500 cc $m^{-2}$ |
| Aspen branches | (2.7 $yd^3$ $A^{-2}$) |

The experimental design was a 2 X 2 X 3 factorial replicated three times for a total of 36 plots. Treatments combinations were assigned to the plots randomly. In addition to the three control plots included in the experimental design, nine additional control plots were established adjacent and up-gradient of the experimental area and are reported as an outside-plot control.

The number of tree seedlings in each of the plots was observed from May 28 through September 9, 1989 at two week intervals. In May and June, when seedlings were tiny and numerous, numbers were estimated by counting all seedlings within a 0.05 $m^2$ frame that was placed at four random locations within each plot. During the rest of the summer, counts of all seedlings were made. On August 5, August 20 and September 9, the height of each seedling was measured.

Seven surface soil samples were collected from the experimental area in April (before amendment application). Ten surface samples were collected from various experimental plots in September. Several samples were also collected from various locations on the site outside of the study area. Samples were analyzed by The Pennsylvania State University. Soil pH, P (Bray determination), acidity and exchangeable cations were determined by the Merkle Laboratory. Metals were determined by the Soil and Environmental Chemistry Laboratory using a DTPA extraction procedure (Baker and Amacher 1981).

## Results and Discussion

### Soil Analysis

Unamended surface soils in the experimental area were extremely acidic, with high concentrations of available aluminum and low concentrations of available phosphorous and base cations (Table 2). Acidity, pH and aluminum levels were in ranges typically considered toxic to plants (Ciolkosz et al. 1985). Two surface samples, collected from bare soils under an established bigtooth aspen and under recent root sprouts, had soil chemistries similar to bare soils (Table 3). Soil collected from the rooting zone in a dense aspen grove was less acidic, but P and Al levels were the same as samples from bare areas.

Table 2. Pre-amendment soil chemistry of experimental area (April 1989).

| Parameter | Unit | Mean[1] | Range |
|---|---|---|---|
| pH | p(activity) | 3.7 | 3.3 - 3.9 |
| P | lb $A^{-1}$ | 13 | 4 - 21 |
| Acidity | meq 100 $g^{-1}$ | 27.3 | 17.5 - 29.7 |
| CEC | meq 100 $g^{-1}$ | 16.1 | 15.5 - 16.4 |
| Potassium | meq 100 $g^{-1}$ | 0.13 | 0.11 - 0.15 |
| Magnesium | meq 100 $g^{-1}$ | 0.3 | 0.1 - 0.5 |
| Calcium | meq 100 $g^{-1}$ | 0.7 | 0.3 - 0.8 |
| Aluminum | meq 100 $g^{-1}$ | 0.6 | 0.5 - 0.7 |

[1]Mean of seven samples.

Table 3. Soil chemistry in bare areas and in vegetated areas.

| | pH | P | Acid | CEC | K | Mg | Ca | Al[1] |
|---|---|---|---|---|---|---|---|---|
| Experimental area[2] | 3.7 | 13 | 27.3 | 16.1 | 0.1 | 0.3 | 0.7 | 0.6 |
| Bare under adult aspen | 3.6 | 9 | 23.6 | 16.9 | 1.3 | 3.8 | 5.9 | NA[3] |
| Bare under aspen sprout | 4.0 | 21 | 27.3 | 16.0 | 0.6 | 2.2 | 3.1 | NA |
| Root zone soil in aspen grove | 4.1 | 27 | 16.3 | 16.5 | 0.3 | 0.5 | 0.8 | 0.5 |

[1]See Table 3 for units.
[2]Mean of seven determinations (from Table 2).
[3]Not available.

Table 4. Soil chemistry in experimental plots in September 1989.[1]

| | pH | P | Acid | CEC | K | Mg | Ca | Al[1] |
|---|---|---|---|---|---|---|---|---|
| Control | 3.6 | 11 | 16.8 | 16.3 | 0.9 | 0.7 | 0.8 | 0.6 |
| Limestone | 6.2 | 17 | 2.4 | 12.9 | 0.9 | 4.7 | 75.6 | NA |
| Lime/Fert | 7.1 | 78 | 0.0 | 13.9 | 0.20 | 0.7 | 13.3 | <0.1 |
| Lime/Fert/Stems | 5.9 | 81 | 4.9 | 14.4 | 0.19 | 0.6 | 8.7 | <0.1 |
| Lime/Fert/Roots | 6.8 | 80 | 1.0 | 13.1 | 0.20 | 0.7 | 13.4 | <0.1 |

[1]All values the average of two samples, see Table 3 for units.

