Dynamics and Characterization of Soil Organic Matter on Mine Soils 16 years after Amendment with Topsoil, Sawdust, and Sewage Sludge.

By

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Thesis submitted to the Faculty of
Virginia Polytechnic Institute and State University
in partial fulfillment of the requirements
for the degree of

MASTER OF SCIENCE

in

FORESTRY

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July, 1999
Blacksburg, Virginia

Keywords: soil quality, organic amendments, reclamation, and reforestation

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DYNAMICS AND CHARACTERIZATION OF SOIL ORGANIC MATTER IN MINE SOILS 16 YEARS AFTER AMENDMENT WITH TOPSOIL, SAWDUST, AND SEWAGE SLUDGE.

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(ABSTRACT)

The present state and future prospect of the world's soil resources has prompted scientists and researchers to address the issue of soil quality and sustainable land management. Soil quality research has focused on intensively-managed agricultural and forest soils, but the concept and importance of soil quality is also pertinent to disturbed systems such as reclaimed mine soils. The restoration of soil function and mine soil quality is essential to long-term ecosystem stability. The objectives of this study were (i) to determine the comparative ability of topsoil, sawdust, and sewage sludge amendments, after 16 years, to positively affect mine soil quality using the following key soil quality variables: organic matter content, aggregate stability, and mineralizable nitrogen, (ii) to determine the effects of these key soil quality variables on plant productivity, and (iii) to determine the comparative ability of trees and herbaceous plants to persist and to conserve or maintain mine soil quality. In 1982, a mined site was amended with seven different surface treatments: a fertilized control (2:1 sandstone:siltstone), 30 cm of native soil + 7.8 Mg ha^{-1} lime, 112 Mg ha^{-1} sawdust, and municipal sewage sludge (SS) at rates of 22, 56, 112, and 224 Mg ha^{-1}. Four replicates of each treatment were installed as a randomized complete block design. Whole plots were split according to vegetation type: pitch x loblolly pine hybrid (*Pinus rigida* x *taeda*) trees and Kentucky-31 tall fescue (*Festuca arundinacea* Schreb.). Soil analyses of composite samples for 1982, 1987, and 1998 were evaluated for changing levels of mine soil quality. The positive effect of these organic amendments on organic matter content, total nitrogen, and other soil parameters was most apparent and pronounced after 5 growing seasons. However, after 16 years, soil organic matter content and total nitrogen appear to be equilibrating at about 4.3 and 1.5%. There was a significant difference in organic matter content and nitrogen mineralization potential between vegetation types. Organic matter inputs by vegetation alone over the 16-yr period in the control plots resulted in organic matter and nitrogen mineralization potential values comparable to levels in the organically amended plots. The results suggest that about 15 years is needed for climate, moisture availability, and other edaphic features to have the same influence on overall organic matter decomposition, N accretion, organic nitrogen mineralization levels, system equilibrium, and overall mine soil quality as a one-time 100-Mg ha^{-1} application of organic amendment. Tree volume and biomass were measured as indices of the effects of organic matter content 16 years after initial amendment. Individual tree volumes of the sawdust, 22, 56, and 112 Mg ha^{-1} SS treatments retained 18 to 26% more volume than the control, respectively. Overall, fescue production was the same among treatments. Organic amendments improved initial soil fertility for fescue establishment, but it appears that they will have little or no long-lasting effect on plant productivity.

Key words: soil quality, reclamation, organic amendments, reforestation

ACKNOWLEDGEMENTS

I am grateful to Dr. James A. Burger for his guidance and counsel during my graduate program, for his professionalism and dedication to quality research, and for his devotion to his students. I thank the rest of my committee members for their advice and encouragement: Dr. A. Ozzie Abaye, Dr. W. Lee Daniels, and Dr. Charles M. Feldake. Dave Mitchem deserves special thanks for his assistance in the field and laboratory. His friendship and patience is greatly appreciated. I am thankful for my fellow students and the camaraderie that I have enjoyed during graduate school and as part of the forest soils program. I thank the Powell River Project for their financial support to this research.

Finally, and most importantly, I am grateful to my wife, Mary, and children, Freeman and Hannah, for their loving support and encouragement. I will always be thankful for their patience and understanding during this endeavor.

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CHAPTER I. INTRODUCTION

STUDY RATIONALE

Soil degradation and the decline in the productive capacity of land resources is an immediate environmental and social concern. Decline in soil productivity and quality invariably affects all aspects of human welfare and the environment. Increased awareness of soil degrading processes, declines in global food production, and the limited ability of synthetic fertilizers to positively influence long-term soil physical, chemical and biological properties have been the impetus for recent soil quality discussion and research. However, the issue of managing natural resources responsibly has been promoted and emphasized by scientists and land managers for many years. Soil degradation and loss of soil quality has been of particular interest to intensively-managed agriculture and forestry sectors as available land resources diminish and currently managed lands require more inputs to maintain current yields.

Within a surface mining context, the detrimental effect of mining on soil, water, and air quality has been and will continue to be a subject of concern. Soil degradation and the environmental impact of mining operations are extensive and are expected to increase as society's energy use and demand continues to escalate as world population grows and new technologies are developed. Between 1930 and 1971, U.S. mining operations disturbed 1.6 million ha of land, and only 40 % of this land has been reclaimed (Sopper, 1992). This disturbance is expected to increase as more mining resources are identified. In 1988, U.S. coal resources amounted to 2.15 trillion Mg. Of these coal resources, 396 billion Mg are classified as reserve bases and subject to surface and deep mining. In Virginia, 616 million Mg of reserve base coal is accessible by surface mining (Doolittle and Hossner, 1988). Doolittle and Hossner (1988) and McMahon (1987) estimate that surface mining of reserve coal bases will disturb 4 million ha of land in the U.S. (excluding Alaska and Hawaii) alone.

The mass disruption of natural ecosystems and the subsequent decline in soil productivity, water quality, and ecosystem functions during coal mining operations were part of the impetus for the Surface Mining Control and Reclamation Act (SMCRA) of 1977 which was promulgated in anticipation of continued coal use and future land disturbances. The long-term success of land reclamation efforts requires the maintenance and improvement of soil quality and the restoration of soil physical, biological, and chemical functions. Sopper (1992) states that soil ecosystem stability is a function of soil organic matter accumulation, transformations, and function that may require 30 years or more to come to equilibrium. Hence, establishing vegetative cover and monitoring initial productivity may not ensure long-term reclamation success; 5 yr growth patterns stipulated by legislation for eastern USA conditions may be followed by reductions in plant productivity, organic matter accumulation, and plant available N and P that would indicate a lack of recovery. Due to the complex interactions and dynamic nature of young mine soils, numerous researchers have emphasized the need for monitoring soil functions through time. Evaluation of reclamation research on the changing levels of mine soil quality through time will enable management procedures, reclamation guidelines, and practices to be altered or corrected in accord with new information gathered.

A fundamental provision of the SMCRA law requires that land disturbed by mining be reclaimed to its original use or one of higher value, and that its productivity be equal to or greater than it was before mining. The need for maintenance and improvement of mine soil quality is implicit in this provision if land productivity is to meet or to exceed pre-mining conditions. In regard to soil improvement, Tiessen et al. (1994) maintain that land management strategies have to be implemented that minimize organic matter and nutrient losses, aim to sustain soil fertility of present lands in production, reverse soil degrading processes, and enhance the potential productive capacity of marginal or depauperate soils. The long-term restoration of disturbed sites and ecosystem function depends on reclamation that allows re-establishment of nutrient cycles so vegetation can thrive and be sustained in perpetuity. Bradshaw (1987) reiterates the importance of achieving a stable, self-sustaining vegetative cover. He points out that the reconstruction of an ecosystem is dependent on vegetation for improving the physical and biological diversity of disturbed sites. A presupposition for establishing vegetation is a soil medium that allows for adequate growth and productivity.

Three processes that are integral to soil function and provide an amenable growth medium are organic matter accumulation, aggregate stability, and nitrogen mineralization (Burger and Kelting, 1998; Karlen et al., 1997; Bauer and Black, 1994). Seaker and Sopper (1988) propose characteristics of productive reclaimed mine soils, that are closely associated with these mechanisms, that include organic matter accumulation and decomposition, organic C and N contents, root proliferation, and microbial activity. In agriculture, manures and synthetic fertilizers are traditional soil conditioners added to enhance these three processes. On drastically disturbed sites, organic amendment treatments such as municipal biosolids have been used to ameliorate disturbance, accelerate nutrient cycling, provide a receptive environment for vegetation, and improve overall soil quality and productivity.

Organic amendments may accelerate the initial process of reclamation, but the vegetation that is established on these sites should also be assessed to determine how trees and herbaceous plants conserve or maintain soil quality through time and potentially assist in the attainment of long-term reclamation goals. A basic assumption with organic amendment treatments is that their additions will positively affect key soil quality variables that include soil organic matter, aggregate stability, and mineralizable nitrogen. A second supposition is that if these key variables are effectively influenced with organic amendments, overall plant and forest productivity will be enhanced. Consequently, improved plant and forest productivity may conserve or maintain mine soil quality through positive or negative feedback mechanisms. Feedback may be monitored by evaluating the changing levels of key soil indicators. If the feedback is positive and levels increase, the vegetation will be self-perpetuating and mine soil quality will be sustainable. However, if the feedback is negative, vegetation may regress and soil quality may be maintained at a lower level than desired. Hence, periodic fertilization or more drastic management efforts will be required to maintain the site and the vegetation. The intent of reclamation efforts is to restore soil functions so vegetation can thrive and so the whole system can become self-sustaining. These scenarios and the objectives of this organic amendment study are depicted in figure I.1.

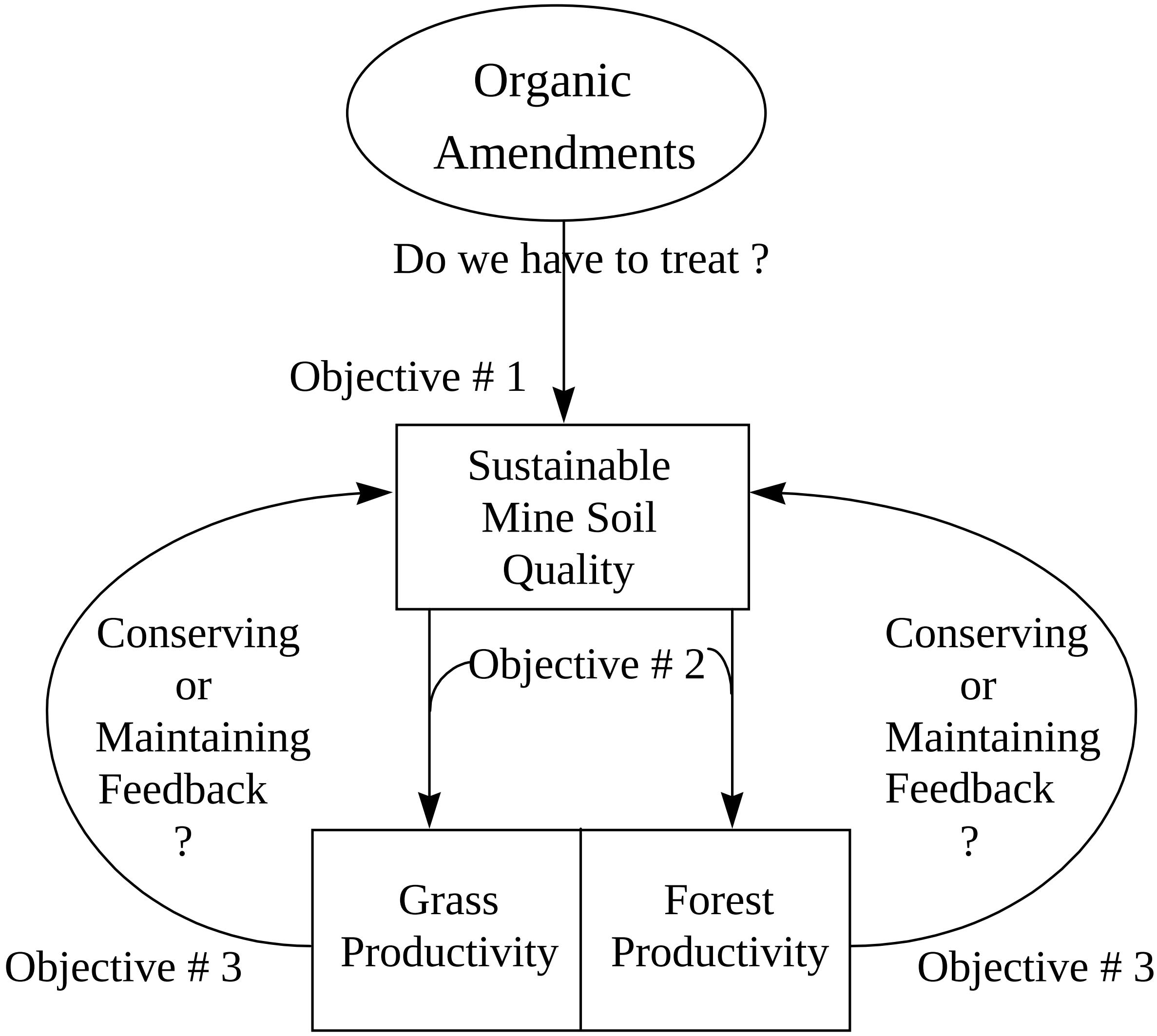

Figure I.1. A conceptual model of how sustainable mine soil quality might be achieved with organic amendments, improved soil function, and positive feedback from vegetation. The objectives of this research are to determine the effects of organic amendments on sustainable mine soil quality as measured by changing levels of key soil and plant variables.

Long-term ecosystem restoration and sustainable mine soil quality depend on soil functions and positive feedback. In this context, positive feedback is defined as the conservation or addition of organic matter or nutrients added to the system by vegetation alone. Characterization and knowledge of important mechanisms such as soil organic matter accretion, aggregate stability, nitrogen mineralization, and other soil properties will help natural resource managers assess ecosystem stability, soil function and quality of marginal or depauperate soil. Assessment of these functions and qualities will allow for earlier intervention and proper design of site specific management practices. Tree and vegetation response to soil organic matter amendments through time is also critical in assessing whether mined land reclamation requirements that meet short term success standards are effectively achieving long-term ecosystem and sustainable mine soil quality goals. Similarly, comparison of tree and vegetation response to unamended and amended overburden material will allow evaluation of the effects that tree and herbaceous vegetation alone have on the rehabilitation and recovery of soil quality on degraded sites.

OBJECTIVES

An important aspect of land management on disturbed and marginal soils is the restoration and maintenance of key soil quality variables including soil organic matter, mineralizable nitrogen, and aggregate stability for soil fertility, plant productivity, and building soil structure. Most mine soils and many agricultural soils have very low organic matter content, limited N availability, and poor aggregate stability. The objectives of this study were:

(i) To determine the comparative ability of topsoil, sawdust, and sewage sludge amendments, after 16 years, to positively affect mine soil quality using the following key soil quality variables: organic matter content, aggregate stability, and mineralizable nitrogen.

(ii) To determine the effects of these key soil quality variables on plant productivity.

(iii) To determine the comparative ability of trees and herbaceous plants to persist and to conserve or maintain mine soil quality.

CHAPTER II. LITERATURE REVIEW

INTRODUCTION

Natural ecosystems are generally physically and biologically diverse and the function of components' interaction and interdependency. The interaction and interdependence of soil components are often expressed as a function of vegetation, climate, parent material, topography, biota, and time. The interdependency of vegetation is similarly expressed as a function of the same variables and the addition of a soil component (Ashby, 1987; Kimmins, 1997). Due to the impact of humans on soil and vegetation communities, humankind has recently been included as potentially the sixth soil-forming factor (Karlen et al., 1992). In studying the effects of perturbations on natural ecosystems, knowledge of these complex interactions is needed to understand how to mitigate adverse effects and restore the system as a whole. Waring and Schlesinger (1985) explain that ecological studies must encompass how energy and matter is circulated, transformed, and accumulated through biotic and abiotic influences. Ecosystem stability is achieved as the individual components and populations adapt to the biotic and abiotic factors of the site and climate.

Odum (1989) explains that as vegetation and species diversity increases within a community, so does the resilience stability and the ability to recover from disturbance. Odum also states that species diversity may also increase the ability of ecosystems to resist change caused by disturbances. The integration and proper function of the diverse physical, chemical, and biological components of a mine soil ecosystem are important goals for, and characteristics of, successful reclamation efforts. The proper function and whole restoration of these components may increase the resilience and resistance of these ecosystems to future disturbance, improve the hydrologic balance of mine soils, and enhance mine soil quality.

Ecosystem Stability

Stability is defined as requiring little input or minimum maintenance. Nutrient cycling and stable vegetation are two components critical to the proper function of natural ecosystems and their long-term stability. These two components are also paramount to the restoration of degraded sites. Waring and Schlesinger (1985) point out that in early stages of succession, forest growth and productivity are modulated by abiotic factors such as nutrient availability, climate, parent material, and time. However, in the latter stages of succession, forest productivity is regulated by biotic factors and the positive feedback that vegetation provides for nutrient cycling. The long-term reconstruction of mine soil quality is also dependent on and could benefit from the early establishment of nutrient cycles and the later development of a positive biological feedback by vegetation. Nutrients in plant available forms and appropriately balanced can enhance forest and/or plant productivity enabling vegetation to more effectively ameliorate disturbed sites (Aber, 1987).

For instance, trees such as pine (*Pinus* spp.) can positively influence the water budget, soil porosity and structure, soil organic matter accumulation, soil temperature, and nitrogen availability (Ashby, 1987). These are a few site variables, which in turn also affect tree growth. Leaf litter and aboveground return of nutrients to the soil are also pertinent to tree growth and ecosystem stability. The annual transfer of nutrients from foliage and litterfall to the soil significantly contributes to N, P, K, Ca, and Mg availability (Waring and Schlesinger, 1985). Plants rely on decomposition of litterfall and internal cycling to meet annual nutrient requirements. Knowledge of these plant and site interactions will assist in evaluating if reclaimed sites are beginning to function as neighboring undisturbed or stable ecosystems.

Soil Degradation

As world population approaches six billion, the importance of soil fertility and conservation continues to increase. Global food, fiber, and energy needs continue to escalate with rising population, necessitating the cultivation of available land. Jenny (1980) reminds us that soil is formed by unique factors and is essentially a nonrenewable resource that must be managed wisely by society. However, much of the available land can be characterized as marginal or depauperate as 78 % of the total earth's surface area is categorized as unsuitable for agriculture (Lal and Stewart, 1992). Degradation refers to a significant loss of the soil quality that requires specific and significant remedial measures to restore soil functions (Bouma, 1997). In a recent assessment of global soil degradation, the United Nations Environmental Program (UNEP) states that approximately 40 % of agricultural soils have been adversely degraded by anthropogenic causes. The U.N. program found 6 % of this land irreparably degraded so that restoration to an acceptable level of productivity is economically unfeasible.

World Resource Institute estimates 2 billion ha of soil have been degraded to some extent (Lal et al., 1995) and the process of soil degradation continues. It is estimated that 5 to 7 million ha of arable land are annually degraded. An estimated 3.7 billion ha are susceptible to desertification (Lal and Stewart, 1992). In the African Sahel, desert conditions are expanding a distance of 17 km annually (Harrison, 1990). There are approximately 323 million ha of land affected by salinization. Similarly, many lands are sensitive to accelerated erosion and rivers to sedimentation (Lal and Stewart, 1992).

In North America, Bremer et al. (1994) report that within agricultural soils approximately 15 to 40 % of original soil organic matter has been lost due to cultivation. This decline in soil organic matter and nutrient content due to intensive agriculture and the non-renewable nature of some constituents has been termed “fertility mining” (Lal and Stewart, 1992). Several degrading processes occur when land is mismanaged: soil erosion by wind and water, compaction, crusting, loss of soil organic matter, nutrient imbalance, acidification, salinization, alkalization, inadequate soil aeration and drainage (McBride, 1994; Arshad and Coen, 1992; Lal and Pierce, 1991). Soil degradation and disturbances are endemic to all regions of the world.

Sustainable Mine Soil Quality

Due to the rate and extent of land degradation, achieving sustainability has been and continues to be a subject for discourse within the scientific community. Scientists and practitioners have tried to identify specific soil indicators that confer attributes of soil quality, fertility, and health (Larson and Pierce, 1994; Karlen and Stott, 1994; Doran and

Parkin, 1996), and are essential to overall soil function and sustainable systems. Key indicators of soil quality include physical, chemical, and biological properties. Karlen et al. (1992) maintain that the nutritional quality of plants produced should also be used as a criterion for evaluating soil quality. Soil quality has been defined as the capacity of soil to function, within ecosystem boundaries, to sustain biological productivity, maintain environmental quality, and promote plant, animal, and human health (Doran and Parkin, 1994).

More recently, soil quality has been addressed as a criterion for sustainable forest management (Burger and Kelting, 1998; Powers et al., 1998). The goal of sustainable forestry practices is to ensure that vital soil functions are maintained at qualitative as well as quantitative levels in perpetuity. In forest soils, these functions are the ability (i) to sustain plant and animal productivity, (ii) modulate hydrologic, carbon, and nutrient cycles, and (iii) maintain the structural and environmental integrity of land systems (Burger and Kelting, 1998). Like agricultural soils, forest soils are characterized by complex interactions and dynamic processes; therefore, they can be profoundly affected by intensive management practices. Hence, the variability of soils in time and space must be considered in soil management and sustainable forestry practices. Burger and Kelting (1998) propose that key forest soil variables be monitored on a spatial and temporal scale to facilitate management of these soils and enhance the resolution and ability to evaluate changes in soil quality and productivity than allowed by present management. Monitoring forest soils on a finer spatial and temporal scale will allow for systematic detection of soil degradation and ensure that long-term goals of sustainable forestry are appropriately quantifiable at a manageable reference unit.

Soil physical, chemical, and biological properties proposed as basic indicators or part of a minimum data set (MDS) of soil quality are soil tilth, soil organic matter (SOM), total organic nitrogen and carbon, aggregate stability, aeration, macroporosity, water holding capacity, microbial biomass, mineralizable C and N, bulk density, resistance to erosion, nutrient retention capacity, pH, and electrical conductivity (Doran and Parkin, 1994; Larson and Pierce, 1994; Karlen et al., 1992; Parr et al., 1992). Larson and Pierce (1994) identified soil organic matter content as one of the most important soil properties that contributes to soil quality and stability. Soil organic carbon was also designated as a master variable to estimate soil attributes such as cation exchange capacity (CEC), water retention characteristics, and leaching potential (Doran and Parkin, 1996). Similar soil properties are used as analytical parameters for mind land reclamation (Burger and Kelting, 1998; Williams and Schuman, 1987), along with P-availability, acid forming parent material, and heavy metal concentrations (Torbert et al., 1989; Daniels and Zipper, 1988; Daniels and Amos, 1981). These soil functions are particularly important on surface mined and other disturbed sites where the capacity of soils has been dramatically altered. Identifying important soil properties and potentially growth limiting factors are essential for managing soil resources and assessing the soil’s capacity to function and meet sustainable goals.

Visser and Parkinson (1992) explain how different microorganisms function in basic soil processes such as C and N cycling and vary in their sensitivity to ecosystem stresses, hence, microorganisms are also sensitive soil quality indicators. Physical attributes like aggregate stability are also a result of biological activity and the byproducts formed during SOM decomposition (Karlen et al.,1992). Polysaccharides, roots, and fungal hyphae are all integral in the formation of aggregates. Tate (1985) summarized the role and importance of microorganisms in mine soils and ecosystem restoration. He concluded that soil structure development, nutrient cycling, and soil chemical and physical limitations to plant growth are mediated and mitigated by microorganisms. Soil quality assessment of this diagnostic nature will assist natural resource managers and serve as a baseline for evaluating the sustainability of present management practices.

Sims et al. (1997) state that soil quality has become a much broader environmental issue as urbanization, industrialization, and suburban sprawl impact more ecosystems and soils. Within the context of increased environmental impacts, some researchers and advocacy groups have proposed that a soil quality act be federally legislated, similar to present air and water quality legislation, to monitor and regulate the health and cleanliness of soils (Sims et al., 1997; Wagenet and Hutson, 1997). The need for systematic monitoring and management of the world's soil resource base is unarguable.

Organic Matter and Mine Soil Quality

In studying the long-term fate and potential benefits of organic amendments on mine soil quality, the definition and role of organic matter in soils must be thoroughly understood. Soil organic matter is defined by the Soil Science Society of America as the organic fraction of soil exclusive of undecayed plant and animal residue (SSSA, 1987). Soil organic matter can be classified on a continuum from easily decomposable material (simple sugars and proteins), slowly decomposable or physically protected (cellulose and hemi-cellulose) and recalcitrant or passive materials (lignins, waxes, chitins, and polyphenols)(Gregorich et al., 1994). Soil organic matter is a product of plant residues, root slough of epidermal cells, rhizodeposition of organic compounds, and microbial biomass. Stevenson (1994) defines soil organic matter as including litter, microbial biomass associated with the light organic fraction (< 2.0 g cm^{-3}), and humus. The primary plant nutrient found in soil organic matter is carbon comprising 45 to 58% by weight. The decomposition and resynthesis of carbon and these materials are mediated by microbial activity that comminute plant litter, churn soil, and facilitate nutrient turnover rates. Soil humus is one product of this resynthesis and catabolic processes and is defined as the soil fraction that no longer visibly resembles the organic material from which it was derived (McBride, 1994).

Rhizodeposition of organic compounds was shown to be significant during the vegetative growth stage in a study by Sauerbeck and Johnen (1977). In this study, the researchers used ^{14}C as a tracer to track root growth and decomposition in mustard and wheat residues. The researchers hypothesized that more root matter is lost and organic carbon formed during growth than is usually accounted for by normal methods of examination. Using the tracer, root organic carbon was 20 to 50% higher than the other means of calculation. Therefore, the contribution of rhizodeposition to organic matter formation is 2 to 3 times larger by Sauerbeck and Johnen’s estimation. Marschner (1995) states that 28 to 59 % of net photosynthetic carbon in annuals is allocated to root formation. Of this percentage, 42 to 90% is lost by the root to the soil through respiration and rhizodeposition. Carbon cycle studies have also concentrated on carbon loss after deposition in the rhizosphere as plant litter and root slough is converted to soil humus. Young (1989) cites earlier work by Nye and Greenland (1960) that states that 10 to 20 % of litter carbon is converted into soil humus and 20 to 50% for root residues. From these figures, the researchers estimated loss of carbon to the atmosphere in litter to humus conversion to be 80 to 90 % for plant litter and 50 to 80% for root residues.

The soil organic matter formed from rhizodeposition and other soil processes is integral to the cycling of other macronutrients. Soil organic matter in the surface horizon accounts for 95 % or more nitrogen (N) and sulfur (S) and 20 to 75 % of total soil phosphorus (Brady and Weil, 1996; Paul and Collins, 1998). Nitrogen and P are two of the most limiting nutrients in most forest and some agricultural ecosystems. The C:N:S ratio of agricultural soil is approximately 130:10:1.3 and 200:10:1 for forests and prairie grasslands (Paul and Collins, 1998).

Recent soil organic matter studies have focused on identifying the bio-reactive or labile fractions of soil organic matter pools; that is, the plant and microbial materials that are being actively transformed or mobilized through biological processes that contribute to soil fertility and quality. Gregorich et al.(1994) in an effort to assess the quality of soil organic matter content proposed a minimum data set of pertinent soil attributes associated with these active fractions. The proposed minimum data set is composed of total soil C and N, light fraction and macro-organic matter, mineralizable C and N, microbial biomass and soil respiration, carbohydrate content, soil enzyme activity, and water soluble organic carbon. Aggregate stability is not included in this minimum data set, but, a strong positive correlation between soil organic matter content and aggregate stability has been reported in several studies (Tisdall and Oades, 1982; Chaney and Swift, 1986; Oades, 1993; Beare et al., 1994). Similarly, aggregation and the distribution of soil organic carbon in particle size fractions are being investigated for their role in soil organic matter storage and carbon sequestration (Carter and Gregorich, 1997).

Soil organic matter is an important component in the management and maintenance of soil fertility, productivity, and quality. However, organic matter content in surface soils is highly variable, ranging from < 0.1 % in some arid conditions to 100 % in organic soils (Stevenson, 1994), and can be rapidly lost by poor management. Despite this variability, soil organic matter influences many soil constituents and properties. Childs et al. (1989) studied the importance of soil physical properties on long-term forest productivity and found that organic matter content directly affects several soil properties and processes. Young (1989), in a review of how agroforestry contributes to soil conservation, lists the physical, chemical, and biological effects of organic matter on soil fertility. Similarly, Wander et al. (1994), in a study on the effects of organic and conventional agricultural management of biologically active soil organic matter pools, reiterate how soil organic matter is associated with many physical, chemical, and biological properties that positively influence soil productivity.

Physical properties affected by organic matter content are porosity, structural stability, aeration, water storage, infiltration and water flow, heat storage and flow, and strength. Soil carbon and organic matter content are also pertinent to soil chemistry. Organic matter increases retention of nutrients by cation exchange, enhances the availability of nutrients (e.g., NH_4^+, NO_3^-, PO_4^{3-}, SO_4^-) and trace elements by mineralization, improves soil buffering capacity, chelates metallic ions increasing the availability of some nutrients, and decreases the toxicity of other ions. (McBride, 1994; Stevenson, 1994; Fisher, 1995; Johnson, 1995). On a biological level, organic matter improves microbial activity and adsorbs pollutants such as Pb and Cu and certain pesticides (Young, 1989; McBride, 1994; Brady and Weil, 1996).

Environmentally, soil organic matter content and soil carbon storage is of concern as a potential source or sink for atmospheric carbon dioxide and other greenhouse gases (Kirschbaum, 1994; Flach et al., 1997). Therefore, knowledge of how soil carbon and organic matter aggrades or degrades in soil is integral to any land management plan, but particularly on landscapes where large amounts of native topsoil and soil organic matter have been displaced by previous land use.

Organic Amendments and Soil Quality Attributes

Recently, municipal biosolids have been applied as organic amendments to disturbed sites to increase soil organic matter content, improve nutrient cycling, and enable vegetation establishment. Johnston (1991), in a study on soil fertility at the Rothamsted experiment station in England, explains that some of the important functions of organic matter in soil cannot be replaced by synthetic fertilizers. Nair (1993) states that if organic matter content is maintained at proper levels, overall improvements in nutrient cycling and productivity can be achieved. Municipal wastes and biosolids have been applied to agricultural land, forests, and minesoils as organic matter amendments to improve soil fertility and restore ecosystem integrity (Sopper, 1992; Page and Chang, 1994). Lue-

Hing et al. (1994) report that 2.34 million dry Mg of municipal sewage sludge were land applied in 1989 and that this amount represents 33.3 % of the total sewage sludge produced in the U.S.. Management and application of municipal biosolids is costly due to transportation expenditures and logistical constraints that occur when transporting mass quantities of biosolids from one locality to another. Hence, applications are generally limited to areas close to municipal sources.

Soil productivity of disturbed sites has increased with these applications (Daniels and Haering, 1994). Similarly, several studies have shown that with proper management there is little or no deleterious effect from municipal sewage sludge applications on the environment. The public remains wary of the threat of heavy metals entering the food chain, phytotoxicity of trace elements, the hazards that surface runoff of PO_4^{2-}, or NO_3^- leaching pose for water quality, the fate of pathogens, and to a lesser degree, offensive odors (Henry et al., 1994; Page and Chang, 1994).

Despite these concerns, Harrison et al. (1995) adds that municipal biosolids have enhanced initial productivity of forest lands. However, they point out that the actual long-term fate of soil organic matter and nutrients added in the form of municipal biosolids is not clear. Furthermore, it is not known if applications have a permanent effect on nutrient and soil organic matter contents. This point is particularly pertinent in mined land reclamation on sites that have received a one time application or amendment of municipal biosolids. Does soil productivity recover and aggrade after the actual 5 yr bond period? Actual site productivity may degrade if soil organic matter and nutrient contents are decreasing at the time of bond release and established vegetation is not productive enough to properly replenish soil organic matter and nutrient pools (Sopper, 1992). Barrios et al. (1996) claim that for soil organic matter to be in equilibrium and maintained, ecosystem outputs must be balanced with proper inputs. Harrison et al. (1995) categorize the potential fate of soil organic matter and nutrient contents after amendment as follows:

(1) Increases in organic matter and nutrients above the level of application due to synergism of the productive capacity and/or internal retention mechanisms.

(2) Organic matter and nutrient levels enhanced from amended level.

(3) A gradual loss of or return of added organic matter or nutrients to original nutrient content.

(4) Loss of organic matter or nutrients beyond that added due to an antagonistic effect of the original amendment, such as increased cation leaching due to nitrification.

These scenarios are depicted in figure II.1. The trend of soil quality recovery as it pertains to the intent of SMCRA and long-term ecosystem stability may follow similar patterns as soil organic matter and nutrient content increases or decreases through time.

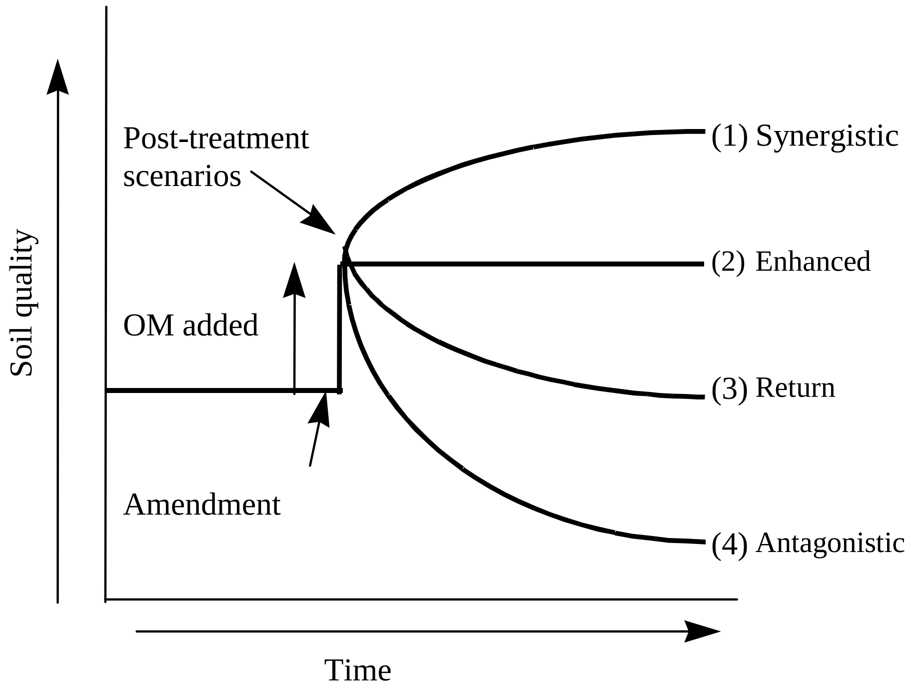

Figure II.1. Hypothetical changes through time of soil quality and post-treatment scenarios within the organic amendment study.

However, given the logistical constraints of transporting these amendments a great distance, and public concern about land application of municipal biosolids and the potential ephemeral effect of these amendments on long-term soil productivity, simply applying organic amendments alone without further monitoring of vegetation and successional processes may be insufficient for restoring a disturbed ecosystem and sustainable mine soil quality.

Soil Organic Matter Fractionation

Classical Chemical Fractionation

Generally, soil organic matter fractionation procedures seek to partition the humified and non-humified substances. Two categories that have been predominantly used in laboratory analysis to isolate organic matter are chemical and physical fractionation. Classical chemical fractionation is based on the different solubility of organic material in alkali and/or acid solutions such as NaOH (sodium hydroxide) or $Na_4P_2O_7$ (sodium pyrophosphate) (Cheng and Mollina, 1995; Stevenson, 1994; Turchenek and Oades, 1979) (Figure II.3). NaOH and $Na_4P_2O_7$ at a 0.5 N to 0.1 M concentration dissolves soil organic carbon very effectively, removing the materials that are associated with polyvalent cations. Subsequent acidification of the extract to a pH of 2 divides the substances into three fractions: humic acid (HA), fulvic acid (FA), and humin. Humic acids are soluble in alkali solutions but precipitate at a pH of 2. Humic acids are the most stable fractions and increase in predominance as soil organic matter degrades (Paul and Collins, 1998). Fulvic acids are soluble in both alkaline and acid solutions, but not at neutral pH (Sanchez, 1998). Humin is insoluble in water at all pH levels. Sanchez (1998) states that the humin fraction is closely associated with clay particles and tightly bound by polyvalent cations. The limitations of chemical fractionation based on solubility are that alkaline extracts dissolve silica altering the soils structural framework and exposing organic constituents to oxidative processes thereby changing the inherent nature of the substance. The procedure is laborious, time-consuming and only suitable for a very small number of samples. These three fractions have varying turnover rates with fulvic acid having a rate of a few hundred years and the other two fractions even slower at thousands of years (Collins et al., 1997).

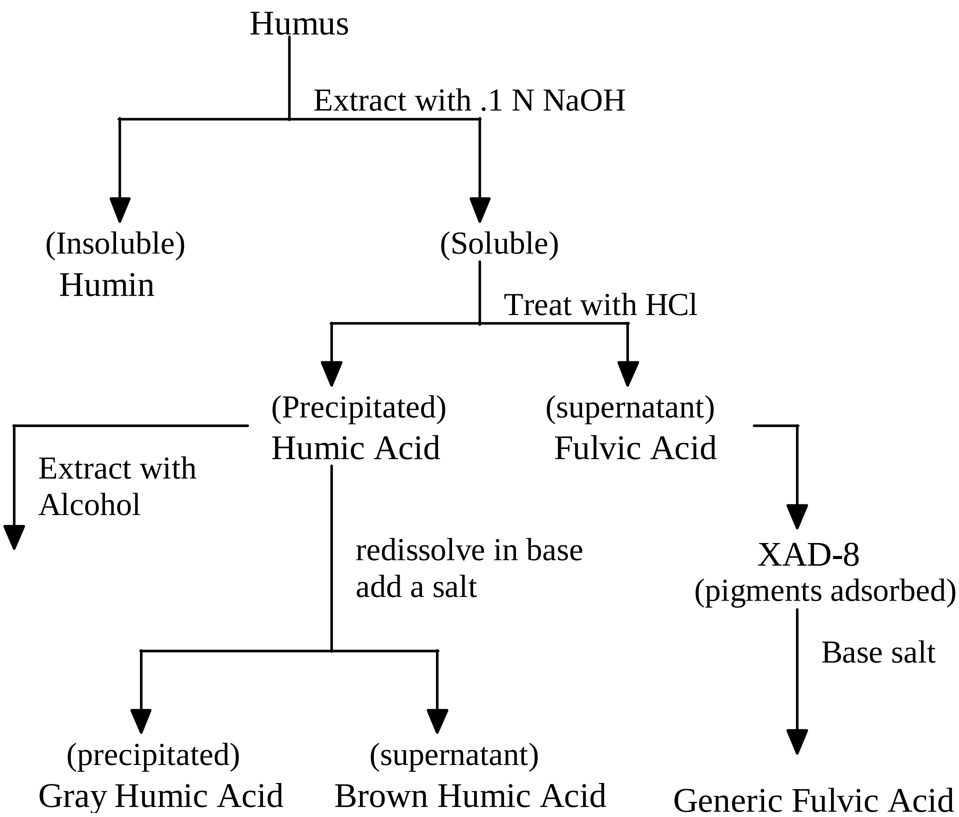

Figure II.2. Classical chemical fractionation of soil organic matter based on solubility.

As an alternative to chemical fractionation, physical fractionation procedures are used to recover the light fraction and partially decomposed products, to determine the role of different organic matter in formation of water-stable aggregates, and to establish the biological and environmental significance of organic matter in organo-mineral complexes (Stevenson, 1994). Elliot and Cambardella (1991) suggest that chemical extraction methods provide information on the type of organic matter present in the extracts, but physical separations provide information on where the organic matter is located within the soil matrix. Generally, physical fractionation is achieved by disruption or separation. Disruption involves three methods: sonication, shaking, and chemical. Methods used under separation techniques include dry and wet sieving, sedimentation, and density flotation.

Recent studies have attempted to physically isolate the non-humified soil organic matter components, the light fraction and macro-organic matter, according to particle size or specific density (Spycher et al., 1983; Strickland and Sollins, 1987; Wander et al., 1994; Bremer et al., 1995; Ellert and Gregorich, 1995; Meijboom et al., 1995; Barrios et al.,1996; Barrios et al.,1997). The light fraction has been recognized as an important substrate for biologically-mediated soil processes, and as an indicator of soil fertility. Christensen (1992) emphasizes that physical fractionation methods are chemically less destructive, than the classical solubility extraction with acid and base reagents, so the soils resemble their natural state and function.

Light Fraction and Macro-organic Matter

Terms or definitions given to fractionated soil organic matter pools have varied with researchers and methodology. Light and heavy fractions are commonly used for the two distinct soil organic pools obtained from densimetric techniques. Particulate (POM) and macro-organic matter (macro-OM) are terms used in particle size separations (Elliott and Cambardella, 1991; Gregorich and Ellert, 1993). The decomposition and carbon turnover rates of the light fraction, particulate, and macro-organic matter fractions range from 1 to

15 yr so the terms are rather synonymous regarding activity. However, the particulate and macro-organic matter fractions are closely associated with sand-size soil particles. Gregorich and Ellert (1993) explain that the reasons for physically fractionating soil organic matter is to separate non-humified material in order to categorize chemically the humified materials, to distinguish and measure the easily decomposable matter, and to study the processes of plant decomposition and their effects on soil fertility, organic matter persistence, and micro-organisms. Greenland and Ford (1964) termed the light fraction to be non-mineral materials, like fresh plant residues and faunal detritus with a rather high C:N ratio, with a specific density < 2.0 g cm^{-3}, and with a rather rapid turnover rate.

The light fraction is an important substrate of potentially mineralizable N. Janzen et al. (1992) found the light fraction highly correlated with soil respiration and microbial biomass. Some researchers have attempted to further divide the soil into a light, intermediate, and heavy fractions but actual interpretations are less tenable than for just light and heavy fractions (Christensen, 1992). The light fraction is particularly important to nutrient dynamics, Greenland and Ford (1964) found that up to 48 and 32 % of total soil organic C and N were isolated within this fraction. Less decomposable, more stable C sources that are not easily oxidized accumulate in the heavy fraction. This fraction functions as the primary sink for C (Boone, 1994). The most common methods for isolating these fractions involve wet sieving and density flotation.

The light fraction is normally derived by density flotation. With this procedure, organic materials, due to their comparatively light nature (< 1.5 g cm^{-3}), can be separated by mixing with heavy liquids having densities of ≈ 2.0 g cm^{-3}. Organic liquids that have been used in density flotation techniques, but are used less frequently due to the toxicity of halogenated hyrdocarbons and potential for contamination of carbon content, are tetrabromomethane ($C_2H_2Br_4$, 2.96 g cm^{-3}), bromoform ($CHBr_3$, 2.88 g cm^{-3}), and tetrachloromethane (CCl_4, 1.59 g cm^{-3}) (Pearson and Truog, 1937; Christensen, 1992). Most recently, aqueous solutions of inorganic salts are being used for this purpose. Inorganic salts being used are sodium iodide (NaI, 1.7 g cm^{-3}), magnesium sulfate ($MgSO_4$), zinc bromide ($ZnBr_2$), and sodium polytungstate (Strickland and Sollins, 1984; Gregorich and Ellert, 1993; Biederbeck et al., 1994; Boone, 1994; Bremer et al., 1994; Wander et al., 1994; Barrios et al., 1997). The most chemically inert substance used so far when compared to inorganic and organic heavy liquids is a stable silica-gel suspension (Meijboom et al., 1995; Barrios et al., 1997). The advantage of the silica suspension is the cost, non-toxicity of silica, and an easier separation time for organic materials, allowing larger samples (500 g) to be analyzed. Organic carbon content can be ascertained in the light fraction and whole soil by dry combustion techniques after the light and heavy fractions are separated and passed through a series of washing, centrifuging or sedimentation, filtering, and sieving (< 250 μm) (Gregorich and Ellert, 1993).

Christensen (1992) cites a study by Greenland and Ford (1964) where the light fraction comprised 0.03 to 8.2 % of soil mass and accounted for 1 to 85% and 11 to 100% of the whole soil C and N, respectively, of different Australian soils. Bremer et al. (1995) studied the effects and dynamics of changing cropping practices on total organic C concentrations at a depth of 0-5 cm and found that decomposition of the light fraction during fallow periods accounted for a 33 % decrease in total organic C content. In this study, the researchers concluded that light fraction C is a good indicator of management-induced changes due to its consistency and sensitivity to differences over time. In another study on the same site, Bremer et al. (1994) graded the sensitivity of total soil organic C, light fraction and mineralizable C to soil management impacts; and assigned light fraction the highest grade (2.5), mineralizable C at 1.5, and the lowest grade (0.2) for total organic C. In both of these studies, maintaining native grass and soil amendments enhanced carbon accumulation in the light fraction and the whole soil.

Boone (1994), in a study on the origin and contribution of light fraction soil organic matter under different vegetative covers, found that the light fraction of corn (*Zea mays*), white pine (*Pinus strobus*), and sugar maple (*Acer saccharum*) stands contributed 11, 13, and 2 %, respectively, to net N mineralization. The annual contribution of light fraction to N mineralization under pine and maple trees was estimated at 3 kg ha^{-1} yr^{-1}. However, net N mineralization of the light fraction of these two stands was not as high as the heavy fraction. On a dry weight basis, the light fraction under corn, white pine, and sugar maple comprised 13, 14, and 5 % of the whole soil organic matter.

Barrios et al. (1997), in an agroforestry study of several sub-Saharan cropping systems looking at how tree fallows can replenish soil fertility versus a continuous maize crop, found that the light fraction and its contribution to inorganic N and mineralization did vary with tree species. The researchers used a silica-gel suspension as their extractant and fractionated the macroorganic matter rather than the whole soil; that is, soil organic matter associated with sand-size particles (> 53 μm). This fraction was then isolated and divided into three specific density classes: light, intermediate, and heavy (< 1.13g cm^{-3}, <1.37g cm^{-3}, and > 1.37g cm^{-3}). However, in their final analyses the researchers determined that the light and intermediate classes should be pooled because the classes were not statistically different. The mean percent of C, N, and P this pooled fraction contributed to the whole soil for nine tree, fallow, and maize treatments were 2.9, 2.8, and 0.3 %, respectively. When the fraction (>1.37g cm^{-3}) was pooled along with the other classes, the percentages were slightly higher: 3.6, 3.3, and 0.4, respectively. Barrios et al. (1997) concluded that the quality of leaf litter affects inorganic N, aerobic and anaerobic mineralization, and the light fraction availability and accumulation. Amounts for these variables were higher where the lignin-polyphenol to nitrogen ratio was lowest.

Tiessen and Stewart (1983) used particle size fractionation to categorize soil organic matter loss during 4 yr of cultivation. Sand size particles >50 μm and associated floatable organic matter were defined to be particulate or macroorganic matter, with a high C/N ratio, low specific density, and readily decomposable. In their study, 43 % of initial SOM loss was from this size fraction through oxidation and redistribution in the coarse silt fraction. This redistribution of organic matter resulted in the mineralization of organic P into inorganic P signifying its role as a source of P. The fine silt fraction is generally considered to contain the highest organic matter content. However, in analyzing the fine clay fraction, the organic materials were characterized with a low C/N ratio and lability. Tiessen and Stewart (1983) concluded that the fine clay fraction and the macroorganic matter fraction play an important role in soil fertility. In a similar study addressing soil organic matter turnover in agricultural soils, Cambardella and Elliott (1992) discovered that 39 % of total soil C was comprised of macroorganic matter carbon. Under a native grass cover, POM-N represented 29 % of total soil N. In their analysis, they suggest that the POM fraction is similar to the light fraction due to these similar percentages and activity. Ellert and Gregorich (1995) state that although each of these fractions contain fresh plant residues, the equivalency of the two is uncertain. In contrasting the two fractions, macro-OM represented a larger proportion of soil mass than the light fraction due to the association with sand particles and generally has a higher ash content. According to Dalal and Mayer (1987), the light fraction may more accurately separate the labile from the non-labile organic matter. Christensen (1992) and Gregorich and Janzen (1996) recommend that, due to this uncertainty and similar characteristics, the two fractions be used as companion analyses in defining soil organic matter distribution.

Aggregate Stability

Aggregate stability is a soil quality indicator of how resilient and resistant a soil is to disruptive forces. Aggregation functions in promoting soil fertility, proper aeration, water availability, microbial activity, and plant productivity as root growth opportunities increase. Kemper and Rosenau (1986) define an aggregate as a group of primary particles that cohere to each other more strongly than to other surrounding soil particles. Soil organic matter enhances soil structure and tilth through the formation of micro- and macro-aggregates. Organic materials such as fungal hyphae, fine roots, root exudates, microbial gels, and bacterial polysaccharides serve as cohesive forces that bind microaggregates together into larger aggregates (Carter and Gregorich, 1997). During this process of aggregation, soil organic matter can become chemically adsorbed and complexed by polyvalent inorganic cations (Ca^{2+}, Mg^{2+}, Fe^{2+}, and Al^{3+}) in the clay fractions and/or physically occluded within micro-sites of soil aggregates and protected from oxidative reactions that lead to CO_2 efflux (Brady and Weil, 1996).

Recent studies in aggregate stability have attempted to locate and characterize actual turnover rates of soil organic matter within the two aggregate classes. Micro-aggregates are defined soil structures (50-250μm diam.) in which primary mineral particles are bound by chemical bonds and complexing of clays, polyvalent metals and humified soil organic matter (Beare et al., 1994). Micro-aggregates are associated with the older, more recalcitrant soil organic matter pool. Macro-aggregates (> 250μm diam.) are considered less resistant and composed of relatively decomposable materials, and therefore, more susceptible to disintegration under intensive management practices.

Soil aggregates can be disrupted by management practices like cultivation and forest site preparation that alter water infiltration into the solum or expose aggregate surfaces to other erosive forces. Soil amendments and the decomposition of soil organic matter can improve the water stability of soil aggregates (Tisdall and Oades, 1982; Chaney and Swift, 1986; Brady and Weil, 1996). Lindsay and Logan (1998) report that biosolids improve aggregate stability through the addition of fats, waxes, oils, resins, and water-soluble polysaccharides.

Soil organic matter can greatly increase the specific surface area of soil particles. Slaking and abrasion of aggregates results in soil organic matter being loosened from soil aggregates (Tisdall and Oades, 1982). Under normal conditions and adequate moisture, organic materials are slowly dissolved and nutrients mineralized from within aggregate microsites to increase soil fertility and nutrient availability. However, if the disruptive forces are greater than the aggregates' internal cohesive forces, soil structure can be degraded or compacted, crusting or puddling can occur at the soil surface, nutrients leached, and normal soil biological processes disturbed. Soil organic carbon is oxidized during this process and released in the form of atmospheric CO_2.