All soil samples collected in September still showed the effects of spring amendment applications (Table 4). Limestone plots had circumneutral pH, low acidity and high levels of exchangeable calcium. Fertilizer plots still had circumneutral pH and moderate phosphate levels. Control plots had low pH and high aluminum levels typical of unamended soils.

Aspen Colonization, Survival, and Growth

Aspen trees produced seed in late May. The viability of these seeds was tested by collecting several dozen seeds from trees surrounding the plots and subjecting them to ideal growing conditions in a greenhouse. Within a week, 90% had germinated.

On May 28, numerous aspen seeds were observed in all experimental plots, but no germination was evident. During the next week, the study site experienced cool and rainy weather that likely produced ideal weather conditions for seed germination.

By June 6, seed production by aspens had ceased and numerous aspen seedlings were observed in many experimental plots (Table 5). Because many aspen seeds blew into the plot clumped together or still attached to catkins, the seedlings were often found in discrete groups within plots. This variation made it impossible to distinguish significant differences between treatments.

On June 18, similar numbers of seedlings were observed in the experimental plots. None of the treatment averages differed significantly between June 6 and 18 (tested by individual ANOVA tests for each treatment). Thus, germination of aspen seeds was largely complete by this date. No mortality of seedlings was evident on this date.

Higher germination of aspens was observed in control plots within the study area than controls adjacent to the study area (control-in and control-out, respectively, in Table 5). This difference likely resulted from some movement of chemical amendments on treated plots to untreated plots during rainstorms in May and June. Two plots contained all of the seedlings observed in the

Table 5. Number of Populus (Aspen) seedings observed in experimental plots per $m^2$.

| Treatment | Date, Weather condition 5/28 rain | 6/6* rain | 6/18* rain | 7/16* dry | 8/5# dry | 8/20# dry | 9/9# dry |
|---|---|---|---|---|---|---|---|
| Control-out[1] | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Control-in | 0 | 2 a[2] | 10 a | 2 a | 0 | 0 | 0 |
| Limestone | 0 | 28 a | 67 a | 10 a | 0 | 0 | 0 |
| Fertilizer | 0 | 81 ab | 54 a | 14 a | 2 a[3] | 2 | 1 a |
| Roots | 0 | 37 a | 39 a | 2 a | 0 | 0 | 0 |
| Stems | 0 | 35 a | 10 a | 0 a | 0 | 0 | 0 |
| Lime/Fert | 0 | 101 ab | 85 a | 33 ab | 4 a | 4 a | 4 a |
| Lime/Root | 0 | 112 ab | 140 ab | 66 c | 0 | 0 | 0 |
| Lime/Stem | 0 | 46 a | 15 a | 2 a | 0 | 0 | 0 |
| Fert/Root | 0 | 160 b | 218 b | 3 a | 1 a | 1 a | 1 a |
| Fert/Stem | 0 | 63 ab | 51 a | 10 a | 1 a | 1 a | 1 a |
| Lime/Fert/Root | 0 | 63 ab | 102 a | 31 a | 11 b | 10 ab | 10 ab |
| Lime/Fert/Stem | 0 | 79 ab | 121 ab | 54 bc | 14 b | 13 b | 12 b |

*Estimates based on counts of seedlings in four 0.05 $m^2$ frames in each plot.
#Actual count of all seedlings.
[1]Average of eight plots, adjacent and up-gradient of experimental area that did not receive any amendments.
[2]Means with same letter within the same column are not significantly different at 0.05 significance level (Student-Newman-Keuls test).
[3]Only plots containing aspen seedlings were included in statistical comparison of means on 8/5, 8/20, and 9/9.

control in plots and they were both immediately down-slope of plots that received fertilizer and limestone amendments.

The site was visited on June 30, but counts of seedlings were not made. Close inspection of the plots, however, did not indicate stressful conditions or mortality of seedlings.