Aggregate stability is necessary to reduce and avoid erosion losses, but can vary under the influence of differing vegetation. Graham et al. (1995), in a study contrasting aggregate stability under oak and pine vegetation, reported that after 41 yr, aggregate stability in an A and Oi horizon, was approximately 15 % greater and the volume of aggregates seven times larger under scrub oak (*Quercus dumosa* Nutt.) than Coulter pine (*Pinus coulteri* B.Don). Organic carbon under oak vegetation was significantly different from pine (35 vs. 12.9 g kg^{-1}). They attribute these differences to the abundance of worm casts and fungal hyphae under oak vegetation. Comparing no-tillage and conventional practices, Beare et al. (1994) found that whole soil organic carbon was 18 % higher under no-till system compared to conventional tillage (30.7 vs. 26.1 Mg C ha^{-1}). Nitrogen contained in particulate organic matter was also higher under the no-till system. Macro-aggregates were fewer and less stable with conventional tillage. They attribute these differences to the importance of plant residue management on soil water availability and temperature and organic matter accumulation.

Stability differences can also be attributed to mineralogy, Levy and Miller (1997) compared the aggregate stability of 11 soils in Georgia and report that 6 of these were found to have low aggregate stability. They concluded that the variability in aggregate stability of these soils might be due to varying kaolinitic clay contents and that stability of soil may depend more on fine primary particles than on coarser size structural units. Generally, aggregate stability is positively correlated with organic matter content; as organic matter content declines, aggregate stability declines. Hence, Beare et al. (1994) reported that kaolinitic clays were more susceptible to dispersion and losses of soil organic matter.

Soil Aggregation and Global Climate Change

The dynamics of carbon sequestration and the role of soil organic matter storage are additional reasons for characterizing aggregate stability and aggregate size classes. Aggregate stability is integral for soil fertility and water infiltration, but aggregation is also the chief process for carbon capture. During aggregation, soil organic matter and carbonaceous material are physically occluded or chemically bound within aggregates. Recently, public concern about global climate change has been the impetus for significant research on the subject of CO_2 emissions and investigations into the sources and sinks of CO_2. The importance of soil organic carbon in agriculture and soil productivity has spawned other research efforts. Similarly, Powers et al. (1990) encouraged North American land and forest managers to assess current policies and practices and critically evaluate how higher latitude forests might modulate atmospheric CO_2 emissions. Coal mine operators and regulators of mined land reclamation should also evaluate how their current management and policies could be altered to more proactively contribute to regulation of CO_2 emissions, especially as environmental restrictions become more stringent for natural resource industries.

Houghton et al. (1983) researched the changes in carbon content that have occurred from 1860 to 1980 in different ecosystems. They concluded that there has been a significant net release of CO_2 to the atmosphere especially since 1958. In 1980, the estimated net release of carbon to the atmosphere was 1.8 - 4.7 x 10^{15} g and a total estimated release of 38 to 76 x 10^{15} g for the a 22 yr period from 1958 to 1980. Paul and Collins (1997) report that atmospheric concentrations of CO_2 have increased from 260 $\mu L\ L^{-1}$ to present 360 $\mu L\ L^{-1}$ over the same period. Prior to the study by Houghton et al. (1983), the null hypothesis for the cause of increased atmospheric carbon concentration was due to fossil fuel combustion and consequent release of CO_2. However, increased land use was a secondary cause as agricultural practices intensified and resulted in further deforestation and land clearing. As land is left exposed after these activities, a significant portion of soil carbon is oxidized: 20% for agriculture and up to 70% for deforested areas (Houghton et al., 1983). Although the percentage for coal mining or similar operations was not mentioned in the previous study, soil carbon oxidized during coal mining operations could approach or exceed the latter figure if carbon extracted in the form of coal was accounted for in a carbon budget. Reforestation and mined land reclamation may mitigate CO_2 losses from these disturbed sites as ecosystem stability and soil quality are restored.

Flach et al. (1997) report that due to the urgency of this issue and the potential for the U.S. agriculture sector to mitigate CO_2 emissions, the U.S. Environmental Protection Agency (EPA) sponsored an initiative to evaluate management and conservation efforts that might positively effect soil C sequestration in agroecosystems. A similar initiative has been undertaken for forest environments and is also warranted in mined land reclamation efforts. Within this framework, models for simulating organic matter turnover, C accretion and the potential for C sequestration were developed to forecast the changes in soil organic matter from 1990 to 2030.

Aggregation and particle size fractions are soil parameters used to define soil organic matter pools. These soil organic matter pools with varying decomposition rates are then applied to these models. The decomposition and turnover rates of plant residues tagged with isotopic carbon are traced in aggregates. Soil organic matter losses from aggregates are measured and followed as soils are exposed to disruptive forces and soil organic matter undergoes decomposition. E.A. Paul (1984) cites other studies where ^{14}C was used to trace the decay or half-life of various carbonaceous material within soil aggregates.

Nitrogen Mineralization

Nitrogen mineralization is a soil attribute that Gregorich et al. (1994) recommend as a biological indicator of soil organic matter quality and overall soil quality. Nitrogen is frequently the most limiting nutrient for crop production on mine soils (Roberts et al., 1988), so the nitrogen mineralization potential of a soil is an indicator of fertility and site productivity. Nitrogen is essential to plant physiology and the formation and efficacy of chlorophyll. Nitrogen is very mobile, elusive and easily leached in soil. N undergoes many changes that are facilitated by soil biological, physical, and chemical processes (Brady and Weil, 1996).

Gregorich et al. (1994) state that the mineralizable fraction of nutrients is integral for element cycling and plant availability. Readily mineralizable N comprises approximately 1/3 of total organic soil N (Barber, 1995). Nitrogen mineralization is the microbial decomposition of soil organic matter that mediates the transformation of N from an organic to inorganic form. This procedure is dynamic and there is a continual competition between microorganisms and plants for this essential nutrient as N is immobilized and mineralized within soil. In the organic form, N is immobilized and not available for plant uptake. Similarly, C content and quality can be implied from this process as immobilization normally occurs as C content increases creating a higher C/N ratio. Therefore, N mineralization is a relative index of soil organic matter's ability to supply plant available N as nitrate or ammonium. Anderson (1982) suggests that N mineralization derived during the incubation periods provides a sound bioassay of microbial activity and the potential for N.

Stanford and Smith (1972) showed that the amount of potentially mineralizable N and the rate at which it mineralizes (*k*) can be determined by aerobically incubating the soil under laboratory conditions and measuring the nitrogen mineralized. The following equation was derived by Stanford and Smith (1972) to describe the kinetics and change in mineralizable N:

$$[1] \qquad Nmin = N_o(1 - e^{-kt})$$

Burger and Pritchett (1984) in a study of the effects clearfelling and site preparation argue that laboratory aerobic incubation studies of potentially mineralizable nitrogen (N_o) are more accurate in assessing nitrogen availability than the C:N ratio. However, the results of potentially mineralizable Nincubation studies are not often correlated to field measurements. In an evaluation of laboratory incubations, Cabrera and Kissel (1988) report that Stanford and Smith's method overpredicted N mineralized 67 to 343 %. They attribute the overestimate to soil drying, sieving, and to high water content in incubated samples. Drinkwater et al. (1996) suggest that short-term aerobic incubations (14 to 28 d) may more accurately assess the total potentially mineralizable nitrogen pool and better represent nitrogen availability under field conditions. Polglase et al. (1992), in a study of the potential of organic matter to supply N and P, found that N mineralization under long-term laboratory incubation was little affected by weed control and fertilizer treatments. They report that understory vegetation may cycle more available N than pine.

Anaerobic incubations are also used, in conjunction with or as an alternative to aerobic procedures, to quantify mineralizable nitrogen (Keeney, 1982). Anaerobic incubation procedures are simple and rapid in comparison to aerobic incubations. Another advantage of this procedure is that all N mineralized is retained NH_4^+-N (Drinkwater et al., 1996). Anaerobic incubations are assumed to imitate the fumigation-incubation method of microbial biomass determinations. However, the correlation between the two procedures has been inconsistent for different soils (Drinkwater et al., 1996). Powers (1980) used this technique to diagnose N availability in forest stands. He suggests that anaerobic incubations should be used in conjunction with foliar analysis, and that it may serve as an alternative for foliar sampling for tall trees due to its effectiveness in predicting mineralizable N mineral soils at a 0 to 15-cm depth.

Organic amendments are commonly added to disturbed sites to provide a source of N and encourage mineralization for crop growth. Total organic N added in these amendments may be substantial. However, a high percentage of potentially mineralizable N can be mineralized within the first year after application and continue to decline through time resulting in a rather short-term (1 to 5 yr) benefit to the N cycle. Clapp et al. (1986) reported in an aerobic incubation study that 55% of organic N added in sewage sludge was mineralized the first year. In two other long-term incubation studies, N mineralization ranged from 14 to 61 % for anaerobically and aerobically-digested sludge. The ability of sewage sludge applications to supply organic N declines through time and often additional applications of N fertilizer are needed to sustain crop growth (Roberts, 1986). This was also confirmed by Clapp et al. (1986). In a California and Wisconsin study where sewage sludge was applied, the percent N mineralized declined linearly over the first years for each study: 35, 10, 6, 5 and 20, 15, 6, 4, 2 %, respectively. In a study of mine soil genesis, Schaffer et al. (1980) reported that, after reclamation, litter accumulation from initial vegetation can increase the C/N ratio in soils and reduce available N. This reduction in available N caused a reduction and stagnation of plant productivity. The long-term restoration of soil quality and permanent re-establishment of vegetation depends on the continual function of N mineralization as a soil mechanism. Due to the decay in mineralizable N through time, the N cycle should be monitored on sites that have received a one-time application of sewage sludge.

Agroforestry and Tree-based Systems

Agroforestry and other tree based systems have been recommended as alternative methods for improving land productivity and soil fertility of degraded lands (Gold and Hanover, 1987; Nair, 1993; Torbert and Burger, 1993). In an agroforestry system, a tree is integrated with an agricultural or forage crop in a spatial and temporal arrangement so that each crop provides products and services concurrently or concomitantly so that overall productivity of the system is enhanced. Nair (1993) reported that agricultural productivity can be improved with increased tree products, improved yield of associated crop, reduction of cropping system inputs, and increased labor efficiency.

Torbert and Burger (1993) advocate commercial forest land as a positive post-mining tree based system that can satisfy all parties (i.e. landowners, coal operators, and society) that have a stake in reclamation by providing the aesthetic and environmental services desired by society, potential for a viable long-term management plan for the land owner, and a less onerous reclamation requirement for the coal operator, without disregarding the intent of governing legislation.

In the southern U.S., forage crops have been integrated into forested landscapes to provide short-term services and products (Johnson and Davis, 1982). A similar system might be applicable in mineland reclamation. Tree and agricultural crops have different types, rates and times of litterfall and harvest so an agroforestry arrangement can help maintain or cycle soil organic matter. Organic litter and residue of the two crops serve to enhance soil carbon and condition the soil in various degrees throughout the year.

Knowledge of how to synchronize the decomposition of organic residues and the transformation of these materials for permanent or long-term retention of plant available nutrients remains a subject of research. Questions remain on how to manage or manipulate the beneficial attributes of organic matter and the decomposition process so soils that have been drastically disturbed or are inherently lacking in organic matter may become productive and resilient growth media.

Amelioration of degraded soils depends on the restoration of important soil mechanisms and functions. Fisher (1990) defined five mechanisms by which trees can ameliorate soils. These mechanisms are the main attributes of agroforestry systems and the major suppositions for use of tree based systems in restoration of degraded agro-ecosystems and marginal soils (Young, 1989; Nair, 1993). The five mechanisms that trees can affect are di-nitrogen (N_2) fixation through symbiotic relationships or actinorhizal growth in the root zone, efficient nutrient cycling by accessing ions deeper in the rooting zone, organic matter addition from litterfall, fine root mortality and root slough, micro-climate moderation, and rhizosphere interactions. If trees confer such ameliorative benefits on soils and provide interim services and commercial products, the potential for trees as a means of long-term reclamation and productivity may be more important than previously thought.

Vegetation Response to Organic Amendments

Use of Bioassays

Bioassays of vegetation response to land management are common indices applied to evaluate site productivity and estimate present soil quality. The limitations of bioassays such as site index, comparisons of wood volume across successive rotations, and crop yield performance standards has been reviewed (Burger, 1996; Doran et al., 1994; Jones, 1991; Van Lear, 1991). The reviewers argue that landscape, ecological, and soil factors must be considered to honestly classify potential site productivity and assess the impact intensive management practices will have on long term soil quality. Doran et al. (1994) explain that crop yield performance standards or bioassays must be used in conjunction with soil properties and processes to accurately assess soil quality and productivity. Similarly, Karlen et al. (1992) report that, due to the dynamic nature of soils, a collection of indicators is needed to evaluate changing levels of soil quality. Karlen et al. (1992) suggest that plant nutrient content or quality be used as a criterion of soil quality with soil physical, chemical, and biological indicators. Nutrient cycling is integrally linked to tree and plant production, biomass accumulation, and overall soil quality (Waring and Schlesinger, 1985; Switzer et al., 1968). Therefore, soil and foliar nutrient analysis procedures are commonly used to monitor soil fertility and plant nutrition.

The Surface Mining Control and Reclamation Act of 1977 (PL 95-87) requires that land be returned to its pre-mining productive capacity. The performance standard in this regulation is a short-term bioassay that assesses whether coal operators’ reclamation efforts complied and fulfilled the intent of the law. If the performance standard is met at the end of the 5-yr bond period, coal operators achieve bond release. A short-term success standard may be insufficient to assess trends in soil recovery and the stability of the vegetation system (Sopper, 1992). Initial productivity and 5 yr growth patterns may be followed by reductions in growth due to deficiencies in plant available N and P that would indicate the site is not self-sustaining. Even established vegetation can deteriorate if C and N cycles are not stabilized with balanced inputs and outputs (Seaker and Sopper, 1988). Many studies in forest fertilization and mine land reclamation show positive initial growth responses to organic amendments and fertilizers; there are fewer studies that have followed vegetative response after the fertilization regimen has been abandoned.

Tree Response to Amendments

Tree establishment and survival rates have been studied to determine optimum sludge applications and response to different types of organic amendments (Harris et al., 1984; Moss, 1986; Schoenholtz et al., 1992). Harris et al. (1984) report that aspen (*Populus* spp.) growth was 78 % greater in sludge-amended treatments than controls in the first growing season at applications of 4.8 to 46 Mg ha^{-1}. The sludge treatments increased understory vegetation 200 to 500 % in the same test. After five years, growth was a small fraction of the first year growth increment. In the same study looking at red pine (*Pinus resinosa* L.) response, no increase in girth (dbh) was detected after six growing seasons, but there was still a 50 % increase in ground cover density under the red pine canopy. Sopper and Kardos (1973) reported of the 5 yr growth response of coniferous and hardwood species planted on an old field irrigated with 5 cm of sewage effluent per week. Total tree height growth of Norway spruce (*Picea abies* L.), Austrain pine (*Pinus nigra*), pitch pine (*Pinus rigida*), white spruce (*Picea glauca*), red pine, white pine (*Pinus* *strobus*), Japanese larch (*Larix leptolepis*), European larch (*Larix decidua*) on irrigated sites ranged from 3.6 to 8.7 ft, whereas total heights for non-irrigated trees were between 1.6 and 2.8 ft. European larch, Japanese larch, and white pine had the greatest growth response to the sewage effluent irrigation: 8.7, 8.4, and 6.3 ft, respectively. A similar rate of effluent was applied to a hardwood forest and annual diameter growth for the 5 yr period was 0.22 and 0.13 in for the irrigated and control blocks.

Schoenholtz (1992) found that after 2 yr, ground line diameter and stem volume (d^2h) of pitch x loblolly hybrid pine trees growing in mine soils amended with wood-chips was 49 and 203 % greater than a control and 33 and 105 % compared to a topsoil treatment. Chaney et al. (1995) found that the growth of black walnut (*Juglans nigra* L.) and northern red oak (*Quercus rubra*) on reclaimed mind land was very slow, 10 cm yr^{-1}, 12 yr after reclamation. Survival for black walnut and northern red oak was 61 and 39 % where ground cover was controlled, and a dismal 2 and 0.2 % where ground cover was not controlled; survival declined rapidly after liming and inorganic fertilization and continued until age 12. Chaney et al. (1995) attributed the poor growth (10-cm yr^{-1}) and survival rate to weed competition and compaction.

Kost et al. (1997) evaluated the effects of sludge amendments on tree establishment of three hardwood species. Tree survival was 65 % when sludge was incorporated into the soil with a back-hoe to a depth of 0 to 60 cm. Sopper (1992) reports that the growth, vigor, and survival of four hardwood and one coniferous species were enhanced by high rates of organic matter even though municipal sewage sludge amendments (28 to 96 Mg ha^{-1}) contained high doses of copper and zinc. Overall, these studies show tree response and survival to be variable and dependent upon the application rate. The trees, generally, respond positively to increased organic matter content and available N.

Herbaceous Vegetation Response

Sopper (1992) reports tall fescue yields of 11 Mg ha^{-1} on sludge-amended (112 and 224 Mg ha^{-1}) mine spoils in West Virginia. Tall fescue production was 818% more than the control. After this high initial yield, the crop was harvested and production gradually declined on the site. The researchers maintain that on sites where the fescue was not harvested, annual yields continued to increase. Sopper (1992) also cited previous findings by Greibel et al. (1979) that concludes that rates $\geq$ 112-Mg ha^{-1} are required for higher yields and beneficial residual effects on nutrient cycling. An Illinois study of coal refuse material amended with anaerobically digested sludge (542 Mg ha^{-1}) and agricultural lime (89.6 Mg ha^{-1}), reported that dry matter yields of plots seeded with a mixture of tall fescue, alfalfa (*Medicago sativa* L.), and bromegrass (*Bromus inermis* L.) increased annually over a three year period and that the highest yield (6.0 Mg ha^{-1}) was observed the third year (Pietz et al., 1989). Seaker and Sopper (1988) studied organic matter accumulation and biomass yields 5 yr after sludge amendment at rates of 120 and 134 Mg ha^{-1} compared to inorganic fertilizer treatment only and determined biomass yields were more than double (16.6 $\pm$ 3.76 vs. 5.2 $\pm$ 0.9 Mg ha^{-1}). These results were similar to local maximum agronomic hay yields. In a study by Joost et al. (1987), four years after coal refuse was amended with varying rates of sewage sludge alone (225 to 900 Mg ha^{-1}), mixed with limestone (45 Mg ha^{-1}), and limestone only (45 to 180 Mg ha^{-1}), herbage yields for tall fescue, reed canarygrass (*Phalaris arundinacea* L.), and redtop (*Agrostis alba* L.) averaged 4 Mg ha^{-1} on all plots. In a greenhouse study, Stucky and Newman (1977) found yields of tall fescue and alfalfa significantly higher on amended agricultural soils compared to amended strip-mined soils. Yields increased with increasing application rates on both soils. Clapp et al. (1994), in a review of crop yields where sludge was annually applied for 20 yr, reports that the twenty-year average for corn grain and fodder was 8.4 and 17.4 Mg ha^{-1}. The yields of control plots were slightly less at 7.7 and 15.9 Mg ha^{-1}. On the same site with annual applications for 12 yr, average annual yields of reed canarygrass approached 11.0 and 9.6 Mg ha^{-1}. These studies, collectively, show the initial response of herbaceous vegetation to sewage sludge application to be positive, but biomass yields decline through time unless managed and harvested as an agronomic crop with annual amendment applications.

These studies of tree and ground cover response show that organic amendments are more beneficial than inorganic fertilizer in increasing organic matter and confer longer lasting fertility effects. However, plant responses were variable and may depend on climatic and other edaphic features such as water availability, extent of compaction, nutrient deficiency or toxicity, and level of soluble salts. However, even if organic amendments were widely available and cost-effectively applied, their long-term effect on mine soil recovery and plant system success is uncertain and appears variable.

SUMMARY

The recovery and rehabilitation of degraded or derelict lands such as surface mined sites requires knowledge of natural ecosystems and their physical, chemical and biological composition. Otherwise, reclamation efforts may be short-term and not fulfill the long-term goal of reconstructing a stable ecosystem and sustainable mine soil quality. Balanced nutrient cycling and self-sustaining vegetation are characteristic of stable ecosystems. Concerning soil quality, current research has focused on identifying indicators and composing minimum data sets for assessing soil quality. Key soil quality variables addressed in this review and relevant to sustainable mine soil quality were soil organic matter, mineralizable nitrogen, and aggregate stability. These variables enhance overall soil function and are essential to nutrient cycling and plant productivity. Addition of organic amendments has initially improved these soil variables but long-term improvement due to the amendment is less evident. Vegetation and tree growth response to organic amendments has generally been positive but this effect has varied with species and application rates. However, the literature shows that these soil variables and are important to ecosystem stability, indicative of nutrient cycling patterns, and influence the success of vegetation establishment and long-term sustainable mine soil quality. Soils play an important environmental and biological role in ecosystem function and sustainable land practices. Soils must be restored and maintained to meet reclamation goals and restore the vitality of previously disturbed sites.

CHAPTER III. METHODS AND MATERIALS

Study Site Description and Design

The experiment was initiated in the coal mining region of southwestern Virginia in Wise County on the Powell River Project (37 °00' N latitude and 82° 41' W longitude). Long-term mean annual precipitation and temperature for the region is 1150 mm and 11° C, respectively. The mine soils were classified as loamy-skeletal mixed, mesic Typic Udorthents (Roberts, 1986). In April and May 1982, researchers from the Crop and Soil Environmental Science (CSES) Department and Powell River Project personnel, uniformly applied a 1-m-deep base layer of 2:1 sandstone:siltstone overburden material over the study site prior to surface treatment applications. Experiment plots, 7m x 3.5m, were established in a randomized complete block design with four replications. Seven amendment treatments were applied consisting of a control constructed of strictly 2:1 sandstone:siltstone overburden material, 30cm native soil composed of a mixture of A, E, B, C, and Cr horizon material from neighboring forest soils, hardwood sawdust applied at 112 Mg ha^{-1}, and applications of aerobically digested municipal sewage sludge at the following rates 22, 56, 112, and 224 Mg ha^{-1}. Due to the detrimental effect the 224 Mg ha^{-1} sludge treatment had on initial tree seedling survival (Moss, 1986), and the fact that increasing the loading rate from 112 to 224 Mg ha^{-1} had no further effect on tall fescue yields (Daniels and Haering, 1994; Roberts et al., 1988), the 224 Mg ha^{-1} application rate was not analyzed in this study.

All plots were mulched with 900 kg ha^{-1} straw, hydroseeded with 170 kg ha^{-1} KY-31 tall fescue (*Festuca arundinacea* L. Schreb.) seed and 940 kg ha^{-1} paper fiber mulch.

Whole plots were split by vegetation type (tall fescue vs. pine trees) to allow for parallel studies on the establishment and productivity of tree and agronomic crops (Fig. III.1 and III.2). In April 1983, glyphosate (N-phosphonomethyl glycine) (Roundup 41S at 18.7 L ha^{-1}) was applied to sub-plots designated for tree planting. To maintain a pure tall fescue cover, selective herbicide applications were annually applied (1982-1985) to leguminous and broadleaved species. The tree sub-plots were originally planted to 12 pitch x loblolly pine hybrid (*Pinus rigida* x *taeda*) seedlings and 4 loblolly pine (*P. taeda*) seedlings. Initial macro-properties and elemental composition of the sewage sludge are summarized in Table III.1.

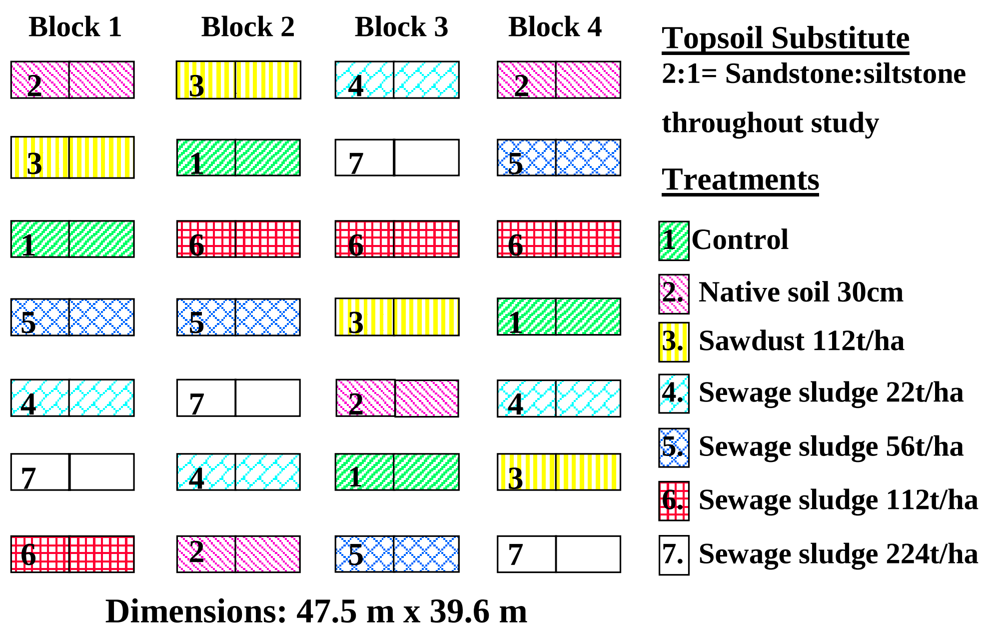

Figure III.1. A layout of the organic amendment study.

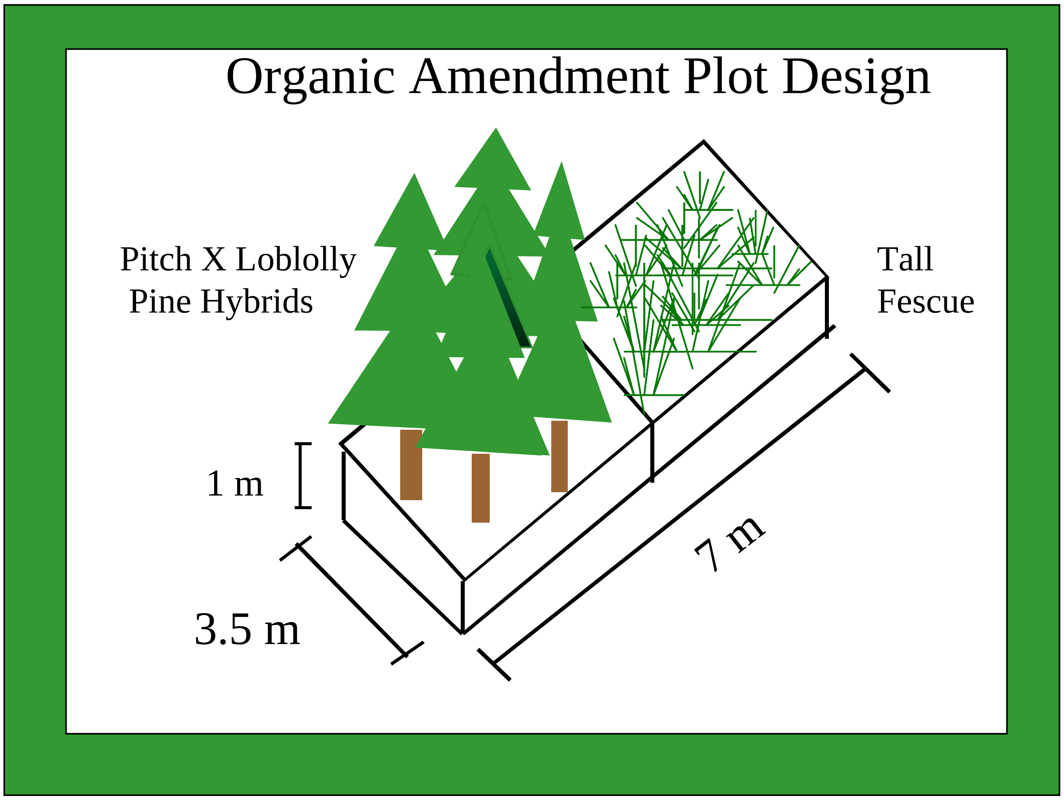

Figure III.2. Plot design and vegetation types.

Table III.1. Composition of sewage sludge applied as an organic amendment in May 1982.

Macroproperties		Microelements	($mg\ kg^{-1}$)
pH	6.8	B	32
NH_4^+ - N ($mg\ kg^{-1}$)	3000	Cd	6
NO_3^- - N ($mg\ kg^{-1}$)	24	Cu	245
Total N ($g\ kg^{-1}$)	26.0	Pb	175
Total P ($g\ kg^{-1}$)	6	Mn	320
Ca ($g\ kg^{-1}$)	37.0	Hg	3.5
Mg ($g\ kg^{-1}$)	3.8	Ni	80
K ($g\ kg^{-1}$)	8.8	Zn	880
Organic Matter ($g\ kg^{-1}$)	380		
Total Solids ($g\ kg^{-1}$)	450		

At the time of plant establishment, the control, native soil, and sawdust plots each received additional fertilization with N-P-K at rates of 168, 147, and 137 kg ha^{-1}. The native soil plots were also limed at a rate of 7.8 Mg ha^{-1} to bring the pH (4.4) to a level comparable to the overburden material. The sawdust plots received an additional 336 kg ha^{-1} of slow release N (Isobutyl Di-Urea; IBDU) fertilizer to offset potential nutrient immobilization caused by an initially high C:N ratio and increased microbial activity. Initial macronutrient (N-P-K) status of all treatments at time of study establishment is presented in Table III.2. Soil properties analyzed in this study are summarized in Table III.3. The partial ANOVA for a split-block design is presented in Table III.4.

Table III.2. Total N, P, and K added at study establishment as either chemical fertilizer or municipal sewage sludge.

Treatment	N	P	K
		Kg ha^{-1}	
1. Control (2:1 sandstone:siltstone)	168	147	137
2. 30cm Native soil + Lime (7.8 Mg/ha)	168	147	137
3. Sawdust (112 Mg/ha) + IBDU †	504	147	137
4. Sewage Sludge(22 Mg/ha)	582	29	56
5. Sewage Sludge (56 Mg/ha)	1455	74	140
6. Sewage Sludge (112 Mg/ha)	2910	147	280
7. Sewage Sludge (224 Mg/ha)	5820	295	560

† The sawdust treatment received an additional 336 kg N ha^{-1} as slow release Isobutyl Di-Urea (IBDU) fertilizer.

‡ Additional sources of organic matter that were uniformly applied to all treatments Include 900 kg ha^{-1} straw mulch and 940 kg ha^{-1} paper fiber mulch.

Table III.3. Summary of soil properties analyzed.

Biological	Chemical	Physical
Available Phosphorus	Effective Cation Exchange Capacity (ECEC)	Aggregate Stability
Foliar N	Electrical Conductivity	Available water at 33kPa
Foliar Nutrient Concentrations	Exchangeable Acidity	Bulk Density
Light Fraction Determination	pH in water	Coarse Fragments
Mineralizable Nitrogen		Micro- and Macro Porosity
Aerobic Method		Particle Size Analysis
Anaerobic Method		
TKN of Light fraction		
Total Carbon of light fraction		
Total Kjeldahl Nitrogen		
Total Organic Carbon		

Table III.4. Partial ANOVA for a split-block design.

Source	df	SS		EMS
Blocks	3	SSB	MSB	------
Treatments	6	SST	MST	$\sigma_\varepsilon^2 + 2\sigma_\delta^2 + 56\kappa^2_A$
WP Error 1	18	SSE1	MSE1	$\sigma_\varepsilon^2 + 2\sigma_\delta^2$
Vegetation	1	SSV	MSV	$\sigma_\varepsilon^2 + 7\sigma_\eta^2 + 28\kappa^2_B$
WP Error 2	3	SSE2	MSE2	$\sigma_\varepsilon^2 + 7\sigma_\eta^2$
Trt x Veg.	6	SSTV	MSTV	$\sigma_\varepsilon^2 + 4\kappa^2_{AB}$
Error 3	18	SSE3	MSE3	σ_ε^2
Total	55	SSY		

$$y_{ijk} = \mu + \rho_i + \alpha_j + \delta_{ij} + \beta_k + \eta_{ik} + (\alpha\beta)_{jk} + \varepsilon_{ijk}$$

For $i = 1,2,3,4$
$j = 1,2,\ldots,7$
$k = 1,2$

μ = overall mean, a constant
ρ_i = effect due to the *i*-th replicate
α_j = effect of the j-th level of factor A(Treatment)
δ_{ij} = error component of the j-th level of factor A(Treatment)
β_k = effect of the k-th level of B(Vegetation)
η_{ik} = error component of the k-th level of factor B(Vegetation)
$(\alpha\beta)_{jk}$ = interaction effect of the j-th level of A and the k-th level of B
ε_{ijk} = residual component, the error component of the AB interaction

CHAPTER IV. MINE SOIL QUALITY AND THE EFFECTS OF SEVERAL ORGANIC AMENDMENTS AFTER 16 YEARS

INTRODUCTION

In an effort to address the issue of sustainable land management, soil scientists and practitioners have tried to identify specific soil physical, chemical, and biological indicators that confer attributes of soil quality, fertility, and health (Doran and Parkin, 1996; Karlen and Stott, 1994; Larson and Pierce, 1994; Parr et al., 1992), that are integral to overall soil function. Soil quality has been defined as the capacity of soil to function, within ecosystem boundaries, to sustain biological productivity, maintain environmental quality, and promote plant, animal, and human health and habitation (SSSA, 1995; Doran and Parkin, 1994).

Much discussion and soil quality research has focused on intensive agriculture, its effect on soil quality, and soil management alternatives for sustainable agricultural systems (Doran and Parkin, 1994; Karlen et al., 1992; Parr et al., 1992; Larson and Pierce, 1991). Soil attributes proposed by these authors as basic indicators, or part of a minimum data set (MDS) of soil quality, are SOM, total organic N and C, aggregate stability, aeration, macroporosity, water holding capacity, microbial biomass, mineralizable C and N, bulk density, resistance to erosion, nutrient availability, pH, and electrical conductivity.

Soil quality is particularly relevant to surface mined and other disturbed sites where the capacity of soils has been dramatically altered or degraded and corrective actions are required. Soil erosion, compaction, acidification, and inadequate aeration and drainage are compounded and accelerated during mining operations. The Surface Mining Control and Reclamation Act of 1977 was passed to address these issues, and the dramatic decline in soil productivity and water quality associated with surface coal mining practices. Monitoring the dynamics of soil quality on reclaimed lands will enable and assist in the design of site-specific management and remediation practices necessary to restore specific soil functions and the hydrologic balance of these sites.

The long-term restoration of mine soil quality and ecosystem function of disturbed sites depends on reclamation that allows re-establishment of nutrient cycles so vegetation can thrive and be sustained in perpetuity. Bradshaw (1987) reiterates the importance of achieving a stable, self-sustaining vegetative cover. He points out that the reconstruction of an ecosystem and mine soil quality is dependent on vegetation for improving the physical and biological diversity of disturbed sites. A presupposition for establishing vegetation is a soil medium that allows for adequate growth and productivity.

Physical and chemical parameters that have been reported to limit soil productivity and plant growth on mine soils are soil acidity (Daniels and Amos, 1981), N and P availability (Daniels and Zipper, 1988; Roberts et al., 1988; Bradshaw, 1983; Marrs et al. 1983), micronutrient imbalance or toxicity, high electrical conductivity (Torbert et al., 1989), compaction and inadequate depth of rooting zone (Torbert et al., 1988; Daniels and Amos, 1984;), and low water holding capacity (Barnhisel, 1977). Low mine soil fertility is always a limitation. Bradshaw (1983) recommended 1000 kg ha^{-1} soil nitrogen as a minimum standard for proper plant growth and ecosystem development. Marrs et al.,

(1983) explain that N accumulation is very slow in new ecosystems and that total soil N capital should be 10 to 20 times the annual plant uptake.

Bradshaw (1983) further reports that P may be limited due to the high fixation capacities of exposed mine spoil composed of sandstone. This P binding potential of mine soils was clearly demonstrated in a study by Roberts et al. (1988). They reported a P deficiency in tall fescue due to the high adsorptive capacity of mine soil material.

Torbert et al. (1988), McFee et al. (1981), Krause (1973), and other researchers emphasize that properly identifying growth limiting factors of mined-land soil can assist in reclamation endeavors. Knowledge of growth limiting factors and characteristics of productive and stable ecosystems can serve as a minimum data set for evaluating mine soil quality. Seaker and Sopper (1988) proposed characteristics of productive reclaimed minesoils that are adversely affected by growth limiting factors; they include SOM accumulation and decomposition, organic C and N contents, root proliferation, and microbial activity. Other mechanisms that are integral to soil function and provide an amenable growth medium are N mineralization and aggregate stability.

In agriculture, crop residues, compost, manures, and synthetic fertilizers are traditional soil conditioners added to enhance soil function and quality. Most surface-mined sites lack organic matter necessary for plant establishment and productivity. Researchers have tested sawdust, municipal biosolids, fly ash, papermill sludge, and topsoil as potential SOM sources (Kost et al., 1997; Moss et al., 1989; Roberts et al., 1988; Epstein et al., 1978). The depauperate nature of mined lands requires the natural or amended addition of an organic mineralizable material to initiate nutrient cycling and overcome the chemical and physical limitations of these soils. Accordingly, we hypothesized that organic amendments (topsoil, sawdust, and different rates of municipal sewage sludge) could be used on surface mined lands to ameliorate disturbance, accelerate nutrient cycling, provide a receptive environment for vegetation, and improve overall soil quality and productivity. The specific objectives of this study were to determine the comparative ability of topsoil, sawdust, and sewage sludge treatments (i) to improve mine soil quality as measured by soil organic matter (SOM), aggregate stability, and mineralizable nitrogen and (ii) to determine the effect of these treatments on long-term changes in SOM content and related chemical and physical properties after 16 years, and (iii) to develop a predictive model of tree and herbaceous productivity as a function of key soil quality variables.

MATERIALS AND METHODS

Field Sampling

The effects of topsoil, sawdust, municipal sewage sludge, and control treatments on soil parameters were evaluated at different stages over a 16-yr period. These stages were (i) the first growing season after initial amendment; (ii) after three to five growing seasons; and (iii) 16 yr after initial amendment.

Composite soil samples, collected in 1982, 1987, and 1998, were composed of three randomly-located subsamples within each subplot. Subsamples were collected from the surface horizon at a depth of 0- to10-cm in 1998. Care was observed to avoid previous sampling sites by using a gridded map. Soils were air-dried and sieved to pass a 2-mm sieve to ascertain fine earth and coarse fragment contents. Subsamples were collected and oven dried at 105° C for 24 hr to determine gravimetric moisture content. In 1998, two bulk density cores were obtained from each subplot using a double-cylinder, hammer-driven core sampler. Bulk density was determined for both the fine earth fraction and whole soil. The fine earth fraction was determined by adjusting the total mass and volume of the cores based on the coarse fragment content and assuming a particle density of 2.65g cm^{-3}. Micro- and macro-porosity were determined using a tension table with a 50-cm tension. Available water at 33 kPa was obtained using a pressure plate (Klute, 1986). Bulk density core samples were oven dried at 105°C for 24 hr to obtain the unit mass per unit volume.

Soil Characterization and Laboratory Analysis

Soil biological, chemical and physical properties were characterized. Particle size analysis of the silt and clay fractions was performed by the hydrometer method (Gee and Bauder, 1986). Total organic C and SOM were estimated using the Walkley-Black wet oxidation method (Nelson and Sommers, 1982). A companion dry combustion analysis of total organic C was performed using the loss-on-ignition method (Sybron-Thermolyne Muffle Furnace, Inc.) at 550°C for 24 hr. For soil samples collected in 1998, light and heavy fraction SOM was determined using the modified density flotation procedure of Strickland and Sollins (1987) and Gregorich and Ellert (1993). Twenty-five grams of air-dried soil (< 2mm) was weighed into a 100-ml centrifuge tube and 50 ml sodium iodide (NaI) (density ≈1.7 g cm^{-3} obtained by adding 199 g NaI to 150 ml diH_20) liquid solution was added to the tube. The mixture was placed on a reciprocating shaker for 60 min.

After dispersion, the suspension was allowed to settle for 48 hr at room temperature (Strickland and Sollins, 1987). The suspended light fraction (LF) was aspirated off the top of the mixture through a Tygon vacuum hose with a vacuum pump and into a 500-ml vacuum flask. The supernatant containing the LF was decanted and gravimetrically filtered through Whatman no.50 qualitative filters. The LF collected on the filter unit was washed with at least 75 ml 0.01 M $CaCl_2$ to remove NaI from the LF. The 0.01 M $CaCl_2$ solution was followed by an additional wash of 75ml de-ionized H_20. The LF was washed from the filters onto pre-weighed drying tins and placed in an oven at 65°C to obtain the LF dry weight (Gregorich and Ellert, 1993).

After adding 25-30 ml NaI to the remaining suspension, the heavy fraction (HF) residue was re-suspended in the centrifuge tubes and the procedure was repeated to obtain a second aliquot of LF. The two aliquots of LF were combined and ground with a mortar and pestle to pass a 250μm sieve. A subsample of whole soil was sieved in a similar manner. Ash content was determined by placing an aliquot in a muffle furnace at 550° C for 4 hr. The oven dried samples (<250 μm) were analyzed for total organic C by dry combustion with a LECO C analyzer. Nitrogen was measured as NH_4^+-N colorimetrically on a Technicon Autoanalyzer II (Technicon Industrial systems, Tarytown, NY.) after a modified micro-Kjeldahl digestion (0.3g samples) (Strickland and Sollins, 1987).

Aggregate stability of the 1998 samples was determined using a wet sieving procedure (Kemper and Rosenau, 1986). Four-grams of air-dried 1 and 2 mm aggregates were placed in sample cups with 70-μm screen bottoms and slowly wetted to approximate -33 kPa water potential capacity by aerosol misting. The samples were wet-sieved in the cups using de-ionized water, a stroke length of 1.3 cm, and a frequency of 35 cycles min^{-1} for 3 min. Unstable aggregates passing through the sieve were oven dried at 105°C and weighed. Stable aggregates retained on the sieve were then completely dispersed with 2-g NaOH L^{-1} solution and resieved. All residues were oven-dried at 105°C and weighed. The stable fraction was determined and calculated on a sand-free basis.

Nitrogen mineralization potential was estimated in 1985 and 1998 using an aerobic incubation procedure that was outlined by Stanford and Smith (1972) and was later modified by Burger and Pritchett (1984). A 70-g, sieved field-moist subsample (< 2 mm) of the composite samples was mixed with 185 g acid-washed quartz sand to enable adequate aeration and packed in a polyvinyl chloride leaching column. The leaching columns were incubated under ideal moisture and temperature conditions to encourage microbial activity at 70 % field capacity and 35° C for 16 wk. Available mineral nitrogen was leached from the columns with 0.01 M $CaCl_2$ to a tension of -33 kPa at 1, 2, 3, 4, 6, 8, 10, 12, 14, and 16 weeks. After each leaching, the columns were bathed in 40ml nutrient-N solution. Leachate was weighed and analyzed for NO_3^--N and NH_4^+-N colorimetrically using a Technicon Autoanalyzer II (Technicon Industrial Systems, Inc.).

Inorganic plant available NH_4^+-N was evaluated using an anaerobic mineralization incubation procedure (Keeney, 1982). A 5-g subsample was anaerobically incubated in a sealed vial at 40°C for 7 days. After the incubation period, the contents of the vial were quantitatively transferred to a 150ml flask with 3 M KCl. The filtrate was spectroscopically determined with a Technicon Auto-analyzer II. Inorganic forms of nitrogen, NO_3^--N and NH_4^+-N, were extracted with a 2M KCl solution (Bremner, 1965) and analyzed as the aerobic mineralization procedure. Total N was determined with a modified micro-Kjeldahl digestion procedure (Bremner and Mulvaney, 1982), and ammonium was analyzed colormetrically for NH_4-N with a Technicon Autoanalyzer II (Technicon Industrial Systems, Tarytown, NY.).

Available P was extracted with 0.5 M $NaHCO_3^-$ adjusted to pH 8.5. Extractable P was determined using the modified Murphy-Riley ascorbic acid procedure and analyzed by spectrophotometry (Olsen and Sommers, 1982). Exchangeable cations (Ca^{2+}, Mg^{2+}, K^+, Na^+) were extracted with 1 M NH_4^+OAc solution, buffered to pH 7, and concentrations determined by inductively coupled plasma spectrometry (ICP). Soil reaction was determined with a pH electrode in a 2:1 water:soil extract. Exchangeable acidity was determined using a 1 N KCl replacing solution and titration to a phenolphthalien end point with a Mettler Dl2 auto-titrator (Mettler Instruments, Inc., Hightstown, NJ.). Electrical conductivity (EC) was measured with a conductivity meter in a 5:1 water:soil extract and standardized to 0.01 M KCl reference solution (Rhoades, 1982).

Vegetation Sampling and Analyses

In July 1997, tree survival was determined for each treatment. Tree heights and diameters were measured to obtain a relative volume index of individual tree growth, basal area, and overall plot productivity. Heights of all the trees in the plot were measured to the nearest centimeter. Tree diameters at breast height (dbh ≈1.37 m above ground level) were measured to the nearest tenth of a centimeter. Trees were classified according to crown class: dominant, co-dominant, and intermediate. Suppressed trees were not included.

In October 1997, standing biomass of herbaceous vegetation was measured. Each 3.5m x 3.5m plot was gridded into 100 0.1 m^2 quadrats. Using a random numbers table, three 0.1m^2 clip plots were randomly located and aboveground biomass was clipped to ground level. Aboveground biomass was categorized into three vegetation classes (grass, legumes, and forbs) and separated within each plot. Composite samples for each cover class were obtained for each plot, dried at 65°C, and weighed. Cumulative biomass for each plot was then calculated. Plant tissue samples were analyzed as described for pine needle samples.

Statistical Design and Data Analysis

The study design was a randomized complete block design with seven treatments and four replicates. Due to the detrimental effect the 224 Mg ha^{-1} sludge treatment had on initial tree seedling survival (Moss, 1986), and the fact that increasing the loading rate from 112 to 224 Mg ha^{-1} had no further effect on tall fescue yields (Daniels and Haering, 1994; Roberts et al., 1988), this highest sludge application rate was not analyzed in this study. In 1998, soil data and treatment effects were analyzed as a split-block design with treatment means of the main effect pooled across vegetation types (herbaceous vegetation vs. pine trees). A general linear model procedure was used for an analysis of variance and determination of the effects of amendments on mine soil parameters (SAS, Cary, NC, 1993). When significant F-values were obtained ($P < 0.10$), Duncan's multiple range test was used to separate the mean responses of the treatments: control, topsoil, sawdust, and different levels of sewage sludge. A non-linear procedure (PROC NLIN) was used to determine potentially mineralizable nitrogen (N_o) and the rate constant (k) for each treatment.

Dependent and independent variables were analyzed for significant correlations using PROC CORR procedure, and Pearson's correlation coefficients were computed. Multiple linear regression analysis was used to test key soil variables' (soil organic matter content, aggregate stability, and nitrogen mineralization potential) ability to predict tree volume per plot and herbaceous biomass production. A stepwise selection procedure was used within the multiple linear regression analysis to determine the best model by optimizing R^2, mean square error (MSE), and the sum of squares of all predicted error.

RESULTS AND DISCUSSION

Soil Quality Indicators

Soil Organic Matter

In 1982, soil organic matter content of the amended mine soils was commensurate with the amount of SOM added as part of the treatment (Figure IV.1). Sawdust-treated plots contained the highest percentage of SOM. The percent SOM for the sewage sludge treatments increased linearly with increased application rates. The control and native-soil-treated plots contained comparatively low SOM levels and were lower than levels typical of forest soils (Pritchett and Fisher, 1987). The low level on the native-soil-treated plots may have been a dilution effect due to mixing of the A, E, B, C, and C_r soil horizons and incorporation of this material with the minespoil. After five growing seasons, SOM levels of all treatments (excluding sawdust) increased. The sawdust treatment remained rather constant over the 5-yr period. SOM levels for the control, topsoil, and sludge (22 and 56 Mg ha^{-1}) more than doubled during this time. The 56 Mg ha^{-1} sludge-treated plots experienced the greatest increase in SOM concentration, increasing from 3.2 to 7.9%. The incremental change in SOM levels during this time suggests increased SOM inputs by the vegetation and increased cycling through decomposition. Tall fescue productivity and yields were highest during the first 4 yr of the study (Roberts et al., 1988) and corresponded with high SOM levels, but after the 1986 growing season, biomass productivity began to decline.

In 1998, 16 years after initial amendment, SOM concentration levels in the fine earth fraction for all treatments (except the native-soil-treated plots) appear to have equilibrated at around 4.3 %. (Fig.IV.1). Clapp et al.(1986) reported similar declines 4 yr after sewage sludge amendments were applied. Unlike the other treatments, the native soil-treated plots contained only 2.8% SOM. However, the native soil-treated plots had twice as much fine earth as the other treatments (68 vs. 35%), hence the soil organic matter is spread through more soil. Due to the difference in fine earth content and a higher bulk density, total soil organic matter content was slightly higher on the native-soil-treated plots. By 1998, soil organic matter content across treatments leveled off at about 10000 kg ha^{-1} (Fig.IV.1); no treatment had significantly more or less than the control. Organic matter inputs by vegetation alone over the 16-yr period in the control.

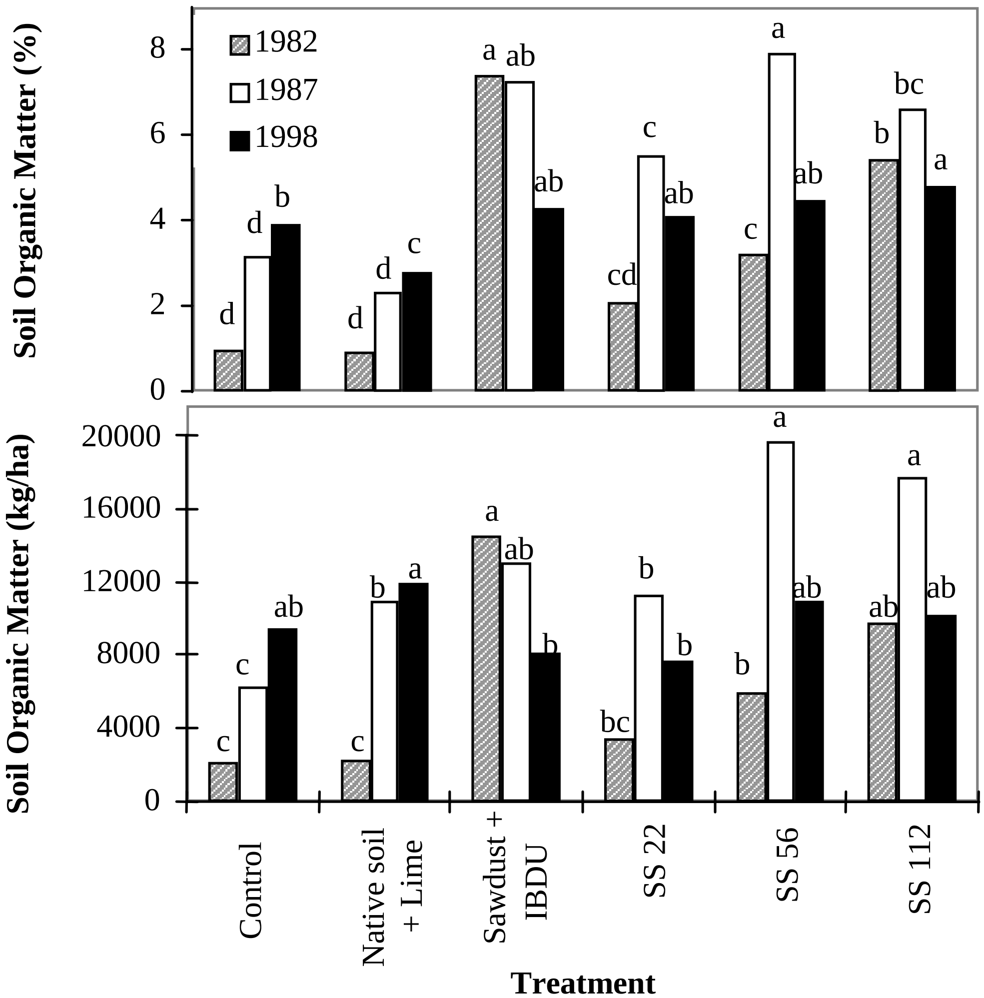

Figure IV.1. Soil organic matter concentration (%) and content (kg ha-1) of organically-amended mine soils during 1982, 1987, and 1998. Means within years followed by the same letter were not significantly different.

plots resulted in organic matter levels comparable to levels in the organically-amended plots, indicating the importance of vegetation in the soil recovery process, and the relatively rapid rate of accumulation in new mine soils

The overall decline (7.4 to 4.3%) in SOM concentration of the sawdust-amended plots over the 16 years may be attributed to mineralization of the high level of biologicallyreactive SOM initially applied. The increase in soil organic matter content during the first 5 years on the sludge-treated plots is mostly due to the high initial biomass productivity and vegetation inputs. The decline in soil organic matter content corresponds with the decline in productivity after the fourth growing season. The decline in soil organic matter content from 1987 to 1998 for the sewage-sludge treated plots suggests that the SOM inputs from vegetation were easily decomposable and were oxidized by biologically-mediated processes through time. In 1998, the distribution of soil organic matter in the light fraction and heavy fraction (Table IV.1) suggests that further decomposition and stabilization occurred during the past ten years, and that much of the organic matter present is more recalcitrant in composition. Mostly stable, recalcitrant organic matter was added to the other treatments at levels below or near what appears to be the equilibrium SOM level consistent with the climate and the vegetation growing on these mine soils. It appears that around 100 Mg ha^{-1} sludge or sawdust or equivalent material must be added during reclamation to achieve initial organic matter contents that will finally be achieved by vegetation inputs alone.