In July, weather at the study site turned hot and dry. During the month, widespread mortality of seedlings occurred. By August 5, complete mortality had occurred in control, limestone, and wood chip plots. Seedlings in the limestone/root chip plots, which had some of highest densities during the first six weeks of the experiment, experienced complete mortality. Two patterns of mortality were observed. In plots not receiving fertilizer, seedlings grew very little after germination. During July, these tiny seedlings turned red and gradually died, probably from stress associated with phosperous deficiency (Salisbury and Ross 1978). This pattern contrasted sharply with several of the fertilizer plots, where seedlings grew tall and spindly during June. By mid-July, many of these seedlings had wilted and died.

Although mortality in fertilizer plots was high, survival of some seedlings was observed in all treatments. In particular, significant survival was observed in plots that received both limestone and fertilizer treatments.

Throughout August, hot and dry weather continued. However, further mortality of seedlings was limited (Table 5, see also total counts in Table 6). Most of the plants that had survived grew 3-5 cm in August and developed several more sets of leaves. Seedlings that did die were primarily those that were very small. The primary source of mortality for larger seedlings (>5 cm height) appeared to be physical damage caused by human or deer movement through the experimental plots.

By the end of summer, seedlings had grown to an average height of 8.0 cm (Table 6). Several seedlings grew to 20-25 cm. The tallest plants were in plots receiving limestone, fertilizer, and either wood chip amendment.

Table 6. Mean height of aspens in treatments containing seedlings.

| Treatment | Date 8/5 #[1] | 8/5 $cm^2$[2] | 8/20 # | 8/20 cm | 9/9 # | 9/9 cm |
|---|---|---|---|---|---|---|
| Fertilizer | 20 | 5.3 | 18 | 6.1 | 18 | 6.4 |
| Lime/Fert | 43 | 6.6 | 46 | 6.4 | 45 | 6.9 |
| Fert/Root | 12 | 4.1 | 12 | 4.1 | 6 | 4.8 |
| Fert/Stem | 9 | 3.3 | 7 | 3.8 | 7 | 3.8 |
| L/F/Root[3] | 130 | 7.9 | 118 | 8.1 | 115 | 8.4 |
| L/F/Stem[4] | 166 | 6.6 | 155 | 7.4 | 142 | 8.6 |

[1]Total number of seedlings in three plots.
[2]Average height of seedlings in cm.
[3]Limestone/fertilizer/root.
[4]Limestone/fertilizer/stem.

## Summary

The first year results of this experiment indicate that the absence of aspen trees on poorly vegetated acidic spoils is caused by a colonization bottleneck that can be removed through the application of one or more surface amendments. The bottleneck has two aspects: 1) germination of aspen seeds is inhibited by extreme chemical conditions, and 2) extreme physical conditions in summer months cause high seedling mortality. The

most likely chemical inhibition of germination is low pH. Although no specific tests with aspens have been conducted, numerous experiments with other plants have shown that soil pH values less than four greatly reduce germination percentages (Foy et al. 1978). By raising the pH of minesoils with either fertilizer or limestone applications, mass germination of aspens was stimulated. The slight germination of seeds that occurred in experimental plots that did not receive direct applications of limestone or fertilizer, but were likely affected by alkaline runoff from adjacent plots, illustrates the minor chemical change necessary to stimulate aspen germination. It also illustrates the need for buffer strips between treatment plots in future experiments.

Once established, aspen seedlings are subjected to temperature and moisture stresses that can cause high mortality (Deely and Borden 1972). The minimal energetic reserves contained in tiny aspen seeds prevent rapid development of an extensive root system and make them particularly vulnerable to these stresses. Bare minesoils are subject to rapid and extreme changes in temperature and moisture because they receive direct sunlight, are wind swept, and have little organic matter (Bell and Ungar 1982). During July the study area experienced normal hot and dry weather, causing high mortality of seedlings. Survival was only observed in plots that received fertilizer, suggesting that in the absence of nutrient additions, aspen seedlings cannot develop root systems extensive enough to survive normal summer stresses. Although limestone, by itself or in combination with wood chips, did not promote seedling survivorship, a very strong positive interaction with fertilizer occurred. One interpretation of this interaction is that the "double-dose" of limestone and fertilizer resulted in a deeper zone of soil amelioration and thus facilitated deeper root systems. Further experiments testing different levels of amendments, as well as soil sampling along a depth gradient are necessary to better understand the basis of this interaction.

Although applications of wood chips in combination with limestone and fertilizer had a significant effect on seedling survivorship (Table 5), a mycorrhizal factor cannot be implicated at this time. Medve et al. (1977) applied macerated roots to mine spoils and attributed subsequent colonization by aspens to a mycorrhizal effect. The present experiment was designed, in part, to test these findings by comparing mycorrhizal (root) and non-mycorrhizal (stem) wood chip treatments. Surprisingly, both treatments had beneficial effects on seedling survivorship in plots that also received limestone and fertilizer. Statistically, the two chip treatments cannot be distinguished. Either chip treatment provides a benefit unrelated to the presence/absence of mycorrhizae, or chipped aspen stems are not the non-mycorrhizal control intended. The hypotheses cannot be currently evaluated as we have not yet inspected the roots of aspen seedlings to evaluate mycorrhizal infection.

Final evaluation of this experiment is not possible until the chemical effects of the amendments dissipate. In most previous experiments where acidic mine spoils were amended with fertilizer and limestone and planted with nursery stock trees or agronomic herbaceous species, high mortality of the planted species occurred when amendment effects dissipated. The important difference between those efforts and the present experiment is that native, self-colonizing aspens were used. As already noted, established aspens can grow in extremely acidic, infertile spoils (Table 3). The important question is not whether aspens can survive in these soils but how long an aspen seedling must be subsidized with chemical amendments before it can survive in an unamended mine soil.

If the experiment proves successful, an easy method of revegetating acidic, barren spoils with native aspen would be to apply enough limestone and fertilizer to stimulate germination and establishment. A similar procedure has been used to revegetate acidic, metal-contaminated soils in the Sudbury, Ontario area (Winterhalder 1988). The cost of the fertilizer and limestone combination used in the experiment is less than $150 per acre. Application of the amendments can be done by hand or with conventional farm machinery. Thus the cost and needs of this method would be within the resources of many local groups who currently perceive the revegetation of barren mine sites as an impossibly expensive, engineering-intensive undertaking.

## Literature Cited


Baker, D. E. and M. C. Amacher. 1981. Development and interpretation of a diagnostic soil testing program. Penna. Agr. Exp. Sta. Bull. 826, University Park, PA.

Bell, T. J. and I. A. Ungar. 1982. Factors affecting the establishment of natural vegetation on a coal strip mine in southeastern Ohio. Am. Mid. Nat., 105:19-31.

Berg, W. and W. Vogel. 1973. Toxicity of acid coal-mine spoils to plants. p. 57-67. In: R. Hutnik and G. Davis (eds.) Ecology and Reclamation of Devastated Land, Vol 1., Gordon and Breach, New York.

Ciolkosz, E. J., Cronce, R. C., Cunningham, R. L. and G. W. Petersen. 1985. Characterization, genesis, and classification of Pennsylvania minesoils. Soil Sci., 139:232-238.

Deely, D. J. and F. Y. Borden. 1972. p. 69-79. High surface temperatures on strip-mine spoils. In: R. Hutnik and G. Davis (eds.) Ecology and Reclamation of Devastated Land, Vol 1., Gordon and Breach, New York.

Foy, C. D., R. L. Chaney and M. C. White. 1978. The physiology of metal toxicity in plants. Ann. Rev. Plant Phys., 29:511-566.

Hedin, R. S. 1987. The consequences of strip mine reclamation: vegetation and economics of reclaimed and unreclaimed sites in west-central Pennsylvania. Ph.D. Thesis. Rutgers University, New Brunswick, NJ, 314 p.

Hedin, R. S. 1988. Volunteer revegetation processes on acid coal spoils in northwestern Pennsylvania. p. 111-117. In: Mine Drainage and Surface Mine Reclamation, Volume II. U.S. Bureau of Mines, IC 9184.

Medve, R. J., Hoffman, F. and T. Gaither. 1977. The effects of mycorrhizal-forming amendments on the revegetation of bituminous stripmine spoils. Bull. Torrey Bot. Club, 104:218-225.

Salisbury, F. and C. Ross. 1978. Plant Physiology. Wadsworth Publishing Co., Belmont, CA., p. 422.

Winterhalder, K. 1988. Trigger factors initiating natural revegetation processes on barren, acid, metal-toxic soils near Sudbury, Ontario smelters. p. 118-124. In: Mine Drainage and Surface Mine Reclamation, Volume II. U.S. Bureau of Mines, IC 9184.

# Posters

This page intentionally left blank.