After 16 years, the light fraction organic matter (LF) in the 1998 samples showed that this fraction accounted for 1.9 to 2.9% of the whole soil on a dry weight basis (Table IV.1). These light fraction estimates are comparable to levels reported by Christensen (1992). The light fraction contained a significant proportion of whole soil organic matter, but the proportion was not significantly different among treatments. The heavy fraction, composed of the more physically and chemically recalcitrant organic matter, comprised 32 to 49% of total organic matter content (Table IV.1). The amount of organic carbon in the light fraction corresponded with whole soil organic carbon levels. Overall, light fraction organic matter was more concentrated in the control, sawdust and sludge-treated plots than in the native soil-treated plots.

We hypothesized that the SOM light fraction heavily influences soil quality indicators and so may be a good indicator itself, and we hypothesized that the LF as a proportion of whole-soil OM content would be different among treatments. Indeed, LF was positively correlated with N mineralization potential (N_o), anaerobically mineralizable N (Nmin), aggregate stability, and total porosity (TP) (Table IV.2) which are commonly found in minimum data sets of soil quality. However, the ratio of LF to SOM was similar among treatments (Table IV.1). LF:SOM ratios were not compared earlier in this study, but even if they were different, after 16 years of plant input and carbon cycling, there is little difference in the physical nature of SOM.

Table IV.1. Soil organic matter, light fraction organic matter composition, heavy fraction organic matter, and aggregate stability after 16 years.

Treatment	Soil Organic Matter (SOM)	Light Fraction OM	Organic Carbon	Light Fraction Carbon	Heavy Fraction OM	Aggregate Stability
	% of Whole Soil				(% of SOM)	%
Control	3.9 b	2.0 bc*	2.3 b	0.7 ns	49	57 ab
Native soil + lime	2.8 c	1.9 c	1.5 c	0.5	32	52 b
Sawdust + IBDU †	4.3 ab	2.8 a	2.5 ab	0.9	35	56 ab
Sludge (22 Mg/ha)	4.1 ab	2.2 abc	2.4 ab	0.6	46	61 a
Sludge (56 Mg/ha)	4.5 ab	2.7 ab	2.7 ab	0.9	40	62 ab
Sludge (112 Mg/ha)	4.8 a	2.9 a	2.9 a	1.0	40	65 a

* Means within columns followed by the same letter were not significantly different ($P < 0.10$).

† IBDU = Isobutyl Di-urea applied as a source of slow release of nitrogen

Aggregate Stability after 16 Years

Aggregate stability of the 1998 samples varied slightly among treatments; it ranged from 52 to 65% (Table IV.1), differences that, if significant, could influence soil quality.

Table IV.2. Correlation coefficients (r) of selected chemical and physical properties of treated mine soils 16 years after amendment (n= 48).

	Organic Matter (OM)	Light Fraction (LF)	Heavy Fraction (HF)	Aggregate Stability (AgStab)	Nitrogen Mineralization Potential (N_o)	Anaerobic N (Nmin)	Total Kjeldahl Nitrogen (TKN)	Phosphorus (P)	PH	Cation Exchange Capacity (ECEC)	Bulk Density (BD)	Total Porosity (TP)
OM	1.0000	0.2184	0.6819*	0.0810	0.2689*	0.2251	0.4286*	0.3024*	– 0.3465*	0.1747	– 0.2153	0.2676*
LF	0.2184	1.0000	– 0.2449*	0.2656*	0.3104*	0.3440*	0.2350	0.2248	0.0610	0.1507	– 0.2028	0.2741*
HF	0.6819*	– 0.2449*	1.0000	– 0.0422	0.1313	0.1454	0.3710*	0.2471*	– 0.2167	0.1884	– 0.0033	– 0.0150
AgStab	0.0810	0.2656*	– 0.0422	1.0000	0.5923*	0.3266*	0.1929	0.2927*	– 0.0420	0.3007*	– 0.2185	0.2432*
N_o	0.2690*	0.3104*	0.1313	0.5923*	1.0000	0.2811*	0.4787*	0.4812*	– 0.1997	0.4987*	– 0.1604	0.2605*
Nmin	0.2251	0.3440*	0.1454	0.3266*	0.2811*	1.0000	0.2935*	0.3289*	0.1004	0.3068*	– 0.2867*	0.2583*
TKN	0.4286*	0.2350	0.3710*	0.1929	0.4787*	0.2935*	1.0000	0.3571*	0.0881	0.5004*	– 0.0361	0.0872
pH	– 0.3465*	0.0610	– 0.2167	0.0420	– 0.1997	0.1004	0.0881	0.0082	1.0000	0.1465	0.3898*	– 0.3367
ECEC	0.1747	0.1507	0.1884	0.3007*	0.4987*	0.3068*	0.5004*	0.5192*	0.1465	1.0000	0.0091	0.0549
BD	– 0.2153	– 0.2028	– 0.0033	– 0.2185	– 0.1604	– 0.2867*	– 0.0361	– 0.0534	0.3898*	0.0091	1.0000	– 0.8760*
TP	0.2676*	0.2741*	– 0.0150	0.2432*	0.2605*	0.2583*	0.0872	0.1894	– 0.3369*	0.0549	– 0.8670*	1.0000
P	0.3024*	0.2248	0.2471*	0.2927*	0.4812*	0.3289*	0.3571*	1.0000	0.0082	0.5192*	– 0.0533	0.1894

***** Denotes significance at $p \leq 0.10$

The sewage-sludge-treated plots had the highest levels of stable aggregates, and the native soil-treated plots had the lowest amount of water stable aggregates, probably due to the dilution effect of fewer coarse fragments. Doubling the application of sewage sludge from 56 to 112 Mg ha^{-1} did not significantly increase aggregate stability. Aggregate stability of the treated mine soils was not significantly different from the control, indicating that natural organic inputs form vegetation alone might be more important in the formation of aggregates than any amendments used in this study. Aggregate stability corresponded with current levels of LF organic matter showing that it is a function of soil development through time rather than the type or amount of amendment.

Aggregate stability was significantly correlated with LF ($r = 0.27$), but was not correlated with SOM (Table IV.2). As aggregate stability increased, nitrogen mineralization, ECEC, total porosity, and extractable P increased. However, differences were not significantly influenced by treatment. It appears that natural plant inputs of organic matter and carbon cycling override amendment treatment effect after 16 years.

Mineralizable Soil Nitrogen

Total Kjeldahl N was measured to assess the accumulation of N through time. Organically-amended plots still contained more total N than the control and native-soil treated plots 5 yr after application. TKN levels in the control and native soil treatments tripled by 1998 to levels similar to that of the two highest sludge treatments. Soil N content was moderately correlated with organic matter content ($r = 0.4286$) (Table IV.2).

The mean TKN concentration for the various organic amendments was 1.57g kg^{-1}. Nitrogen accretion occurred on the control and native-soil-treated plots with accumulations of 680 and 920 kg ha^{-1}, respectively. The decline in total N in the 112 Mg ha^{-1} sludge treatment plots, and subsequent increase in the unamended control plots suggests that the system may be equilibrating at levels between 500 and 900kg ha^{-1} (Figure IV.2). These values are below the N level recommended by Bradshaw (1983), which suggests N may still be a limiting factor in this system. It appears that about 15 years is needed for climate, moisture availability, and other edaphic features to have the same influence on overall organic matter decomposition, N accretion, and system equilibrium as a one-time 100 Mg ha^{-1} application of organic amendment.

Nitrogen is usually deficient in mine soils and limits vegetation establishment and sustained productivity. Organic amendments are used as a source of mineralizable material to enhance N levels and extend N availability through cycling. Moss (1986) analyzed the nitrogen content of the amended mine soils after three growing seasons. Nitrogen data for 1985 and 1998 are presented here to show the comparative ability of the various organic amendments to affect long-term N availability (Table IV.3). N mineralization potential (N_o) was determined by a laboratory aerobic incubation procedure. In 1985, N_o for all treatments ranged from 6 to 125mg kg^{-1} and was well correlated with soil organic matter ($r^2 = 0.80$; $p > 0.0001$) (Moss, 1986). The nitrogen mineralization potential of control and native-soil-treated plots were significantly lower than the organically-amended plots.

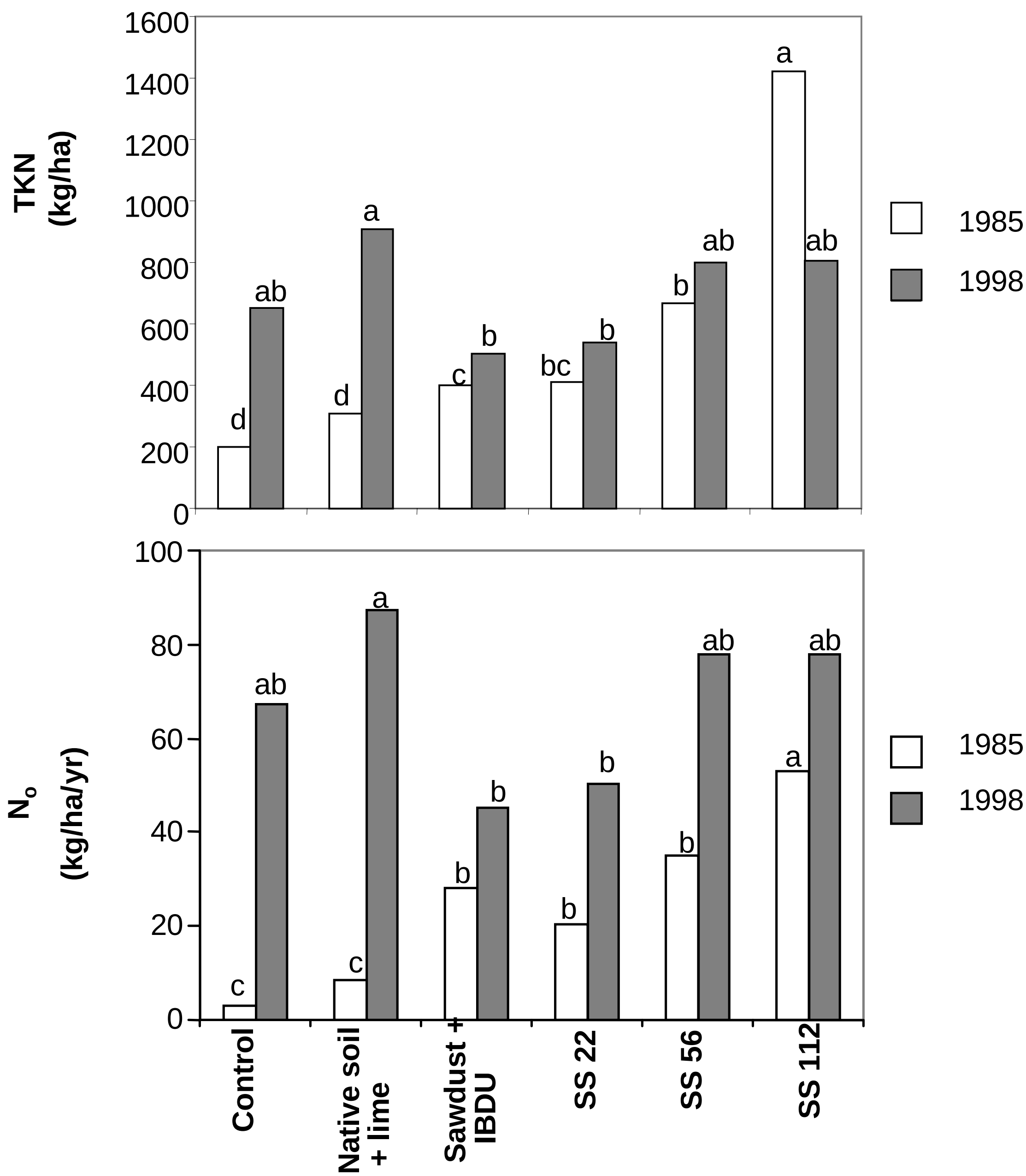

Figure IV.2. Total soil N and mineralizable N on amended mine soil in 1985 and 1998.

Table IV.3. Total N, nitrogen mineralization (No), rate constant (k), anaerobically mineralizable N (Nmin), KCl extractable nitrate and ammonium, and light fraction N in amended mine soils 3 and 16 years after reclamation.

Year	Treatment	TKN	Light Fraction-N §		No	No	Rate Constant	Nmin	NO_3-N	NH_4-N
		g/kg	g/kg		mg/kg	% of TKN	k	——— mg/kg ———		
1985	Control	0.50 d	N/A	N/A	6 c	1.2	0.07 bc	10 d	3.8 cd	1.9 a
	Native soil + Lime	0.33 d	———	———	10 c	3.0	0.09 ab	12 d	3.2 d	1.5 ab
	Sawdust + IBDU	1.12 bc	———	———	74 b	6.6	0.10 a	62 ab	5.4 bc	1.5 ab
	SS 22	1.00 c	———	———	54 b	5.4	0.06 c	28 c	4.9 bcd	0.9 b
	SS 56	1.34 b	———	———	72 b	5.4	0.05 c	38 bc	6.7 b	1.1 ab
	SS 112	2.21 a	———	———	125 a	5.7	0.07 bc	79 a	18.6 a	0.8 b
1998	Control	1.35 bc*	0.23 a		139 bc	11.5 a	0.33 ab	60 bc	2.9 ns	6.7 ab
	Native soil + Lime	1.06 c	0.14 ab		102 d	10.6 ab	0.36 a	45 c	2.4	6.1 b
	Sawdust + IBDU	1.32 bc	0.16 ab		119 cd	8.9 c	0.29 abc	52 bc	3.0	7.5 ab
	SS 22	1.43 bb	0.10 b		135 bcd	9.5 bc	0.27 bcd	63 ab	2.5	7.2 ab
	SS 56	1.65 ab	0.15 b		160 ab	9.6 bc	0.24 cd	64 ab	3.0	8.3 a
	SS 112	1.90 a	0.15 b		183 a	10.8 ab	0.20 d	81 a	3.1	8.5 a

* Means within columns followed by the same letter were not significantly different (alpha = 0.10).

Sludge application rates were Mg/ha.

† IBDU = Isobutyl Di-urea applied as a source of slow release nitrogen

‡ ns = not significant

§ Light fraction of the 1998 soil samples was determined using a modified density flotation procedure of Strickland and Sollins (1987) and Gregorich and Ellert (1993), nitrogen content of the light fraction was determined using a micro-Kjeldahl procedure. Light fraction data were not available for 1985 samples.

Thirteen years later, N_o of all treatments increased. Mineralizable N on the control and native-soil treatments increased by about 20 and 10 fold, respectively, while the other treatments increased by 1.5 to 2.5 fold.

After 16 years, mineralizable N comprised about 10% of TKN; the control, native soil, and SS 112 had the highest values. The rate constant (k) for N mineralization increased slightly after 16 years. Linear regression analysis of N_o as a function of SOM, LF, and total Kjeldahl N across treatments showed a strong treatment relationship. However, the linear regession and the R^2 values of LF and TKN suggest that N_o was derived from more recalcitrant organic material.

The light fraction was also analyzed for organic N content (Table IV.3). Although sludge treatments contained the highest level of total soil N, the treatment contained less N in the light fraction due to a greater level of decomposition. This suggests that a greater amount of the total N in the sludge-treated plots resides in the heavy fraction (specific density > 1.70g cm^3) of SOM. However, light fraction organic matter (LFOM) was more weakly correlated than TKN with N_o (Fig. IV.3), showing that LFOM has less influence on N_o than hypothesized.

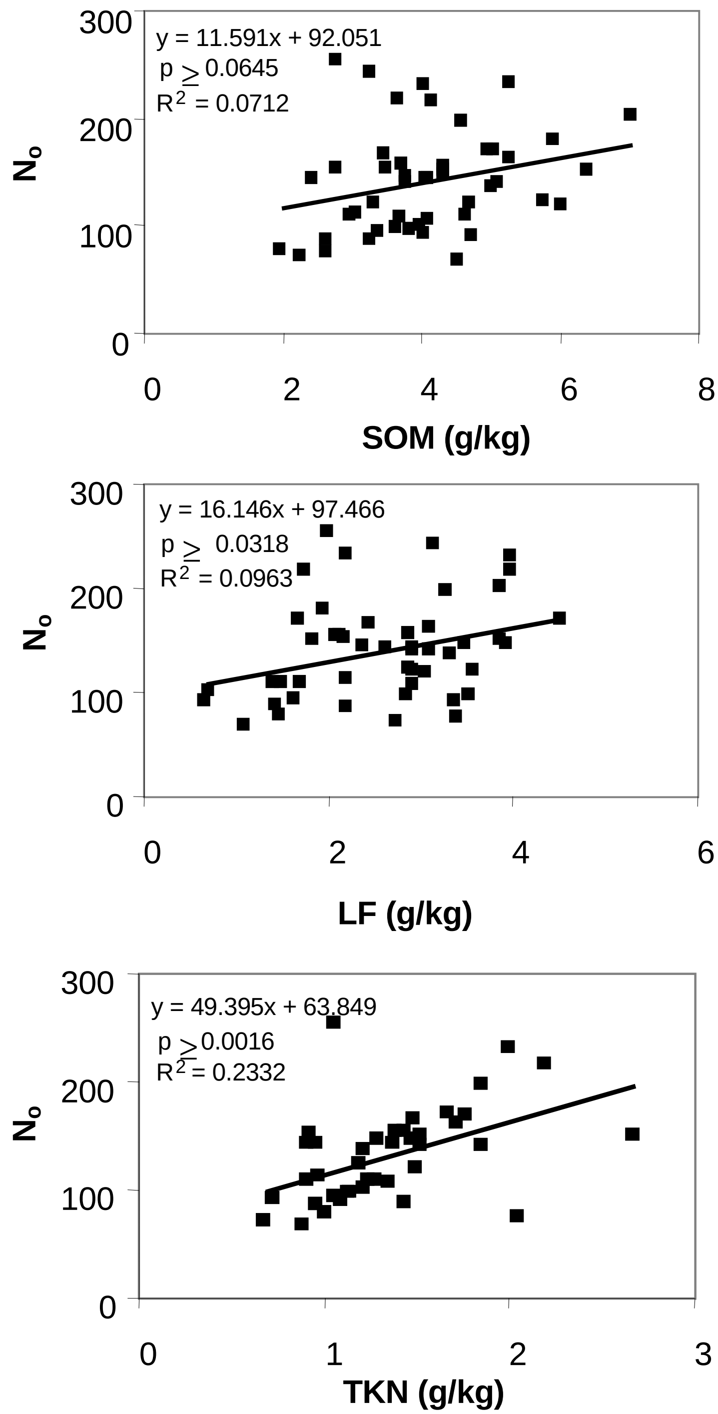

Figure IV.3. The relationship between aerobically mineralizable nitrogen, SOM, LF , and total Kjeldahl nitrogen on amended mine soils.

Potassium chloride (KCl) extractable inorganic nitrate (NO_3^--N) and ammonium (NH_4^+ -N) values for 1985 and 1998 varied. In 1985, NO_3^-N levels were considerably higher than NH_4^+-N levels. However, in 1998, ammonium levels are higher than nitrate levels. The difference in nitrate and ammonium content may be due to different sampling times and nitrification rates. Soil sampling occurred in October 1985 and March 1998. The predominance of inorganic ammonium in the samples of spring 1998 indicate rather fresh N inputs from litter and root decomposition, but also that temperature and moisture conditions may not have been optimum for nitrification. Weathering and hydrated conditions within the lattice structure of the siltstone exchange complex may have contributed to the release of NH_4^+ that was previously geologically fixed, but organic substrates are more probable sources of the increased ammonium. Overall, KCl extractable N (NO_3^- -N + NH_4^+-N) levels increased across all treatments with the exception of the 112Mg ha^{-1} sludge. The nitrate level for the 112Mg ha^{-1} sludge plots was significantly higher than the other treatments after three growing seasons, but in 1998, it was the same as the NO_3^- -N levels of other treatments.

Associated Soil Chemical and Physical Properties

Extractable Phosphorus

Daniels and Zipper (1988) emphasize that N is initially the limiting factor on young mine soils, but through time P becomes more limiting. Extractable P was measured at all three stages of the study. Phosphorus content was highest on the 112 Mg ha^{-1} sludge treatment throughout the 16-yr period (Fig. IV.4). Extractable P was initially quite high

(54g kg^{-1}) in the control treatment. Increased P solubility is common in young mine soils as P bound in carbonates of the overburden material are exposed to the surface and weathering conditions. Through time, the control, native soil, sawdust, and 22 Mg ha^{-1} sludge treated plots approached comparable values. The 56 and 112 Mg ha^{-1} sludge treatments remain significantly higher. A doubling of the amount of extractable P may be the most significant contribution of sludge amendments to mine soil quality (Fig. IV.4). Phosphorus was moderately correlated with organic matter, TKN, mineralizable nitrogen, and ECEC (Table IV.2), showing that the differences in P levels among treatments were a function of P supplied via organic P sources.

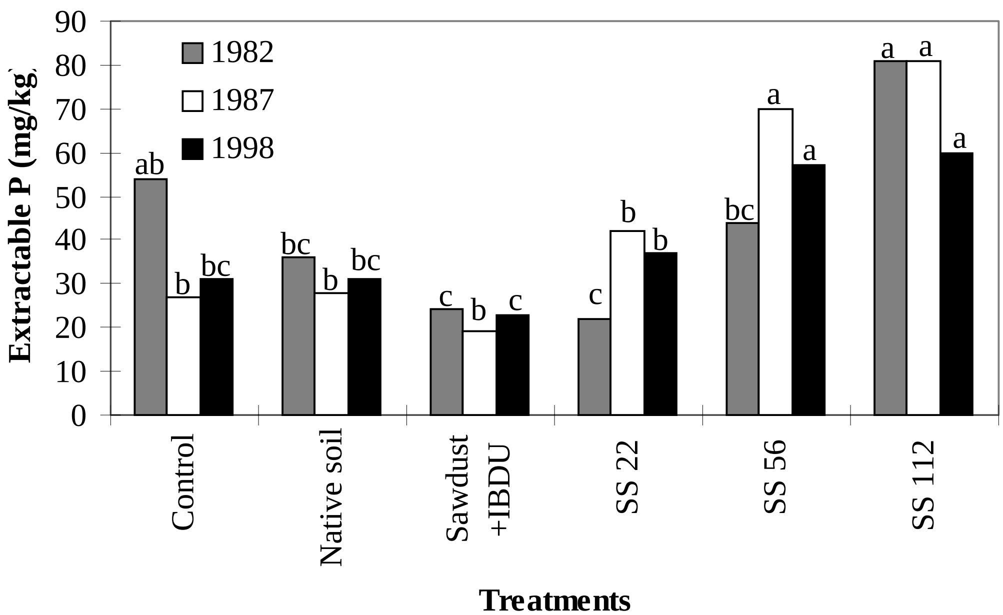

Figure IV.4. Sodium bicarbonate extractable phosphorus of amended mine soils for 1982, 1987, and 1998. Sewage sludge (SS) was applied at rates of Mg ha^{-1}.

After initial amendment, soil pH for all treatments was relatively high compared to native forest soils in the region, ranging from 6.1 to 7.4 (Table IV.4). The lime-amended native-soil and sludge-treated plots were at or above pH 7.0. The pH of the sawdust treatment was significantly lower than the other plots at 6.1. The low pH may have been due to higher levels of carbonic acid and nitrification of biological-reactive organic N sources, with liberation of H^+ ions into the soil solution. Soil pH of the control and sludge-treated plots declined from several tenths to a whole pH unit since the study was initiated, while the pH of the sawdust plots increased 0.2 pH units. The pH of the native-soil-treated plots remained higher than the rest because it was limed (7.8Mg ha^{-1}) initially to raise the pH from 4.4 to a level comparable to the 2:1 sandstone:siltstone overburden material (7.5), and the liming effect has persisted.