# Wildflowers for Reclamation[1]

Walter H. Davidson[2]

Abstract. A 1-acre demonstration seeding using a standard wildflower mix was established at the Peabody Coal Company, Colony Bay Mine at Wharton, West Virginia, in April 1989. Soil samples showed a pH range of 5.9 to 7.1. The site was prepared by tracking-in with a D-9 cat. Seeding was by hydroseeder with 400 pounds of 18-46-0 fertilizer, 600 pounds mulch, and 6 pounds of wildflower seed. By weight, 72.6 percent of the seed mix contained baby's breath (Gypsophila elegans), bachelor's button (Centaurea cyanus), calendula (Calendula officinalis), Chinese forget-me-not (Myosotis sylvatica), candytuft (Iberis umbellata), sweet William (Dianthus barbatus), blue flax (Linum lewisii), and wallflower (Cheiranthus allionii). The remaining 27.4 percent included a mix of 21 other species. After 4 weeks, germination was evident over the entire plot. At 8 weeks, baby's breath was conspicuous throughout, with lesser amounts of dames rocket (Hesperis matronalis), corn poppy (Papaver rhoeas), California poppy (Eschscholzia californica), candytuft, scarlet flax (Linum grandiflorum rubrum), bachelor's button, and toadflax (Linaria maroccana). Cover was estimated at 60 percent. An evaluation on September 14 showed all the above present plus cosmos (Cosmos bipinnatus), garland chrysanthemum (Chrysanthemum coronarium), wallflower, lance leaf coreopsis (Coreopsis lanceolata), plains coreopsis (Coreopsis tinctoria), black-eyed Susan (Rudbeckia hirta), calendula, blanketflower (Gaillardia aristata), and rose mallow (Lavatera trimestris). Total cover was estimated at 70 percent. However, the development of the stand was weak. The flowers were not as large or robust as they should have been. Possibly topsoil or additional organic material would have enhanced development of the wildflowers.

[1]Poster presented at the Conference: Reclamation Technology and Regulation: Achieving a Balance, held in Charleston, WV, April 23-26, 1990.

[2]Walter H. Davidson is Research Forester, Northeastern Forest Experiment Station, USDA Forest Service, Princeton, WV 24740.

This page intentionally left blank.

CONSTRUCTING TREATMENT WETLANDS-MARYLAND'S EXPERIENCE. Frederick L. Bagley and Constance A. Lyons, State of Maryland, Department of Natural Resources, Bureau of Mines, 69 Hill Street, Frostburg, Maryland.

Maryland's experience building wetlands to treat acid mine drainage (AMD) began in 1986. To date, we have constructed or enhanced an additional 10 systems. Four additional systems will be built or enhanced in 1990. Our experience in Maryland indicates that constructing treatment wetlands in the field requires designing for natural differences. Variations in seep chemistry, seep flows, topography, slope and accessibility are factors to consider when planning a wetland system. Designs should plan for seasonal changes and annual flow variations. Future expansion or enhancement strategies can be included during initial construction plans. Flexible construction techniques and proper equipment have helped address the diverse nature of the AMD problem. In Maryland, multiple seeps at AMD sites are a frequent problem. Under conditions of low flow and/or moderate seep chemistry, multiple seeps are pooled before beginning wetland cell treatment. With more severe seep conditions (>10 gal/min, pH<4.5 and, Fe>50 mg/l), systems that separate and treat smaller quantities of water appear more effective. Cell size to flow ratios are 1500-2000 sq ft or larger for each gal/min. Contouring wetland cells to the topography maximizes limited space and provides a more natural looking system. Maryland is presently filling cells with a deep layer of manure compost and hay. Cattails are planted throughout the cells. Snags, up-rooted tree stumps and large rocks are utilized to enhance the naturalness of the wetland. Maryland has a strong commitment to research and develop wetland treatment technology. Water quality data is collected monthly at all sites. Several research projects inhouse and with outside consultants are planned for this year.

ADDITIONAL KEY WORDS: Flow ratios, techniques, multiple seeps, research

This page intentionally left blank.

MINING AND RECLAMATION AT HOBET MINING, INC. IN WEST VIRGINIA.

L. Emerson, Hobet Mining, Inc., Madison, WV.

Hobet Mining, Inc. is one of the largest surface mining operations in West Virginia and the first to utilize a large walking dragline for overburden removal. The No. 21 mine located 28 miles southwest of Charleston employs the area mining method to remove up to four coal seams. A 27-cubic yard electric shovel and front-end loader remove overburden from the mountaintop down to the 5-Block coal seam. The material is transported by 170-ton rock trucks and placed into durable rock fills. After coal recovery, the remaining overburden is excavated by a 72-cubic yard dragline to expose the Upper and Middle Stockton-Lewiston coal seams. This mine has reclaimed over 1500 acres using this method and has been awarded three state awards for excellence in reclamation. Other features include successful use of a substitute topsoil, innovative on-bench sediment control techniques, durable rock fills, a wildlife stocking program, and a 3.3 mile overland belt conveyor for raw coal transport.

This page intentionally left blank.

# DIRECT SEEDING OF ANTHRACITE REFUSE USING COAL FLY ASH AS A MAJOR SOIL AMENDMENT [1]

John K. Buck, Robert J. Houston, and William A. Beimborn[2]

Abstract. The Pennsylvania Power and Light Company (PP&L) used fly ash as a major soil amendment to prepare a 10-acre anthracite refuse site for direct seeding. Results were monitored for four years. Details of the project and results of the first three years of monitoring were published in 1988 by Buck and Houston in the Proceedings of the Mine Drainage and Surface Mine Reclamation Conference (Vol. II, US Dept, of Interior Bur. of Mines IC 9184); this updates that presentation to include monitoring data collected in October 1988. The site is located at Harwood, near Hazleton, PA. Soil treatments were designed after evaluating the chemistry of laboratory mixtures of anthracite refuse and amendments. Fly ash treated and soil-covered control plot areas were seeded in September 1984. Fly ash amendment (200 tons/acre) improved the physical and chemical characteristics of the anthracite refuse by increasing the plant available water-holding capacity, shifting the USDA textural class of the refuse from sandy loam to silt loam and improving the pH and fertility of the coal refuse materials. Excellent erosion control and partial release of the revegetation performance bond were achieved in the establishment year. Vegetation response in fly ash-amended coal refuse areas was comparable to that in the soil-covered control plot. Monitoring through 1988 has shown that vegetation vigor has not decreased with time. Plant tissue analyses conducted in 1987 suggested normal uptake of nutrients and trace elements from fly ash-amended coal refuse materials, with the exception that uptake of Mo and Se were high. The pH and fertility of the growing medium remained acceptable four years after the fly-ash, limestone and fertilizer treatment.

[1]Paper presented at 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]Senior Environmental Specialist, Environmental Manager, and Lead Environmental Specialist, respectively, GAI Consultants, Inc., 570 Beatty Road Monroeville, PA 15146.

This page intentionally left blank.

# POTENTIAL APPLICATION OF WILDLIFE HABITAT EVALUATION PROCEDURES TO DESIGN RECLAMATION HABITATS: CASE STUDY FROM A 3400 ACRE RESERVOIR PROJECT[1]

W. A. Beimborn, G. T. Reese and L. J. Ealy[2]

Abstract. GAI Consultants, Inc., was contracted by the U.S. Army Corps of Engineers (COE) to design replacement wildlife habitat at Cowanesque Lake, Pennsylvania. These replacement upland and wetland habitats were designed to restore habitat unit losses incurred by raising the lake elevation in the spring of 1990. These habitat unit losses were previously determined by a Pennsylvania Modified Habitat Evaluation Procedure (PAM HEP) analysis of the project.

The mitigation plan included design of specific hedgerow patterns to replace upland wildlife habitat, design of wildlife habitat improvement structures for upland habitats, and design of 59 acres of replacing wetlands. Each hedgerow pattern was composed of a variety of plant species beneficial to wildlife, arranged in an innovative pattern which produced diverse, high quality wildlife habitat. This design has described by the Pennsylvania Game Commission as the best they had seen. GAI designed and prepared specifications for artificial den structures for the raccoon and red fox. These structures incorporated both natural and man-made materials and provided both underground and arboreal shelter. The wetlands were designed to provide high quality wildlife habitat by utilizing specific planting schemes and species composition. The wetland design included three types of wetland plantings based on zones of inundation. Barrier islands were designed to prevent ice scouring of the constructed wetlands.