Soil cation exchange capacity is strongly influenced by organic matter content, especially in young mine soils. Cation retention capacity varied slightly across treatments with the 112 Mg ha^{-1} sludge having the highest capacity and the control the lowest in 1982. By the fifth growing season, the CEC for all treatments declined slightly from the original levels. In 1998, the CEC approached typical levels for forest soils of the region ranging from 1.7 to 2.8 cmol $_{(+)}$ kg^{-1}. The low CEC value for the native-soil-treated plots may be a function of the quality and distribution of SOM between the light and heavy fractions as well as the overall concentration. Light fraction SOM is generally composed of more labile and easily decomposed material, while heavy fraction SOM is composed of more recalcitrant and stable humus material. Humus and heavy fraction SOM more directly contribute to soil cation exchange capacity than light fraction SOM. After 16 years, a high percentage of SOM in the native-soil-treated plots was contained in the light fraction, which indicates the organic matter is relatively fresh and not decomposed enough to significantly contribute to CEC.

It follows that exchangeable base cation (K, Mg and Ca) levels were low or near depletion across all treatments. Pichtel et al. (1994) reported decreases of 50% or more in the concentrations of P, K, Mg, and Ca on some topsoil-amended plots with increased soil depth. They attributed the decrease in nutrient content to plant growth and the accumulation of these nutrients on the surface as crop residues. In our study, the forage crop has never been harvested, which suggests similar accumulation of nutrients on the surface may be possible.

Table IV.4. Selected chemical properties of the treated mine soils at time of amendment (1982), after the fifth growing season (1987), and 16 years after initial amendment (1998).

Year	Treatment	TKN	pH	CEC ‡	K	Mg	Ca
		g/kg		Cmol(+)/kg			
1982	Control	0.62 c	6.8 c	7.0 d	0.20 b	3.44 c	3.3 d
	Native soil (30 cm)	0.36 c	7.2 ab	7.8 cd	0.21 b	3.93 b	3.6 d
	Sawdust + IBDU †	1.85 b	6.1 d	10.1 bc	0.42 a	4.56 a	5.1 cd
	Sludge (22 Mg/ha)	1.19 bc	7.4 a	9.3 cd	0.16 b	2.62 d	6.5 c
	Sludge (56 Mg/ha)	1.99 b	7.3 ab	12.4 b	0.16 b	2.36 de	9.9 b
	Sludge (112 Mg/ha)	3.59 a	7.0 bc	15.7 a	0.20 b	2.03 e	13.4 a
1987	Control	1.42 c	6.7 b	6.4 c	0.30 b	2.50 c	3.5 c
	Native soil (30 cm)	1.12 c	7.0 a	6.8 c	0.30 b	2.43 c	4.0 c
	Sawdust + IBDU	2.40 b	6.1 d	9.5 b	0.42 a	3.50 a	5.5 b
	Sludge (22 Mg/ha)	2.70 b	6.4 c	8.9 b	0.34 b	2.69 bc	5.8 b
	Sludge (56 Mg/ha)	3.50 a	6.7 b	11.5 a	0.41 a	2.83 b	8.1 a
	Sludge (112 Mg/ha)	3.70 a	6.6 b	12.3 a	0.39 a	2.87 b	8.9 a
1998	Control	1.35 bc	6.4 bc	2.5 a	0.10 ns §	0.33 ns	1.4 ns
	Native soil (30 cm)	1.06 c	7.2 a	1.7 b	0.09	0.32	1.1
	Sawdust + IBDU	1.32 bc	6.3 c	2.8 a	0.09	0.32	1.4
	Sludge (22 Mg/ha)	1.43 b	6.3 c	2.7 a	0.10	0.34	1.3
	Sludge (56 Mg/ha)	1.65 ab	6.4 c	2.5 a	0.08	0.34	1.2
	Sludge (112 Mg/ha)	1.90 a	6.6 b	2.2 ab	0.09	0.31	1.2

* Means within columns followed by the same letter were not significantly different (P< 0.10).
† Isobutyl Di-Urea (IBDU) was added as a slow release nitrogen source.
‡ CEC 1998 samples calculated as effective cation exchange capacity.
§ ns = not significant

Some of these resources may have also been allocated to woody biomass and overall tree growth over the past 16 years.

Soil Bulk Density and Porosity

Soil bulk density was calculated for the whole soil and the fine earth fraction (< 2mm). The bulk density for the fine earth fraction ranged from 1.09 to 1.38 Mg m^{-3} and appears to be a function of organic matter content of the treatments (Table IV.5). Soil bulk density was inversely correlated to organic matter content. As organic matter accumulated and increased in mine soils, bulk density correspondingly decreased. Bulk density of the native soil treatment was significantly higher than the control, sawdust, and sludge-amended treatments at 1.38 Mg m^{-3}, which is comparable to values for mineral soils. Soil bulk density of the control treatment was 1.24 Mg m^{-3}. Hence, the difference among the native soil and the control may allow for better aeration and root growth opportunities for plants on the control plots. The sawdust and sludge amended mine soils were closely clustered between 1.14 and 1.09 Mg m^{-3}. Vegetation type did not significantly influence soil bulk density. The average bulk density values for herbaceous vegetation and pine trees were 1.18 and 1.19 Mg m^{-3}, respectively.

Whole soil bulk densities ranged from 1.27 to 1.51 Mg m^{-3} with the organically-amended soils exhibiting lower bulk densities than the control and native-soil treated plots. Daniels et al., (1983) reported the existence of traffic pans on the amendment study at the 20 and 60cm depths which might account for the variation and slightly higher bulk densities. Overall, mine soils amended with at least 22 Mg ha^{-1} sludge or more organic material exhibited significantly lower bulk densities than either the control or the native-soil treatments.

Table IV.5. Bulk density and porosity values for amended mine soils after 16 years.

Treatment	BD Fine Fraction	BD Whole soil	Porosity Total	Porosity Non-capillary	Porosity Capillary
	Mg m^3		%		
Control	1.24 b*	1.51 a	53 bc	23 a	30 b
Native soil + lime	1.38 a	1.51 a	46 c	16 b	30 b
Sawdust + IBDU§	1.12 bc	1.35 b	55 ab	25 a	30 b
Sludge (22 Mg/ha)	1.14 bc	1.41 ab	57 a	27 a	30 b
Slduge (56 Mg/ha)	1.13 bc	1.35 b	58 a	25 a	33 ab
Sludge (112 Mg/ha)	1.09 c	1.27 b	59 a	23 a	36 a

* Means within columns followed by the same letter were not significantly different ($p < 0.10$).
§ IBDU = Isobutyl Di-urea applied as a source of slow release nitrogen.

Clapp et al., (1986) analyzed the bulk density of 23 studies where sewage sludge was applied and found the range in bulk density was 0.69 to 1.78 Mg m^{-3}. In the studies analyzed, the highest bulk densities were exhibited on sites where the sludge was applied 4 yr prior to measurement. Kost et al., (1997) reported fine soil bulk densities of 1.15 and 1.11 Mg m^{-3} at the 0 to 15cm and 15 to 30cm depths for an abandoned mine soil amended with paper sludge. Joost et al., (1987) reported the bulk density of coal refuse amended with 225 Mg ha^{-1} sewage sludge to be 1.13 Mg m^{-3} for the 0 to 30cm depth. Schoenholtz et al., (1992) reported values for woodchip-amended and topsoil-amended mine soils of 0.58 and 0.84 Mg m^{-3} after three growing seasons.

In 1998, the control, sawdust, and sewage sludge-treated plots had comparable organic matter concentrations. Total and non-capillary porosity levels on these treatments appear to be a function of organic matter concentration. Total porosity of the amended soils was within the typical range for soils and ranged from 46 to 59% (Hillel, 1998) and was correlated with organic matter levels ($r = 0.2676$; $p > 0.0659$) (Table IV.2). Non-capillary porosity levels were quite high and all treatments appear to be well aerated. The control, sawdust, and sludge-treated plots were significantly higher than the native-soil treated plots. Capillary porosity ranged from 30 to 36% across treatments.

Rock Fragments and Particle Size Distribution

In 1982, the first growing season after initial amendment, coarse fragment contents of the control and various organic amendments were not significantly different, ranging from 65 to 71% (Table IV.6). After five growing seasons, the coarse fragment content of the native soil-amended mine soils was significantly lower than other treatments at 32%. We believe that the change in content was due to the rapid weathering of the native soil material that was made up of large amounts of E, B, C, and Cr horizons. After 16 years, an observable difference remains between the native-soil-treated plots and other treatments. The control, sawdust, and sludge-treated plots all had similar coarse fragment contents, suggesting that the influence of root decomposition and organic acids on weathering may not be as prevalent in the sampling zone.

Soil texture in mine soils is particularly important for nutrient retention and cation exchange capacity. Particle size distribution of the fine earth fraction (< 2mm) was rather uniform across all treatments. Over the 16-yr period, the sand to silt content generally followed a 2:1 ratio that coincided with the proportions of the original sandstone and siltstone mixture that was applied during construction of the experiment plots. In 1982 and 1987, only four treatments were tested for soil texture. The 1987 data showed some slight variation across treatments. However, in 1998, after 15 growing seasons and additional opportunity for weathering, there were virtually no textural differences among treatments or across time (Table IV.6). Clay content for these treatments was 10% and 9% for all other treatments. Clay content for all treatment plots averaged about 10% for all three years.

Table IV.6. Coarse fragment content and particle size distribution of amended mine soils for 1982, 1987, and 1998.

Year	Treatment	Coarse Fragments	Clay	Sand	Silt
			%		
1982	Control	65 ns ‡	9 ns	64 ab*	27 ns
	Native soil (30cm)	66	10	64 ab	26
	Sawdust + IBDU †	65	9	65 a	26
	Sludge (22 Mg/ha)	71	10	62 b	28
	Sludge (56 Mg/ha)	67			
	Sludge (112 Mg/ha)	67			
1987	Control	68 a	8 c	63 b	29 b
	Native soil (30 cm)	32 d	8 c	70 a	22 c
	Sawdust + IBDU	68 a	9 b	60 c	31 a
	Sludge (22 Mg/ha)	64 ab	11 a	60 c	29 b
	Sludge (56 Mg/ha)	56 bc			
	Sludge (112 Mg/ha)	51 c			
1998	Control	61 bc	10 a	64 ns	26 ns
	Native soil (30 cm)	38 d	10 a	65	25
	Sawdust + IBDU	66 ab	9 b	63	28
	Sludge (22 Mg/ha)	67 a	9 b	61	30
	Sludge (56 Mg/ha)	57 c	9 b	66	25
	Sludge (112 Mg/ha)	61 bc	9 b	65	26

* Means followed by the same letters were not significantly different ($P < 0.10$).
† Isobutyl di-urea (IBDU) was added as a slow release nitrogen source.
‡ ns = not significant

Prediction Model of Plant Productivity

Tree Volume per Plot

To better understand the correlation between tree productivity and key soil quality variables, a multiple linear regression analysis was used to test the influence of organic matter content, aggregate stability, and nitrogen mineralization on tree volume per plot. A stepwise selection procedure was used within the multiple linear regression analysis to determine the best model by optimizing R^2, mean square error (MSE), and the sum of squares of all predicted error.

Multiple regression analysis indicated that the inclusion of variables other than those found significant in single factor correlations did not greatly improve models of tree volume per plot. Within these parameters, potentially mineralizable N best fit the model (R^2 = 0.2827)(Figure IV.5). The model and single factor correlation of this variable was significant at the 0.05 level for predicting tree volume per plot. Although organic matter and aggregate stability are important soil quality parameters, organic matter and aggregate stability were not significant and their inclusion did not significantly improve the R^2 value.

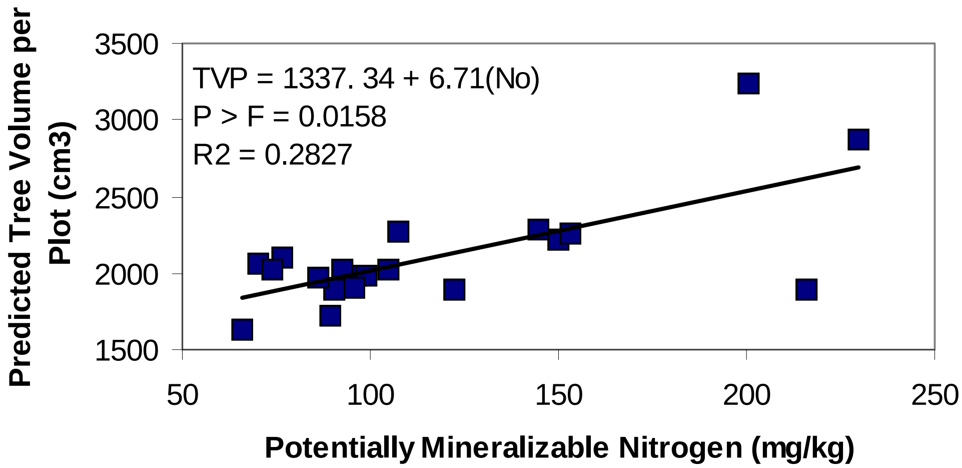

Figure IV.5. Relationship between potentially mineralizable nitrogen (No) and tree volume per plots (TVP).

Herbaceous Biomass Production

Multiple linear regression analysis was also used to test all three variables for their correlation with herbaceous biomass production. No variable was significantly correlated with total herbaceous biomass production. Organic matter content, aggregate stability, and N mineralization potential may not have been significant due to vegetation dynamics and the overall change in composition. Analyzing the correlation between these three variables and grass yield alone shows that there is a weak correlation ($r^2 = 0.1112$; $p > 0.1114$) between grass yield and potentially mineralizable nitrogen (Fig. IV.6). This correlation suggests that an agronomic crop like tall fescue is dependent on mineralizable nitrogen for maintenance and continued sward growth. Roberts et al. (1988) reported that initial tall fescue swards on the control and native soil-treated plots were declining after three growing seasons due to nitrogen stress.

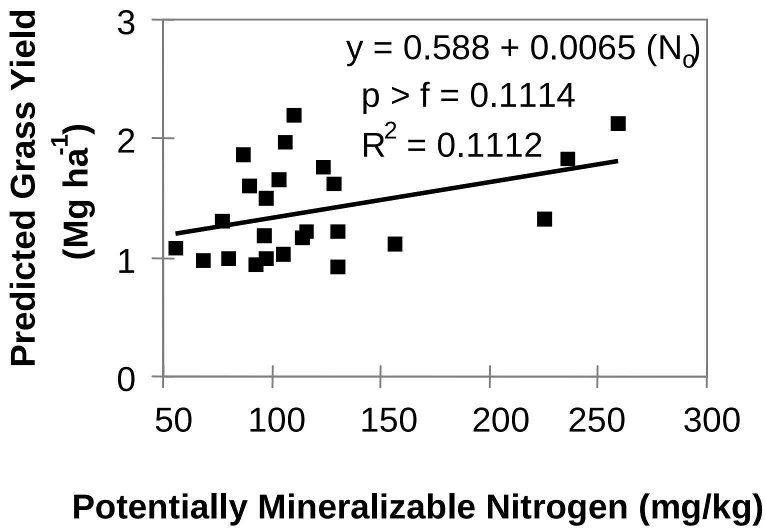

Figure IV.6. A predictive model of grass production using potentially mineralizable nitrogen as a key soil quality variable.

Hornby et al. (1986) explain that sufficient SOM capital needs to be developed in the soil system to serve as a N source and sufficient N mineralization potential needs to be maintained to enable reclaimed lands to sustain vegetation at acceptable production levels without further fertilization. In their study, sufficient SOM capital and N mineralization potential were developed after 4 yr of conventional fertilization and 1 yr after 56 Mg ha^{-1} of sewage sludge was applied and the vegetation was harvested and returned to the mine soil as a mulch. Comparing the unamended control treatment and the 56 Mg ha^{-1} sewage sludge treated in our study (where vegetation was not harvested), SOM concentration, aggregate stability, and N mineralization potentials of these two treatments appear equivalent after 15 years. Multiple regression analysis shows that plant productivity in this mine soil system is primarily a function of SOM decomposition and the nitrogen mineralization potential of these soil processes.

CONCLUSIONS

The sawdust and sewage sludge amendments were initially superior to the control and native soil-treated plots in improving key mine soil quality indicators. The comparative ability of these organic amendments to positively affect organic matter content and total nitrogen was most apparent and pronounced after five growing seasons. However, after 16 years, soil organic matter concentrations and total nitrogen appear to be equilibrating at about 10000 and 750kg ha^{-1}, respectively. Aggregate stability and nitrogen mineralization values were comparable across treatments. Organic matter inputs by vegetation alone over the 16-yr period in the control plots resulted in organic matter and nitrogen mineralization potential values comparable to levels in the organically-amended plots, indicating the importance of vegetation in the soil recovery process, and the rapid rate of incorporation in new mine soils.

A doubling of extractable P levels may be the greatest long-lasting effect of the sludge treatment on mine soil quality, although aggregate stability and bulk density were also positively affected. Available phosphorus was about two times greater in sewage sludge-amended (56 Mg ha^{-1}) mine soils than the 22 Mg ha^{-1} sewage sludge and other treatments. These results suggest that the 22 Mg ha^{-1} sewage sludge treatment might have benefited from the addition of P at the time of application; otherwise, the two treatments were similar in their long-term effect on mine soil quality. Climate, soil temperature, water availability, and other edaphic features appear to be more directly influencing mine soil quality over time than the initial amendments. The results suggest that about 15 years is needed for climate, moisture availability, and other edaphic features to have the same influence on overall organic matter decomposition, N accretion, organic N mineralization levels, system equilibrium, and mine soil quality as a one-time 100 Mg ha^{-1} application of organic amendment. Hence, the one-time application of 100 Mg ha^{-1} organic material appears to accelerate the early recovery of important soil functions.

Ecosystem stability and successful mined-land reclamation are dependent on the restoration of these important soil functions and overall mine soil quality. This study shows that use of organic amendments initially improved organic matter content, total N, potentially mineralizable N, aggregate stability, and other soil properties. However, 15 years later, there appear to be no lasting soil quality improvements over that of the control treatment which received no organic amendments other than natural incorporation of organic matter from plant production.

CHAPTER V. TREE AND VEGETATION RESPONSE TO ORGANIC AMENDMENTS OVER 16 YEARS

ABSTRACT

Organic amendments may be a means of ameliorating mine soils and other disturbed sites. An important component in the definition and assessment of soil quality is the ability of soil to serve as an acceptable medium for plant growth. The specific objectives of this study were (i) to determine the effect on tree and grass productivity after 16 years of a one time application of topsoil, sawdust, and sewage sludge to mine soils, (ii) to determine the effect of organic amendments on plant nutrition, (iii) to develop a predictive model of plant response, and (iv) to determine the effect of vegetation type on key soil properties. In 1982, a mined site was amended with seven different surface treatments: a fertilized control (2:1 sandstone:siltstone overburden), 30 cm of native soil + 7.8 Mg ha^{-1} lime, 112 Mg ha^{-1} sawdust, and municipal sewage sludge rates of 22, 56, 112, and 224 Mg ha^{-1}. The experiment was a randomized complete block design with 4 replicates for a total of 28 plots. Whole plots were split with pitch X loblolly pine hybrid (*Pinus rigida* x *taeda* L.) trees and tall fescue (*Festuca arundinacea* Schreb.). Initial tree and vegetation growth response to organic amendments was favorable and pronounced. After three growing seasons, stem volume on sawdust-amended mine soils was 340% greater than on the unamended control plots. Sixteen years after amendment, the stem volume increment across treatments converged and paralleled the growth of the control treatment. Stem volume of treated mine soils are presently only 18 to 26 % greater than the unamended control comprised of simple overburden material.

Organic amendments increased tall fescue biomass yields during the first 3yr of the study. After 4 growing seasons, productivity declined across all treatments due to N stress. In 1997, 15 yr after establishment, herbaceous biomass yields for the control and native soil-treated plots recovered due to the invasion of sericea lespedeza and other seral species. Tall fescue cover was maintained for a longer period on the sewage sludge treated plots than all other treatments due to higher soil N levels. Mine soils amended with sawdust had a negative effect on herbaceous biomass production compared to other treatments.

The growth response of pitch x loblolly pine and herbaceous vegetation on these amended mine soils indicates that initial soil fertility was favorable and the amendments improved crop production for four to eight growing seasons. The decline in vegetation and tree productivity after this time period indicate a need for monitoring mine soil quality and specific indicators such as organic matter content, aggregate stability, and the availability of nitrogen and phosphorus. The results suggest further soil fertility management is needed to sustain plant productivity for more intensive post-mining land uses. Organic amendments improved initial soil fertility for crop establishment, but it appears a one-time application will have no-long lasting effect on plant productivity. Overall, crop growth was the same among treatments after 16 years.

INTRODUCTION

Surface mining for coal removes vegetation and displaces nutrient resources from the site that are necessary for nutrient cycling and plant growth. The disruption of natural ecosystems, vegetation communities, and the subsequent decline in soil productivity and water quality during coal mining operations were part of the impetus for the Surface Mining Control and Reclamation Act (SMCRA) of 1977. This legislation was also passed in anticipation of continued coal use and future land disturbances. A fundamental provision of the law requires that land disturbed by mining be reclaimed to its original use or one of higher value, and that its productivity be equal to or greater than it was before mining. This provision implicitly requires the maintenance and enhancement of soil quality if the requirements of the law are to be fulfilled and a productive vegetation community maintained.

During the past 30 years, mine soils have been amended with inorganic fertilizers to re-establish nutrient cycles, improve soil quality, and facilitate vegetation establishment, but the feasibility and value of organic amendments is uncertain. Several studies have shown that initial growth response to inorganic and organic amendments can be favorable (Chaney et al., 1995; Thorne et al., 1984; Harris et al., 1984). The benefits of organic amendments have been reported to be longer lasting than inorganic fertilizers (Schoenholtz et al., 1992; Schoenholtz, 1990). Sawdust, woodchips, topsoil, fly ash, papermill sludge and municipal biosolids have been applied to mine soils as organic matter sources to increase nutrient capital, improve soil quality, and accelerate biologically-mediated processes needed for sustained plant growth (Kost et al., 1997;

Roberts et al., 1988). A number of studies have shown an initially-positive vegetation response to these amendments (Schoenholtz et al., 1992; Moss et al., 1989; Roberts et al., 1988; Seaker and Sopper, 1984). Overall, these studies show that tree response and survival is variable and dependent upon the application rate. The trees, generally, respond positively to increased organic matter content and available N. Similarly, these studies show that the initial response of herbaceous vegetation to sewage sludge and other organic amendments is positive, but biomass yields appear to decline through time unless managed and harvested as an agronomic crop with annual amendment applications.

Bradshaw (1987) emphasized that full recovery of an ecosystem and mine soil quality is dependent on vegetation for the long-term improvement of the physical and biological diversity of mined lands. The long-term objective of mined-land rehabilitation is to establish a plant community that restores and maintains soil and water quality at satisfactory levels without additional fertility management (Ward and Koch, 1996). Cover crops and trees are the vegetation types commonly used in mined-land reclamation. In agroecosystems and mined-land reclamation, cover crops and residue management are important for maintaining soil quality and productivity and minimizing the environmental impact of agricultural and mining practices. Cover crops and herbaceous vegetation are a source of organic carbon that is critical to the carbon cycle and overall nutrient capital. Similarly, cover crops reduce soil erosion caused by wind and water.

Trees are used in reclamation for several reasons: long-term ecosystem stabilization, potential commercial and aesthetic value, and potential ameliorative effects on soil quality (Torbert and Burger, 1993; Fisher, 1990; Ashby, 1987). Fisher (1990) defined five mechanisms by which trees can ameliorate degraded soils: nitrogen fixation, efficient nutrient cycling, organic matter additions, microclimate moderation, and rhizosphere interactions. These mechanisms are the main attributes of agroforestry systems and the major presuppositions for the use of tree-based systems in the restoration of degraded agroecosystems and other marginal lands. In chapter four of this thesis, the changes in soil physical, chemical, and biological properties were analyzed to evaluate the effects of organic amendments on mine soil quality. The objectives of this sub-study were (i) to determine the effect of a one-time application of organic amendments to mine soils on tree and grass productivity after 16 years, (ii) to determine the effect of organic amendments on plant nutrition, (iii) to develop predictive models for plant response, and (iv) to determine the effect of vegetation type on key soil properties. Increased knowledge of plant response to feasible topsoil substitutes and the changing levels of soil quality variables through time should assist in the evaluation of management techniques, reclamation guidelines, and overall mine soil quality and productivity.

MATERIALS AND METHODS

Vegetation Sampling and Analyses

In July 1997, tree survival was determined for each treatment. Tree heights and diameters were measured to obtain a relative volume index of individual tree growth, basal area, and overall plot productivity. Heights of all the trees in the plot were measured to the nearest centimeter. Tree diameters at breast height (dbh ≈1.37 m above ground level) were measured to the nearest tenth of a centimeter. Trees were classified according to crown class: dominant, co-dominant, and intermediate. Suppressed trees were not included.

In November 1997, 20 fully-elongated needle fascicles of current year's growth were obtained from the upper crown of three dominant or co-dominant pitch X loblolly hybrid trees. Selection of fascicles from a dominant or co-dominant tree was predicated on whether the trees were centrally located within the plot to avoid edge effects and overexposure to light. The needle fascicles were composited in a paper bag and kept in cool storage. The tissue samples were dried to a constant weight at 65°C. Mean fascicle length and weights were recorded for each treatment. Composite needle samples were ground in a Wiley mill to pass a 1mm sieve. Needle tissue samples were dry-ashed at 500°C for at least 12 hr and extracted with 6 M HCl. The extracts were analyzed with an inductively coupled plasma emission spectrophotometer (ICAP) to determine foliar concentrations of P, K, Ca, Mg, Fe, Mn, Zn, Cu and Boron. Foliar N content was obtained by the micro-Kjeldahl digestion procedure (Bremner and Mulvaney, 1982) and analyzed colorimetrically using a Technicon Auto-Analyzer II (Technicon Industrial systems, Tarytown, NY.)