[1]Paper presented at 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]Lead Environmental Specialist (Ph.D.), Environmental Specialist, respectively, GAI Consultants, Inc., 570 Beatty Road, Monroeville, PA 15146.

This page intentionally left blank.

# POTENTIAL APPLICATION OF WILDLIFE HABITAT EVALUATION PROCEDURES TO DESIGN RECLAMATION HABITATS: CASE STUDY FROM A 5 MILE LONG, 4-LANE HIGHWAY PROJECT[1]

William A. Beimborn, George T. Reese and Linda J. Ealy[2]

Abstract. GAI Consultants, Inc., was contracted to design a wildlife habitat mitigation plan for a proposed Pennsylvania Department of Transportation (PennDOT) new four-lane highway construction project near Ebensburg, Pennsylvania. The highway is expected to be completed in late fall 1990.

Mitigation requirements were previously determined (as habitat units) through application of a Pennsylvania Modified Habitat Evaluation Procedure (PAM HEP) to the project area. Mitigation procedures were developed to offset habitat unit losses resulting from highway construction. These mitigation procedures consisted of habitat management techniques designed to maintain quality habitat where it existed and to provide quality habitat and/or specific habitat components where needed. Management techniques encompassing nearly 150 acres included the planting and/or thinning of vegetation to provide food and cover. Soils in the mitigation area were analyzed for compatibility with proposed plant species. Several unique deciduous and evergreen hedgegrow patterns were designed to provide cover and food for wildlife. Each hedgerow was composed of several different tree and shrub species placed in a spatial arrangement which provided maximum diversity while discouraging the movement of wildlife perpendicular to the highway mainline. A Pennsylvania Game Commission Seed mixture was adapted to these plantings to provide food for wildlife, stabilize the soil, and provide minimal competition to planted trees and shrubs.

[1]Paper presented at 1990 Mining and Reclamation Conference and Exhibition, Charleston, West Virginia, April 23-26, 1990.

[2]Lead Environmental Specialist (Ph.D.) Environmental Specialist and Environmental Specialist, respectively, GAI Consultants, Inc., 570 Beatty Road, Monroeville, PA 15146.

This page intentionally left blank.

# ELECTROLYTIC CONTROL OF AMD

ALFRED STILLER
ERIC LELUIKA
THOMAS RYMER
WALTER HART

## ABSTRACT

Many abandoned underground mines in West Virginia experience continuing challenges in the control of acid mine drainage (AMD). Once an underground mine is completed the company is left with few options in the event that AMD becomes a problem. The general solution is to institute an AMD neutralization program consisting of the addition of an alkaline agent such as ammonia, caustic or calcium hydroxide coupled with a sedimentation pond to capture the resulting flocs (yellowboy). These systems are expensive to operate and the duration of treatment is indefinite. The less expensive neutralization agents such as calcium hydroxide require large capital intensive treatment plants while expensive agents such as caustic and ammonia can operate on inexpensive capital facilities. Costa in the range of $5,000 to $20,000 per month are not uncommon for mid scale AMD neutralization systems. An electrolytic process has been developed at the Chemical Engineering Department of West Virginia University as an alternative to chemical neutralization. It is known as the AMD-CELL SYSTEM. This proposal describes its application to an AMD discharge at Beth Energy's operations near Buckhannon, West Virginia. Electrolytic processes have been attempted in the past but have usually failed to operate economically due to their high current consumption. The proposed technology operates on a low current (< 2 amps) format and has been developed and run successfully as a lab bench demonstration unit. The object of this project is to evaluate, in a joint effort with Beth Energy, the scale-up potential of the technology, its effectiveness and practicality under operational field conditions.

This page intentionally left blank.

Wildlife Species Diversity on Revegetated Coal Stripmine Lands in Interior Alaska. Charles L. Elliott, Department of Biological Sciences, Eastern Kentucky University, Richmond, Kentucky 40475.

Wildlife utilization of revegetated stripmine spoils on the Usibelli Coal Mine in interior Alaska were studied from 1980 through 1982. Forty-nine species of birds were recorded on the mine site and adjacent undisturbed areas. Revegetated areas exhibited a lower bird species diversity than was found on undisturbed sites. Permanent water bodies appeared to mainly function as resting habitat for migrating waterfowl; while ephermal water bodies served as feeding areas for sandpipers and related shorebirds. Twenty-six species of terrestrial mammals were observed or trapped on the mine. The greatest number of small mammals observed or captured occurred on the undisturbed shrub tundra. This area also exhibited the greatest diversity of small mammals. Of those sites on which small mammals were observed or captured, the revegetated areas exhibited the lowest species diversity. Only one small mammal species, the tundra vole (Microtus oeconomus), was captured on all reclaimed sites.

ADDITONAL KEY WORDS: Wildlife and reclamation, Alaska

This page intentionally left blank.

Overview of Long-Term Effects of Surface Coal Mining on Ground-Water Levels and Quality in Two Small Watersheds in Eastern Ohio. William L. Cunningham, U.S. Geological Survey, Water Resources Division, Columbus, Ohio.

Two small watersheds in eastern Ohio that have been surface mined and reclaimed were studied during 1986-1989. The water-level and water-quality data collected were compared with premining (1976, 1976-80) and postmining (1977-83, 1981-84) data collected during previous investigations. Before mining, two aquifers were perched above clay beds underlying two major coal seams. The upper perched aquifers and the upper coal seams were removed during mining and were replaced with graded spoil. This created new perched upper aquifers with different hydraulic and chemical characteristics. The perched aquifers undisturbed by mining (middle aquifers) are beneath the underclays of the upper aquifers. Water levels in the upper aquifers reached hydraulic equilibrium from 2 to 5 years after mining. Water levels in the middle aquifers increased more than 5 feet during mining and reached equilibrium almost immediately thereafter. Statistically significant changes in pH and concentrations of many dissolved constituents occurred in both watersheds because of surface mining. In the upper aquifer of one watershed, for example, the premining medians for pH and concentrations of sulfate and dissolved solids were 7.0, 85 mg/L (milligrams per liter), and 335 mg/L, respectively; postmining medians were 6.7, 825 mg/L, and 1,495 mg/L, respectively. The premining median concentration of iron and manganese was 30 ug/L (micrograms per liter) for each constituent; postmining median concentrations were 31 ug/L and 1,015 ug/L, respectively. The water chemistry in the middle aquifer underwent similar changes. The continued decrease in pH indicated that ground water had not reached geochemical equilibrium in either watershed more than 8 years after mining.

ADDITIONAL KEY WORDS: Jefferson County, Muskingum County, geochemical equilibruim, hydraulic equilibrium, spoil aquifer.

This page intentionally left blank.

Using Laboratory Mesocosms to Evaluate the Potential Effectiveness of Constructed Wetlands for Acid Mine Drainage Treatment. R. Kelman Wieder, Associate Professor, Marianne N. Linton and Katherine P. Heston, Research Technicians, Villanova University, Department of Biology, Villanova, PA.

Although constructed wetland treatment of acid mine drainage has grown in popularity, treatment efficiency is both highly variable and difficult to predict. A set of laboratory mesocosms was used to evaluate spatial and temporal patterns of metal retention in wetlands exposed to acid mine drainage. Six model wetlands (2.4 m x 15 cm) were filled to a depth of 15 cm with Sphagnum moss and were planted with cattails (Typha latifolia), living Sphagnum, or left with no living vegetation. Over 16 weeks, each of the wetlands was exposed to synthetic acid coal mine drainage (pH 3.5, concentrations of Fe, Al, Mn, Ca, Mg of 75, 10, 5.2, 12 and 4.5 mg/L, respectively) at a rate of 90 mL/min, 6 hr/da, 5 da/wk. Analysis of metal fractions (exchangeable, organically bound, oxide, residual) in the peat indicated that the wetlands were net sources of Al, Mn, Ca, and Mg, but 79% of the added Fe was retained, with no significant effect of vegetation type. At 16 weeks, the Fe oxide fraction comprised 73-86% of total Fe in the peat, indicating that oxide formation was the major process contributing to Fe retention. Fe retention appeared to reach saturation at the inflow ends of these wetlands at a total Fe concentration of 23% of the dry mass of the peat. Laboratory mesocosms may be useful for predicting the lifetime of effective treatment for a wetland of a given size receiving a known flow and chemistry of mine drainage. The ability to make such a prediction would improve the presently minimal ability to formulate a priori cost-benefit analyses regarding contemplated wetland treatment of acid mine drainage.

This page intentionally left blank.

This page intentionally left blank.

ISBN 0-925500-02-X

Printed by Fairmont Printing Company
Fairmont, WV 26554