In October 1997, standing biomass of herbaceous vegetation was measured. Each 3.5m x 3.5m plot was gridded into 100 0.1 m^2 quadrats. Using a random numbers table, three 0.1m^2 clip plots were randomly located and aboveground biomass was clipped to ground level. Aboveground biomass was categorized into three vegetation classes (grass, legumes, and forbs) and separated within each plot. Composite samples for each cover class were obtained for each plot, dried at 65°C, and weighed. Cumulative biomass for each plot was then calculated. Plant tissue samples were analyzed as described for pine needle samples.

Methods of soil analyses used in assessing the effect of vegetation type on mine soil quality are described in chapter IV of this thesis.

Statistical Design and Data Analysis

The study design was a randomized complete block design with seven treatments and four replicates. Due to the detrimental effect the 224 Mg ha^{-1} sludge treatment had on initial tree seedling survival (Moss, 1986) and the fact that increasing the loading rate from 112 to 224 Mg ha^{-1} had no further effect on tall fescue yields (Daniels and Haering, 1994; Roberts et al., 1988), this high application rate was not analyzed in this study. Tree and herbaceous vegetation response were analyzed independently. Duncan's Multiple Range Test was used to separate response means when there was a significant Pr > F-value (0.10). In 1998, soil data and treatment effects were analyzed as a split-block design with treatment means of the main effect pooled across vegetation types (herbaceous vegetation vs. pine trees). Multiple linear regression analysis of selected soil properties was used to test and predict plant response. A stepwise selection procedure was used within the multiple linear regression analysis to determine the best model by optimizing R^2, mean square error (MSE), and the sum of squares of all predicted error.

RESULTS AND DISCUSSION

Tree Productivity

In 1997, after 15 growing seasons, tree survival rates for the unamended control, sawdust, and native soil treatments were the same, 69, 64, and 61%, respectively. Sewage sludge (SS) decreased tree survival by about 10% compared to the control (Table V.1). After three growing seasons, Moss (1986) reported an adverse treatment effect on seedling survival for the 112 and 224 Mg ha^{-1} sludge treated plots. The survival rate for the 112 Mg ha^{-1} SS was 31% and only 6% for the 224 Mg ha^{-1} SS treatment. Therefore, these treatments are not presented because of the difference in tree density and the fact that they are not suitable treatments for tree establishment. The pure loblolly pine genotype (25% of the total trees planted) suffered near complete mortality due to a very harsh winter in 1985 (Moss, 1986).

Table V.1. Average per plot, survival, number of trees per plot, basal area, height, diameter, and volume of pitch x loblolly pine trees growing on amended mine soils in Wise Co., Virginia 15 years after establishment.

Treatment	Survival	# of Trees	Basal Area	Height	Diameter	Plot Volume
	(%)		(m^2 ha^{-1})	(m)	(cm)	(m^3)
Amendment						
Control	69 a	7	98 ns	7.2 ns	13.1 ab	0.20 b
Topsoil	61 ab	7	96	7.0	11.8 b	0.19 b
Sawdust + IBDU	64 ab	6	110	7.5	14.4 ab	0.23 a
SS 22	56 b	6	107	7.1	13.1 ab	0.22 ab
SS 56	58 b	6	116	7.4	15.0 a	0.24 a

* Means within columns followed by the same letter are not significantly different (alpha = .10 level Duncan's Multiple Range Test).

In 1997, 15 growing seasons after establishment, cumulative basal area was the same across treatments regardless of tree density (Table V.1). Basal area ranged from 96 to 116 m^2 ha^{-1}. After eight growing seasons, basal area of the sawdust plots was greater than other treatments but declined from 1990 to 1996 (Figure V.1). This decline appears due to tree density induced mortality, which commonly occurs as trees mature and compete for available light, water, and nutrient resources. The continued accumulation of basal area on the other treatment plots during this time would suggest that mortality was not as high on the other treatments and that water was not a limiting factor on the site. The sharp increase in basal area accumulation from 1996 to 1997 was due to heavy limb pruning (Figure V.1).

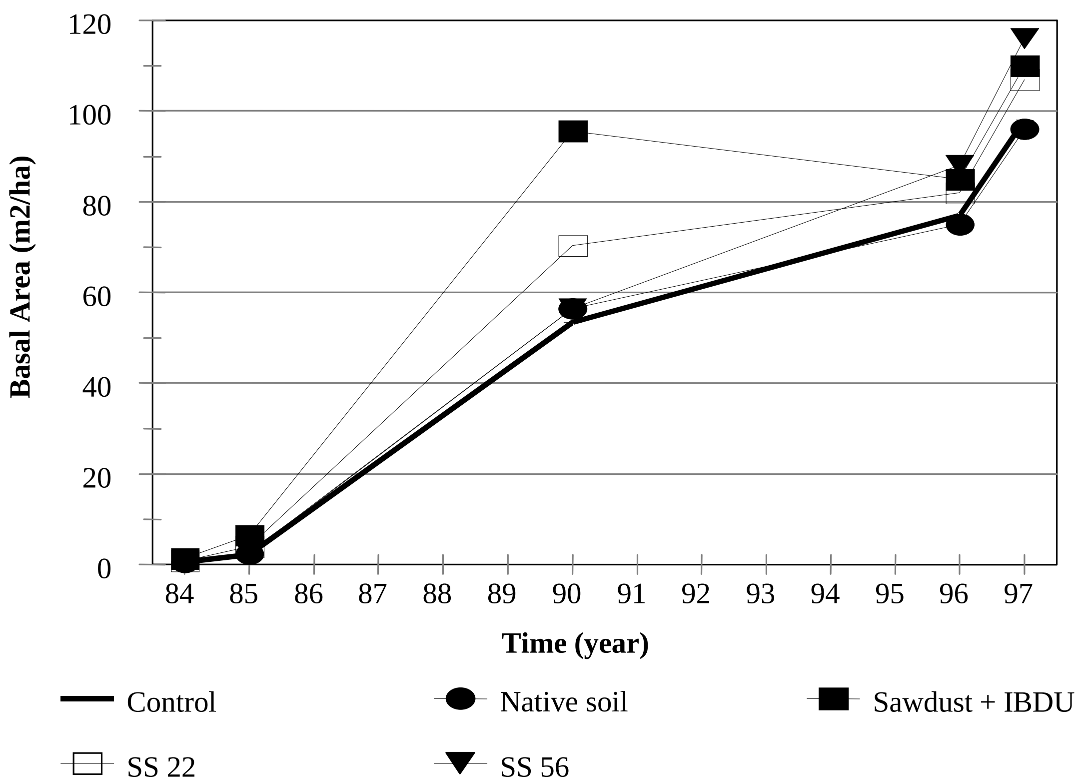

Figure V.1. Cumulative basal area for pitch x loblolly pine trees through time (16 years).

Tree height for all treatments was comparable, ranging from 7.5m for the sawdust treatment to 7.0 m for native soil. Diameter of trees was highest in plots treated with the highest rate of sludge, and lowest in the control and native-soil-treated plots. Tree volume on the sawdust and SS amended plots was slightly higher compared to the control and native soil treatments (Table V.1). The relative effect of these treatments on tree volume varied greatly through time (Figure V.2).

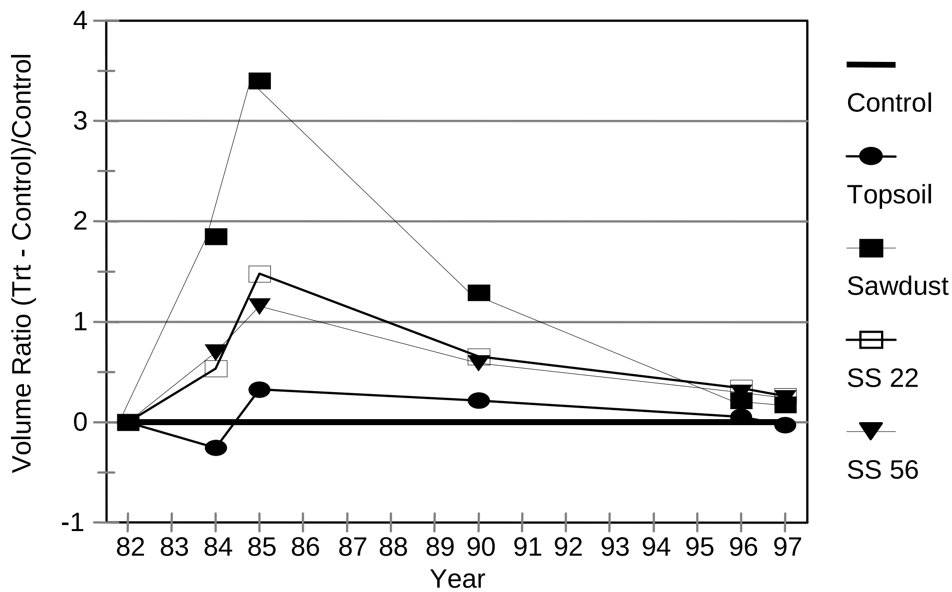

Figure V.2. The relative stem volume of pitch x loblolly pine trees growing on organically amended mine soils in Wise Co., Virginia. Sewage sludge (SS) is followed by the application rate in Mg ha^{-1}. Volume ratio is determined by the equation (Treatment Stem Volume - Control Stem Volume)/ Control Stem Volume.

The 1985 stem volume of trees on the sawdust treatment was 3.4 times or 340% greater than the control (Figure V.2). This difference was similar to that reported by Schoenholtz (1990) who found that a wood-chip-amended mine soil resulted in 203% more stem volume than an unamended control treatment. After the 1985-growing season, seedling volume on the sawdust plots was considerably greater than either the 22 or the

56 Mg ha^{-1} SS treatments (192.4cm^3 vs. 117.5 and 98.5cm^3, respectively). Although stem volume on the sewage sludge (22 and 56 Mg ha^{-1}) treatments was less than one-half that of the sawdust treatment after the three years, the tree growth response to sludge was favorable compared to the unamended plots and resulted in 1.48 and 1.15 times more volume. Sopper and Kardos (1973) reported similar increases in coniferous and hardwood species that received annual applications of effluent.

However, after 8 growing seasons, the tree volume on the organically amended treatments began to converge toward the control. In 1990, the sawdust, 22 and 56 Mg ha^{-1} sludge treatments still contained 129, 65, and 59% more stem volume than the control but the volume increment declined more than 200% for the sawdust-amended plots and about 100% for the sludge-treated plots. In 1997, SS (22 and 56 Mg ha^{-1}) treatments contained slightly more volume than the sawdust treatment compared to the control at 26 and 24%, respectively. At present, all organically-amended treatments contain 18 to 26% more stem volume than the control treatment. The decline in tree response through time and convergence toward the control is due to a decline in treatment effect after about 4 to 8 growing seasons.

This decline in tree growth through time was similar to that reported by Chaney et al (1995) and Harris et al. (1984). Chaney et al. (1995) found that the height growth of black walnut (*Juglans nigra*) and northern red oak (*Quercus rubra*) was very slow (10 cm yr^{-1}) after 12 years. Harris et al. (1984) reported that no appreciable increase in diameter was detected in red pine (*P. resinosa* L.) growing on a sludge-amended site after six growing seasons. In 1997, the relative stem volume of the native soil treatment converged with the volume standard of the control treatment. Hence, the initial difference in performance that was so obvious during the establishment period of this study was no longer evident after 15 growing seasons, and treatment response appears to have equilibrated to the level of the control (Figure V.2).

Foliage Analysis of Pitch X Loblolly Pines

The effects of organic amendments, particularly sewage sludge, on plant nutrition has been the subject of much research as concern about plant uptake of heavy metals with links to the food chain have been raised by society. One reason sewage sludge and other organic amendments are applied to depauperate soils is to increase levels of N and other nutrients essential for sustained productivity and site stability (Seaker and Sopper, 1988). As the most widely planted tree species in the southeastern U.S., loblolly pine nutrition has been well documented, but information on the nutrient requirements of hybrid species is not readily available, and Rathfon et al. (1993) reported that loblolly pine nutrient concentrations are adequate for diagnosing the nutrition levels of hybrid crosses.

Foliar concentrations of N, P, K. Ca and Mg in the pitch x loblolly pine trees were measured to determine nutrient status. Accepted critical levels are 1.1, 0.1, 0.35, 0.12, and 0.07%, respectively (Allen, 1987; Stone, 1968). After 15 growing seasons, foliar N and P percentages for all treatments were below the established critical levels (Figure V.3). Foliar K content was low for all treatments and at critically low levels for the control, native soil, and sludge (22 and 56Mg ha^{-1}) treatments. Calcium and Mg concentrations were sufficient for all treatments, despite low soil Ca and Mg levels.

Nambiar and Fife (1991) point out nutrient retranslocation is mediated by shoot growth requirements and there is continual competition for nutrient reserves as the shoot and crown develop. In assessing crown development and nutrient allocation of the pitch x loblolly pines, symptoms of nutrient deficiencies were observed. Some of the pine needles, sampled from fully-elongated foliage of the upper crown, exhibited chlorotic symptoms with a slight yellow pigment and some necrosis at the needle tips. Although total soil N has aggraded in most treatments, the recalcitrant nature of organic N may be limiting N mineralization. The N deficiency may also be limiting plant uptake of P and K. Nutrient concentrations were a bit lower on the unamended control plots; however, element concentrations on all plots declined during the 12-year period (Figure V.3).

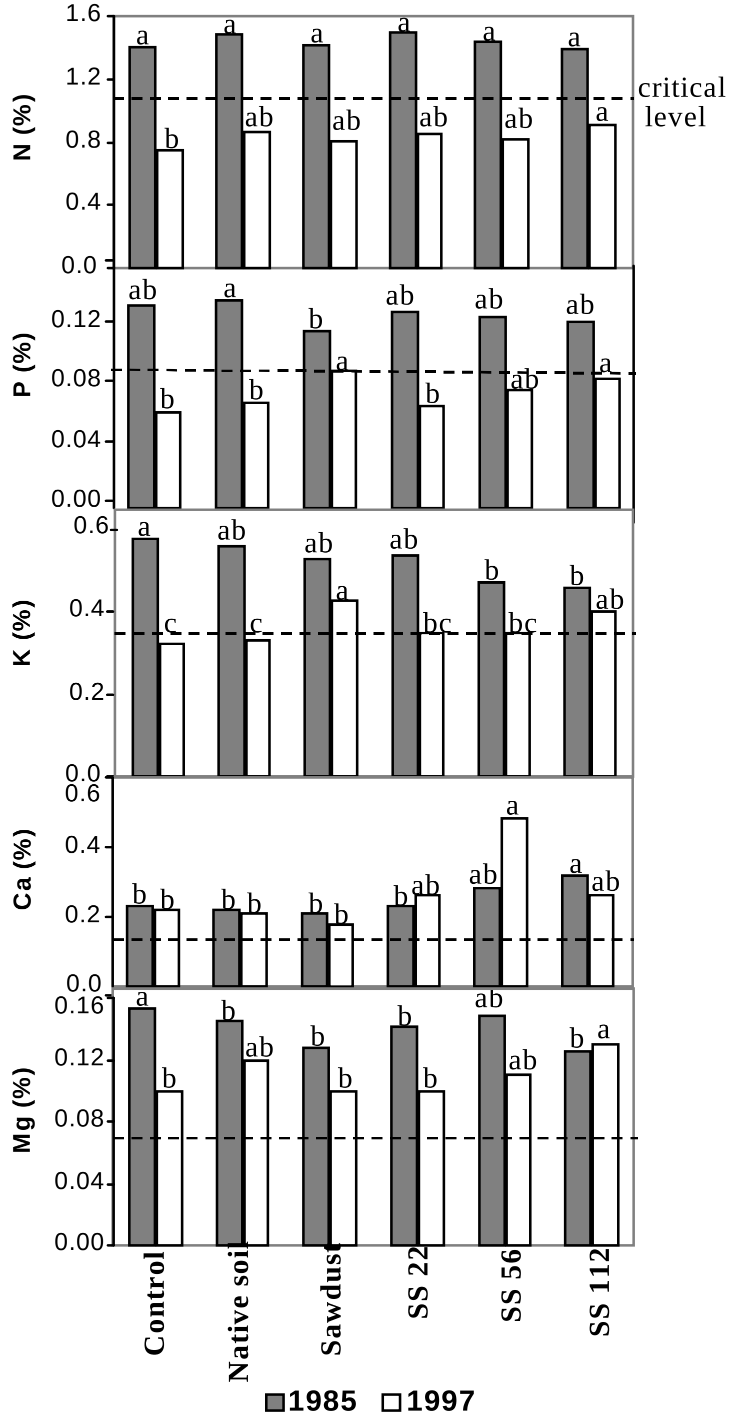

Figure V.3. Foliar macronutrient content for pitch x loblolly pine trees growing on amended mine soils for the years 1985 and 1997. Means followed by the same letters were not significantly different within sampling year.

Definitive information on pitch x loblolly pine micronutrient requirements and deficiency thresholds does not exist. As previously mentioned, nutrient thresholds of the parent or closely related species are relied on to determine possible foliar nutrient deficiencies (Rathfon et al., 1993). Micronutrient critical levels for conifers are not as well documented as macronutrients and there appear to be some incongruities in the literature. Stone (1968) reviewed the micronutrient needs of forest species and reported 306 to 392 mg kg^{-1} to be the intermediate range for Mn in loblolly pine (*Pinus taeda* L.). He reports 330 to 346 mg kg^{-1} to be the intermediate range for Mn in slash pine (*P. elliottii* Engelm.). In contrast, Pritchett and Fisher (1987) report the Mn critical range for slash pine tissue to be 8 to 12 mg kg^{-1} so there appears to be a wide range between the sufficiency and critical levels for Mn. Manganese levels in this study ranged from 78 to 153 mg kg^{-1} (Table V.2) and appear low according to Stone's findings. However, the values for all treatments do not appear to be growth limiting or close to approaching the critical values proposed by Pritchett and Fisher for slash pine.

Table V.2. Foliar micronutrient concentrations of pitch x loblolly pine trees growing in mine soils 16 years after amendment, Wise Co., Virginia.

Year	Treatment	Mn	Zn	Fe	B	Cu
				mg/kg		
1985	Control	198 a	56 ab	105 ns	13.0 ab	4.6 ns
	Native Soil	156 a	49 b	102	13.8 ab	4.3
	Sawdust + IBDU	142 b	58 ab	98	16.9 a	4.6
	Sludge 22 Mg/ha	152 a	58 ab	97	12.0 b	5.1
	Sludge 56 Mg/ha	89 bc	68 ab	95	11.9 b	4.3
	Sludge 112 Mg/ha	66 c	75 a	104	12.3 ab	5.5
1997	Control	153 a	34 d	86 ns	3.9 b	1.6 ns
	Native Soil	114 ab	38 cd	129	5.6 ab	2.1
	Sawdust + IBDU	140 a	44 bc	80	7.4 a	1.3
	Sludge 22 Mg/ha	134 a	42 bc	119	6.4 ab	1.3
	Sludge 56 Mg/ha	142 a	46 b	107	5.9 ab	3.9
	Sludge 112 Mg/ha	78 b	63 a	140	5.3 ab	3.0

* Means within columns followed by the same letter are not significantly different (Duncan's multiple range test, alpha = 0.1).
† IBDU = Isobutyl Di-urea as a slow release nitrogen source.

McBride (1994) points out that the solubility of Mn in soils can fluctuate tremendously through time between deficient and toxic levels as moisture conditions change, affecting soil reduction-oxidation reactions and the mobility of Mn. Hence, plant availability must also fluctuate between these levels making the diagnosis of deficiency in plants difficult. Similarly, the presence of organic acid chemicals such as polyphenol, that are common in pine needles, can reduce Mn oxides and increase Mn solubility. Increased concentrations of Mn and Fe are common as mine spoil material continues to weather. Mills and Jones (1996) listed the sufficiency range for iron concentrations for loblolly pine as 40 to 118mg kg^{-1}, all surface treatments are within this range. Iron concentrations for the SS treatments were higher than previously reported by Moss et al., (1989), while other micronutrient concentrations had declined from previous levels and appear to be in the deficient to critical ranges for pine nutrition.

Mills and Jones (1996) report the sufficiency range for Zn to be 28 to 53mg kg^{-1}, zinc values for all treatments, except for the 112-Mg ha^{-1}SS, are within this range. The zinc values for the 112-Mg ha^{-1} SS treated plots would appear to be luxury consumption. However, boron concentrations for all treatments in this study were below the sufficiency range of 15 to 31 mg kg^{-1} listed by Mills and Jones. Likewise, copper values for the control, native soil, sawdust, and 22 Mg ha^{-1} SS treated plots are below the 3 to 6 mg kg^{-1} range reported for proper loblolly pine nutrition (Table V.2). These nutrient imbalances suggest the lack of an indigenous source of boron and copper in the mine spoil material and the trace amount added at the time of amendment either in sawdust or sewage sludge were fixed by iron oxides as stable humate complexes or were diluted in the foliage.

Herbaceous Biomass Production

Herbaceous ground cover crops are commonly used to reclaim mined sites to fulfill the revegetation requirements for bond release. For productive post-mining land use and ecosystem stability, the aim is to establish an environment in which vegetation can grow abundantly, enhance nutrient cycling, and be self-perpetuating.

Tall fescue was grown and maintained as the dominant vegetative community during the first 5 yr of the study. Leguminous species were eliminated and prevented from establishing a competing vegetative community for the tall fescue. Ground cover and crop response to organic amendments was very favorable, tall fescue yield was highest during the first three growing seasons. Highest yields were obtained and maintained for the three growing seasons on municipal sewage sludge treatments ≥ 56Mg ha^{-1}. In 1983, two growing seasons after amendment, the range of yields for the 56, 112, 224 Mg ha^{-1} sludge treatments were 10.1,13.6, and 16.5Mg ha^{-1} of standing biomass. The level of biomass production declined across all treatments after the 1986 growing season. Yields on these sludge treatments had declined to 5.2, 7.2, and 5.0 Mg ha^{-1} which were comparable to agronomic yields in the area. Roberts et al. (1988) concluded that increasing the loading rate from 112 to 224Mg ha^{-1} had no long-lasting effect on tall fescue yields. Earlier tall fescue biomass yields were presented by Roberts et al. (1988) and Haering et al. (1990), and are presented here to demonstrate the trend and fluctuations in annual yields (Figure V.4). Overall biomass yields were lowest in 1987 ranging from 2.8 to 1.1 Mg ha^{-1}. The increase in biomass from 1987 to 1997 corresponded with the increasing invasion of sericea lespedeza during the 10 yr period.

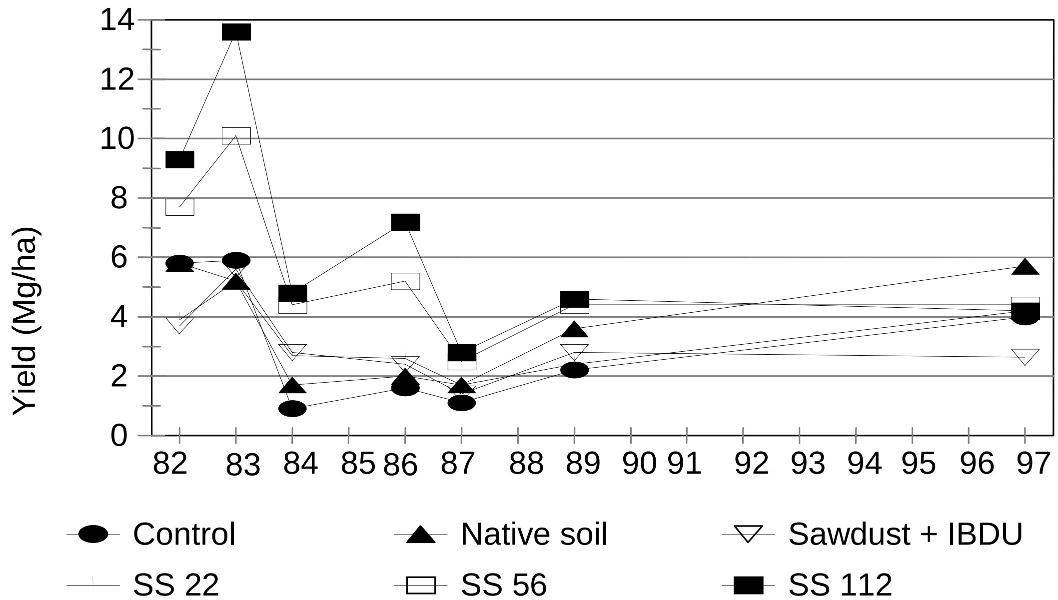

Figure V.4. Dynamics of herbaceous biomass production on amended mine soils through 15 growing seasons.

Ground cover composition of the study plots changed as sericea lespedeza (*Sericea cuneata*), an early seral species, succeeded onto the site after management ceased. Roberts et al.(1988) reported that tall fescue growth was particularly sparse on the control and topsoil treatments due to apparent N stress. Sparse growth offers an opportunity for species that can compete in N deficient sites to get established. Growth and invasion of sericea lespedeza was most noticeable on the control and topsoil plots compared to the other plots.

The influence of this leguminous species on standing biomass yield was pronounced on all treatment plots except the 56 and 112 Mg ha^{-1} SS treatment due to N stress and competition. Tall fescue cover was maintained for a longer period of time on the sewage sludge treatments due to higher soil N levels through time. In 1997, standing biomass was highest on the topsoil treatment (5.66 Mg ha^{-1}), where legumes accounted for 4.06-Mg ha^{-1} or 72 % of the total yield on a dry weight basis (Figure V.5). Total biomass production of the topsoil treatment was not significantly different from the three SS treatments. Biomass yields for all treatments was in the order of topsoil $\geq$ SS (56 Mg ha^{-1}) $\geq$ SS (22 Mg ha^{-1}) = SS (112 Mg ha^{-1}) = control > sawdust. Standing biomass production was lowest on the sawdust treatment at 2.6 Mg ha^{-1}.

Hornby et al.(1986) report that litter accumulation from the early vegetative community can lead to high C/N ratios and reduce available N which may lead to stagnation and plant failure. The poor performance and productivity on the sawdust-treated plots suggest that such nutrient dynamics may have caused this decline despite the addition of a slow release source of nitrogen to the treatment.

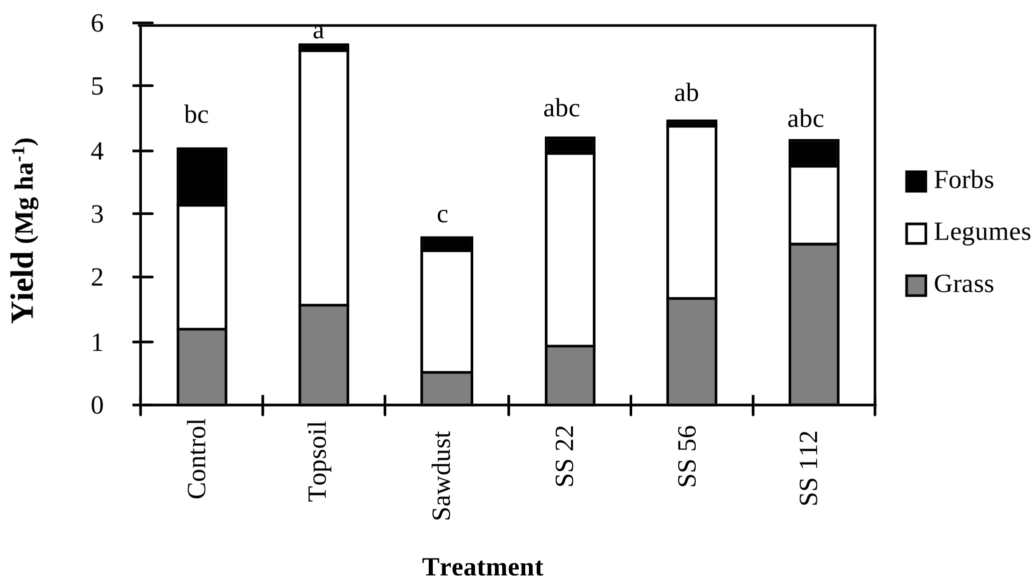

Figure V.5. Annual standing biomass and distribution of herbaceous cover in Fall, 1997.

Foliage Analysis of Herbaceous Cover

Tissue concentrations of N, P, and K were low across all treatments (Table V.3). Foliar N levels in the tall fescue ranged from 6.2 to 7.6 g kg^{-1} and lowest concentrations occurred in the sawdust treatment. Among sericea lespedeza, N levels were expected to be generally higher and ranged from 7.0 to 10.4 g kg^{-1}. These tall fescue N levels are considerably lower than those reported by Haering et al. (1990). In1989, foliar N levels ranged from 12.2-g kg^{-1} for the control to 14.5-g kg^{-1} for the 112-Mg ha^{-1} SS treated plots. The decline in foliar N content for herbaceous vegetation and pitch x loblolly pines appear comparable which suggests less plant available nitrogen for vegetation types through time. Overall N, K, Ca, and Mg levels were not significantly different among treatments, but there was a significant vegetation type ($p = 0.0183$) and amendment treatment $\times$ vegetation type interaction for N concentrations ($p = 0.0571$) (Table V.3). Higher N values for the leguminous species were expected which suggests plant nutritional needs vary and must be considered during species selection. N content of legumes growing on the unamended control and native soil-treated plots were considerably higher than other treatments and were well correlated with higher legume biomass yields.

Tissue P content in both grass and legumes was generally higher on the sewage sludge treatments and were correlated with soil P levels. In 1989, foliar P of tall fescue samples ranged from 2.83 to 3.38 g kg^{-1} (Haering et al., 1990); however, P content for both tall fescue and sericea lespedeza were well below previously reported levels. Bengston and Mays (1978) report a critical concentration of < 2.4g kg^{-1} for P in tall fescue. Tall fescue is clearly P deficient, which illustrates the need for continual maintenance fertilization. Daniels and Zipper (1988) state that for reclaimed sites, N is initially the limiting factor, but that over time P becomes the limiting factor. Tall fescue foliar N, P, and K levels of 1983 and 1997 show that plant nutrient concentration has declined through time and is below critical levels for tall fescue (Figure V.6). The presence of different vegetative communities appears to have affected nutrient dynamics within the organic amendment study. The overall availability of nutrients to each plant species changed as successional processes and interspecific competition affected soil-plant relations. These data show that long-term plant needs and nutrient dynamics are important for determining the overall suitability of reclamation species.

Table V.3. Foliar macronutrient concentrations for herbaceous vegetation growing on amended mine soils.

Treatment	N	P	K	Ca	Mg
	Elemental Concentrations				
			g/ kg		
Amendment (A)					
Control	8.4	0.5 ab*	1.5	5.2	1.4
Topsoil	8.6	0.4 b	0.8	4.7	1.2
Sawdust + IBDU §	7.8	0.5 b	1.6	5.3	1.5
SS (22 Mg ha-1) ¶	7.7	0.8 ab	2.8	5.8	1.6
SS (56 Mg ha-1)	8.3	1.2 a	2.5	5.9	1.6
SS (112 Mg ha-1)	7.3	1.0 ab	3.9	5.8	1.3
Mean Concentration	8.1	0.7	2.2	5.5	1.4
Vegetation Type (V)	p = .0183	ns‡	ns	p = .014	ns
Grass	7.0	0.8	2.2	4.3	1.4
Legumes	9.1	0.6	2.1	6.6	1.4
A x V	p = .0571	ns	ns	Ns	ns

* Means having the same letter within the columns are not significantly different (alpha = .10 level) (Duncan's Multiple Range Test).

‡ ns = Not significantly different at the alpha = .10 level (Duncan's Multiple Range Test).

§ IBDU = Isobutyl Di-Urea as a slow release source of nitrogen

¶ SS = Sewage Sludge

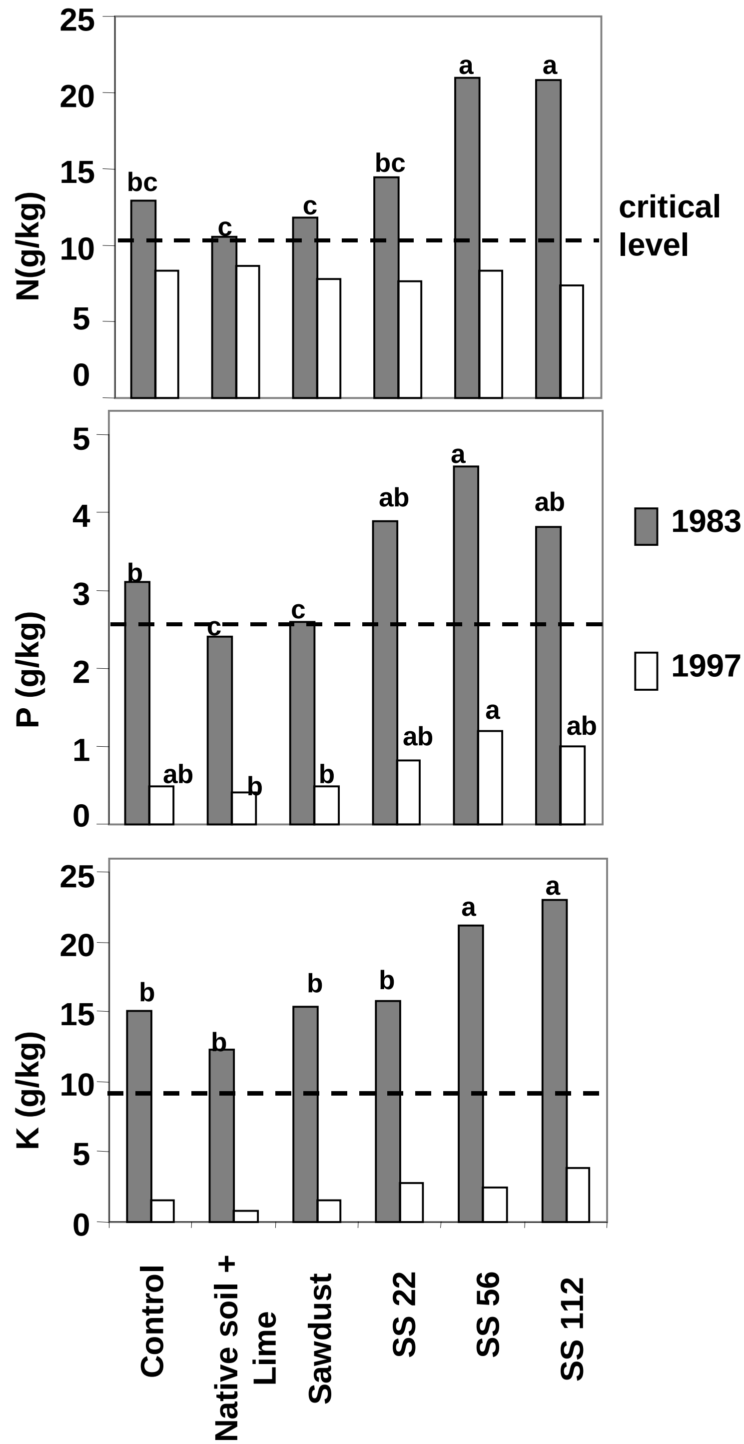

Figure V.6. Tall fescue foliar N, P, and K levels for 1983 and 1997 on amended mine soils.

Native forbs were collected as a third vegetation category. Phosphorus content of available forbs was consistently higher than grass and legume levels across all treatments. This illustrates the importance of considering the characteristics of native vegetation and its influence on nutrient availability in reclamation plans. Phosphorus content is closely coupled with N availability so the decline in each element may be related. P fixation by iron and manganese oxides can commonly reduce P availability for plants.

Foliar potassium levels were depressed to between 0.5 and 3.8 g kg^{-1} for grasses and 0.8 to 4.0 g kg^{-1} for legumes. These levels are lower than previously reported for the surface treatment experiment (Haering et al., 1990). Potassium availability is a function of weathering and clay content of soils. The ability of soil to replenish potassium depends on the soils buffering and cation exchange capacity (Marschner, 1995). However, pH for the study site is relatively high, between 6 and 7. Early plant uptake and biomass yields may have depleted the soils buffering capacity thus accounting for the reduced tissue concentrations. Similarly, Marschner (1995) reports that the nutrient interaction between N and K is closely intertwined. In many cases, potassium response depends on the quantity of N supplied.

The mineral status of calcium and magnesium were adequate with Ca exceeding Mg content. The ratio of Ca to Mg in tall fescue was approximately 3 to 1 for all treatments except the sawdust. The proportion of Ca to Mg in legumes was greater than for grasses. Overall Ca content increased and Mg decreased in herbaceous vegetation tissue samples during the past 10 yr (Table V.3).

Numerous studies on the application of sewage sludge have focused on plant uptake of micronutrients and the fate of heavy metals due to the concern of phytotoxicity and potential accumulation in the food chain. Mills and Jones (1996) report the survey averages of Fe, Mn, B, Cu, and Zn for tall fescue as 354, 71, 9, 34, and 47 mg kg^{-1}. Elemental concentrations of these micronutrients are relatively low to deficient across all treatments and there were no significant differences among treatments (Table V.4). Iron content was highest on the 112 and 22 Mg ha^{-1} SS treatments, while Mn levels were slightly elevated in the control and 56 Mg ha^{-1} sewage sludge treated plots.

Table V.4. Tissue concentrations in 1997/1998 of Fe, Mn, Cu, Zn, and B in herbaceous vegetation growing on amended mine soils.

	Elemental Concentrations				
Treatment	Fe	Mn	Zn	Cu	B
			Grass mg kg^{-1}		
Control	85 a*	109 a	13.6 ab	3.3 a	3.5 a
Topsoil	109 a	50 a	6.1 b	9.8 a	1.9 a
Sawdust + IBDU ‡	66 a	72 a	8.3 ab	2.1 a	1.5 a
SS (22 Mg ha^{-1})§	121 a	94 a	16.9 ab	2.3 a	2.9 a
SS (56 Mg ha^{-1})	92 a	101 a	18.0 ab	2.4 a	1.4 a
SS (112 Mg ha^{-1})	142 a	37 a	21.2 a	3.2 a	5.1 a
			Legumes mg kg^{-1}		
Control	81 a	61 a	11.5 ab	2.8 b	2.2 a
Topsoil	88 a	42 a	11.8 b	4.5 ab	5.4 a
Sawdust + IBDU	116 a	37 a	12.3 ab	2.5 b	4.1 a
SS (22 Mg ha^{-1})	117 a	58 a	18.7 ab	7.4 ab	5.4 a
SS (56 Mg ha^{-1})	68 a	42 a	21.5 a	5.1 ab	4.9 a
SS (112 Mg ha^{-1})	127 a	51 a	14.8 ab	13.3 a	5.1 a
Vegetation Type (V)	ns†	P= 0.0829	ns	ns	ns
Grass	104	51	18.3	8.6	5.1
Legumes	99	48	18.2	9.0	5.0
Treatment x Vegetation	ns	ns	ns	ns	ns

* Means having the same letter within the columns are not significantly different at the alpha = .10 level.
† ns = Not significantly different at the alpha = .10 level (Duncan's Multiple Range Test).
‡ IBDU = Isobutyl Di-Urea as a slow release source of nitrogen
§SS = Sewage Sludge

Predictive Model of Herbaceous Plant Response

Multiple regression analysis was used to test the effect of selected soil properties on plant nutrition and plant response. Table V.5 shows the variables tested and their level of significance. Nitrogen mineralization potential (N_o) was significantly correlated with herbaceous and tree response. Total Kjeldahl nitrogen and calcium were signivicant for herbaceous plant response. All other variables tested were not significantly correlated ($p = 0.10$) with plant (herbaceous and tree) response. These results suggest that the site is N deficient and that soil properties were not significantly different across treatments after 16 years in ways that would affect plant response.

Table V.5. Soil properties tested for correlation with plant response and levels of significance.

	Herbaceous Biomass (Mg/ha)	Tree Volume per Plot (cm^3)
	Pr > F	Pr > F
N mineralization	0.0610	0.0158
Total Kjeldahl N	0.0368	0.2942
Soil Organic Matter	0.4535	0.4384
Aggregate Stability	0.7676	0.3572
P	0.1404	0.3709
Light Fraction	0.5336	0.2435
Heavy Fraction	0.5454	0.7904
ECEC	0.2840	0.3503
Exchangeable Acidity	0.8289	0.8092
pH	0.3924	0.9665
Electrical Conductivity	0.4496	0.3010
Total Porosity	0.4263	0.9769
Bulk Density	0.2564	0.7062
Base Saturation	0.3066	0.4918
Ca	0.0779	0.3159
Mg	0.6547	0.5279
K	0.3769	0.4186

Soil properties and Effect of Vegetation Types

Soil organic matter content and associated properties are shown in Table V.6, and soil chemical properties are shown in Table V.7. Treatment effects are still apparent in these soil properties after 16 years, but the differences are not as pronounced as during the establishment of this study. Vegetation type did not have a significant effect on the majority of soil properties. There was a significant difference in organic matter content, effective cation exchange capacity (ECEC), sodium concentration, and soil pH.

Herbaceous vegetation and pitch x loblolly pine subplots contained 4.3 and 3.8% soil organic matter, respectively. ECEC was greater on tree subplots. Soil pH was slightly lower under pine trees than herbaceous vegetation. Nitrogen mineralization potential was not statistically significant, but nitrogen mineralization averaged 29mg kg^{-1} more under herbaceous vegetation than pine trees, which correlates with the difference in organic matter content of the two vegetation types. The difference in potentially mineralizable nitrogen suggests that the organic matter formed under herbaceous vegetation (grass and legumes) was more readily decomposable than under pine trees.

Table V.6. Total nitrogen, organic carbon, aggregate stability, and proportion of light fraction on whole soil basis for amended mine soils in southwest Virginia, collected March, 1998.

Treatment	OM	Light Fraction OM	Organic Carbon	Light Fraction Carbon	TKN	Light Fraction Nitrogen	1985 No	1998 No	Aggregate Stability	BD Fine Fraction	BD Whole Soil
	% of Whole Soil				g/kg		mg/kg		%	Mg m^{-3}	
Amendment (A)											
Control	3.9 b	2.0 bc*	2.3 b	0.7 ns	1.35 bc*	0.23 a	6 c	139 bc	57 ab	1.24 b*	1.51 a
Native soil	2.8 c	1.9 c	1.5 c	0.5	1.06 c	0.14 ab	10 c	102 d	52 b	1.38 a	1.51 a
Sawdust + IBDU †	4.3 ab	2.8 a	2.5 ab	0.9	1.32 bc	0.16 ab	74 b	119 cd	56 ab	1.12 bc	1.35 b
Sludge (22 Mg/ha)	4.1 ab	2.2 abc	2.4 ab	0.6	1.43 b	0.10 b	54 b	134 bcd	61 a	1.14 bc	1.41 ab
Sludge (56 Mg/ha)	4.5 ab	2.7 ab	2.7 ab	0.9	1.65 ab	0.15 b	72 b	160 ab	62 ab	1.13 bc	1.35 b
Sludge (112 Mg/ha)	4.8 a	2.9 a	2.9 a	1.0	1.90 a	0.15 b	125 a	183 a	65 a	1.09 c	1.27 b
Vegetation Type (V)											
Herbaceous	4.3 a	2.5 a	2.6 a	0.8 a	1.6 a	0.18 a		154 a	59 a	1.18 a	1.41 a
Trees	3.8 b	2.3 a	2.5 a	0.9 a	1.32 a	0.15 a		125 a	59 a	1.19 a	1.39 a
A x V	ns	p = .0093	ns	ns	ns	ns		ns	p = 0.0635	ns	ns

* Means within columns followed by the same letter were not significantly different (alpha = 0.10).

† IBDU = Isobutyl Di-urea applied as a source of slow release nitrogen

Table V.7. Concentration of macronutrients, effective cation exchange capacity, base saturation, P, and associated properties for amended mine soils under different vegetation covers. Collected March, 1998.

Treatment	ECEC	K	Mg	Ca	Na	Base Saturation	P	pH	Exchangeable Acidity	EC
	$cmol_{+}\ kg^{-1}$					%	mg/kg		$cmol_{(+)}\ kg^{-1}$	$uS\ cm^{-1}$
Amendment (A)										
Control	2.5 a	0.10 ns	0.33 ns	1.4 ns	0.038 a	74 b	31 bc	6.4 bc	0.7 a	49 b
Native soil	1.7 b	0.09	0.32	1.1	0.035 ab	92 a	31 bc	7.2 a	0.2 b	46 b
Sawdust + IBDU †	2.8 a	0.09	0.32	1.4	0.035 ab	72 b	23 c	6.3 c	0.9 a	49 b
Sludge (22 Mg/ha)	2.7 a	0.10	0.34	1.3	0.034 ab	65 b	37 b	6.3 c	0.9 a	45 b
Sludge (56 Mg/ha)	2.5 a	0.08	0.34	1.2	0.030 ab	65 b	57 a	6.4 c	1.0 a	50 b
Sludge (112 Mg/ha)	2.2 ab	0.09	0.31	1.2	0.029 b	74 b	60 a	6.6 b	0.6 a	60 a
Vegetation Type (V)										
Herbaceous	2.1 b	0.09 a	0.33 a	1.1 a	0.030 b	76 a	40	6.6 a	0.5 a	51 a
Trees	2.7 a	0.09 a	0.33 a	1.4 a	0.036 a	72 a	39	6.5 b	0.8 a	49 a
A x V	ns	ns	ns	ns	p = .0795	p = .0212	p = .0927	ns	ns	ns

* Means within columns followed by the same letter were not significantly different (alpha = 0.10).

† IBDU = Isobutyl Di-urea applied as a source of slow release nitrogen

An important component in the definition and assessment of soil quality is the ability of soil to serve as an amenable medium for plant growth (Karlen et al., 1992). In evaluating mine soil quality, an important characteristic is that the soils are storing, supplying, and cycling nutrients at levels and rates required by the plants for optimum function and productivity. Foliar concentrations of N, P, and K in tree and herbaceous vegetation were low to deficient which indicates that the quantity of these nutrients provided by the treated mine soils was insufficient, suggesting a decline in mine soil quality.

CONCLUSIONS

After 16 years, vegetation response to organic matter amendments shows that plant productivity and nutrient concentrations on these reclaimed mined soils declined through time. The growth response of pitch x loblolly pine and herbaceous vegetation on sawdust and sewage sludge-amended mine soils indicates that initial soil quality and fertility was favorable and improved crop production. After 16 years, vegetation and tree growth dynamics show that the effect of organic amendments lasted about four to eight growing seasons. Overall, the growth response shows initial soil fertility levels and early plant yields were not sustainable through time; after 15 growing seasons, plant productivity and system function appear to be more directly influenced by site quality and the effects of climate and edaphic features than initial amendments. Bioassays of tree and vegetation response to the 22 and 56 Mg ha^{-1} sewage sludge-treated mine soils show that tree growth and fescue biomass production was comparable.

The results suggest that closer monitoring of plant growth and soil fertility for agronomic crops and commercial forestry is required after the establishment period. Organic amendments improved initial soil fertility for crop establishment and bond release, but it appears a one-time application will have little or no-long lasting effect on plant productivity. Overall, plant growth was the same among treatments after 16 years.

SUMMARY AND RECOMMENDATIONS

Reestablishment of soil processes and functions is requisite for successful mined land reclamation and amelioration of degraded sites. Analysis of organic matter content, nitrogen mineralization, aggregate stability, and other mine soil quality indicators show that the treatment effect of organic amendments (sawdust and sewage sludge) declined over time and the soil system appears to be equilibrating. Organic matter levels declined on the sawdust and sewage sludge-treated mine soils, and increased on the native soil and unamended control treatments. Nitrogen mineralization potential, an important indicator of biological activity and soil quality, increased across all treatments, but most dramatically on the unamended control and native soil-treated mine soils. The increase and improvement indicates the importance of vegetation inputs to the recovery of soil functions. Effective cation exchange capacity of all treatments appears low, but comparable to native forest soils.

After 16 years, organic matter concentration, total organic nitrogen, potentially mineralizable nitrogen, aggregate stability, and ECEC were slightly lower on the native soil-treated mine soils, while bulk density was highest. The quantity of E, B, C, and Cr horizon material added as part of the treatment may have contributed to low soil fertility.

Tall fescue did not persist in the native soil plots due to the treatment's low soil fertility. Failure of the tall fescue sward resulted in the early invasion of sericea lespedeza. Mine soils, amended with sawdust and a time release nitrogen fertilizer, dramatically increased the initial stem volume of pitch x loblolly pine trees. After 16 years, stem volume of sawdust-treated plots was comparable to the 22 and 56 Mg ha^{-1}

sewage sludge-treated mine soils and contained 18% more stem volume than the unamended control treatment. For commercial forestry opportunities, pitch x loblolly pine trees growing in sawdust and sewage sludge-treated mine soils would appear to need additional fertility management after eight growing seasons. Mine soils, treated with sawdust, had a long-term negative effect on herbaceous vegetation production which suggests that the quality of organic matter is important.

A multiple regression analysis using key soil quality variables showed organic matter and aggregate stability were not significantly correlated with volume growth of pitch x loblolly pine trees. A prediction model using potentially mineralizable nitrogen as a dependent variable explained about 28% of the variation in tree volume per plot. Similarly, organic matter and aggregate stability were not correlated with total herbaceous biomass yield. Regression analysis showed that nitrogen mineralization potential was weakly correlated with tall fescue yields and accounted for only 11% of the variation in tall fescue production. Both N and P were severely deficient which may have masked the otherwise positive effects of key soil quality indicators.

After 16 years, evaluation of the effects of the three application rates of sewage sludge (22, 56, and 112 Mg ha^{-1}) on mine soil quality shows that increasing the application rate did not significantly affect organic matter content and most soil properties. The most important difference between these treatments was the level of available P. Sewage sludge treatments (56 and 112 Mg ha^{-1}) increased long-term P availability. Available P was about two times greater in sewage sludge-amended ($\geq$ 56 Mg ha^{-1}) mine soils than the 22 Mg ha^{-1} sewage sludge and other treatments. Bioassays of tree and vegetation response to the 22 and 56 Mg ha^{-1} sewage sludge-treated mine soils show that tree growth and fescue biomass production were comparable through time. These results suggest that the 22 Mg ha^{-1} sewage sludge treatment might benefit from the addition of phosphorus at the time of application; otherwise, the two treatments were similar in their long-term effect on mine soil quality and plant productivity. Tree growth indices show that sewage sludge could be applied at lower rates and have the same growth response. Where sewage sludge is readily available and economically feasible, sewage sludge treatments ($\geq$ 56 Mg ha^{-1}) appear to have a positive influence on the long-term maintenance of tall fescue cover on reclaimed mine soils. Comparing the sewage sludge and the unamended control treatments, it appears that at least 15 years are needed for vegetation inputs alone to have the same effect on mine soil quality as an application of 100 Mg ha^{-1} sewage sludge or similar organic material. In regard to the model proposed by Harrison et al. (1995) for the fate of organic matter through time, organic matter levels were initially enhanced during the first 5 yr of this study but appear to have returned to a level commensurate with the original nutrient content of the mine spoil and the climate and edaphic features for the area. Overall, 15 years after amendment, there appear to be no lasting soil quality and plant productivity improvements over that of the unamended control treatments.

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